Control of Air Pollution from New Motor Vehicles: Heavy-Duty Engine and Vehicle Standards

Regulatory Impact Analysis

United States
Environmental Protection
Agency

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Assessment and Standards Division
Office of Transportation and Air Quality
U.S. Environmental Protection Agency

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United States Environmental Protection Agency

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List of Acronyms

Acronym	Definition
°C	Degrees Celsius
µg	Microgram
µm	Micrometers
20xx$	U.S. Dollars in calendar year 20xx
A/C	Air Conditioning
ABT	Averaging, Banking and Trading
AC	Alternating Current
ACEA	European Automobile Manufacturers Association
ACES	Advanced Collaborative Emission Study
AECD	Auxiliary Emissions Control Device
AEO	Annual Energy Outlook
AES	Automatic Engine Shutdown
AFDC	Alternative Fuels Data Center
AHS	American Housing Survey
Al	Aluminum
Al2TiO5	Aluminum Titanate
AMOC	Atlantic Meridional Overturning Circulation
AMT	Automated Manual Transmission
ANL	Argonne National Laboratory
ANPR(M)	Advanced Notice of Proposed Rulemaking
APU	Auxiliary Power Unit
AQ	Air Quality
AQCD	Air Quality Criteria Document
AR4	Fourth Assessment Report
ARB	California Air Resources Board
ARB HHDDT	California Air Resources Board Heavy Heavy-Duty Diesel Test
ASC	Ammonia Slip Catalyst
ASL	Aggressive Shift Logic
ASM	Annual Survey of Manufacturers
ASTM	ASTM International, formerly American Society for Testing and Materials
ASTM	ASTM International, formerly known as American Society for Testing and Materials
AT	Automatic Transmissions
ATA	American Trucking Association
ATIS	Automated Tire Inflation System
ATRI	Alliance for Transportation Research Institute
ATSDR	Agency for Toxic Substances and Disease Registry
ATUS	American Time Use Survey
Avg	Average
B100	1 methyl-ester biodiesel fuel
B20	0.2 biodiesel blended with 0.8 petroleum distilate diesel fuel
BenMAP	Benefits Mapping and Analysis Program
BETP	Bleed Emissions Test Procedure
BEV	Battery Electric Vehicle
bhp	Brake Horsepower
bhp-hr	Brake Horsepower Hour
BLS	Bureau of Labor Statistics
BMEP	Brake Mean Effective Pressure
BSFC	Brake Specific Fuel Consumption
BTS	Bureau of Transportation Statistics
BTS	Bureau of Labor Statistics

BTU	British Thermal Unit
Ca	Calcium
CAA	Clean Air Act
CAAA	Clean Air Act Amendments
CaCO3	Calcium Carbonate
CAD/CAE	Computer Aided Design And Engineering
CAE	Computer Aided Engineering
CAFE	Corporate Average Fuel Economy
CAN	Controller Area Network
CARB	California Air Resources Board
CaSO4	Calcium Sulfate
CBI	Confidential Business Information
CCP	Coupled Cam Phasing
CCSP	Climate Change Science Program
CDA	Cylinder Deactivation
CDC	Centers for Disease Control
CDPF	Catalyzed Diesel Particulate Filter
CFD	Computational Fluid Dynamics
CFR	Code of Federal Regulations
CH_4	Methane
CI	Compression-ignition
CILCC	Combined International Local and Commuter Cycle
CIPM	International Committee for Weights and Measures (Bureau International des Poids et Mesures)
CITT	Chemical Industry Institute of Toxicology
CMAQ	Community Multiscale Air Quality
CNG	Compressed Natural Gas
CO	Carbon Monoxide
CO_2	Carbon Dioxide
CO_2eq	CO_2 Equivalent
COFC	Container-on-Flatcar
COI	Cost of Illness
COPD	Chronic Obstructive Pulmonary Disease
CoV	Coefficient of Variation
CPS	Cam Profile Switching
CPSI	Cells per Square Inch
CRC	Coordinating Research Council
CRGNSA	Columbia River Gorge National Scenic Area
CRR	Rolling Resistance Coefficient
CS	Climate Sensitivity
CSI	Cambridge Systematics Inc.
CSS	Coastal Sage Scrub
CSV	Comma-separated Values
Cu	Copper
CuO	Copper(II) oxide or cupric oxide
CVD	Cardiovascular Disease
CVT	Continuously-Variable Transmission
CW	Curb Weight
D/UAF	Downward and Upward Adjustment Factor
DAAAC	Diesel Aftertreatment Accelerated Aging Cycle
DARAP	Diesel Aftertreatment Rapid Aging Protocol
DCP	Dual Cam Phasing
DCT	Dual Clutch Transmission

DE	Diesel Exhaust
DEAC	Cylinder Deactivation
DEER	Diesel Engine-Efficiency and Emissions Research
DEF	Diesel Exhaust Fluid
deSOx	Removal of sulfur oxide compounds
DF	Deterioration Factor
DHHS	U.S. Department of Health and Human Services
Diesel HAD	Diesel Health Assessment Document
DMC	Direct Manufacturing Costs
DO	Dissolved Oxygen
DOC	Diesel Oxidation Catalyst
DOD	Department of Defense
DOE	Department of Energy
DOHC	Dual Overhead Camshaft Engines
DOT	Department of Transportation
DPF	Diesel Particulate Filter
DPM	Diesel Particulate Matter
DR	Discount Rate
RIA	Draft Regulatory Impact Analysis
DVVL	Discrete Variable Valve Lift
EAS	Exhaust Aftertreatment System
EC	European Commission
EC	Elemental Carbon
EC	Economic Census
ECM	Electronic Control Module
ED	Emergency Department
EERA	Energy and Environmental Research Associates
EFR	Engine Friction Reduction
EGR	Exhaust Gas Recirculation
EHPS	Electrohydraulic Power Steering
EIA	Energy Information Administration (part of the U.S. Department of Energy)
EISA	Energy Independence and Security Act
EIVC	Early Intake Valve Closing
EMS-HAP	Emissions Modeling System for Hazardous Air Pollution
EO	Executive Order
EPA	Environmental Protection Agency
EPMA	electron probe microanalysis
EPS	Electric Power Steering
ERG	Eastern Research Group
ERM	Employment Requirements Matrix
ESC	Electronic Stability Control
ETC	Electronic Throttle Control
ETW	Estimated Test Weight
EV	Electric Vehicle
F	Frequency
FCEV	Fuel Cell Electric Vehicle
Fe	Iron
FEL	Family Emission Limit
FET	Federal Excise Tax
FEV1	Functional Expiratory Volume
FHWA	Federal Highway Administration
FIA	Forest Inventory and Analysis
FMCSA	Federal Motor Carrier Safety Administration

FOH	Fuel Operated Heater
FR	Federal Register
FRM	Final Rulemaking
FTP	Federal Test Procedure
FVC	Forced Vital Capacity
g	Gram
g/s	Gram-per-second
g/ton-mile	Grams emitted to move one ton (2000 pounds) of freight over one mile
gal	Gallon
gal/1000 ton-mile	Gallons of fuel used to move one ton of payload (2,000 pounds) over 1000 miles
GCAM	Global Change Assessment Model
GCW	Gross Combined Weight
GCWR	Rated Gross-combined Weight (vehicle + trailer)
GDI	Gasoline Direct Injection
GDI	Gasoline Direct Injection
GDP	Gross Domestic Product
GEM	Greenhouse gas Emissions Model
GEOS	Goddard Earth Observing System
GHG	Greenhouse Gas
GIFT	Geospatial Intermodal Freight Transportation
GPF	Gasoline Particulate Filter
GREET	Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation
GSF1	Generic Speed Form one
GUI	Graphical User Interface
GVWR	Gross Vehicle Weight Rating
GVWR	Rated Gross Vehicle Weight
GWP	Global Warming Potential
H2O	Water
HABs	Harmful Algal Blooms
HAD	Diesel Health Assessment Document
HC	Hydrocarbon
HD	Heavy-Duty
HDDE FTP	Heavy-Duty Diesel Engine Federal Test Procedure
HDE	Heavy-Duty Engine
HDOE FTP	Heavy-Duty Otto-Cycle Engine Federal Test Procedure
HDT	Heavy-Duty Truck
HDUDDS	Heavy Duty Urban Dynamometer Driving Cycle
HDV	Heavy-Duty Vehicle
HEG	High Efficiency Gearbox
HEI	Health Effects Institute
HES	Health Effects Subcommittee
HEV	Hybrid Electric Vehicle
HFC	Hydrofluorocarbon
HFET	Highway Fuel Economy Dynamometer Procedure
HHD	Heavy Heavy-Duty
HHDDT	Highway Heavy-Duty Diesel Transient
HNCO	Iso-cyanic Acid
hp	Horsepower
hrs	Hours
HRV	Heart Rate Variability
HSC	High Speed Cruise Duty Cycle
HTUF	Hybrid Truck User Forum

HWFE	Highway Fuel Economy Drive Cycle
hz	Hertz
IARC	International Agency for Research on Cancer
IATC	Improved Automatic Transmission Control
IC	Indirect Costs
ICCT	International Council on Clean Transport
ICD	International Classification of Diseases
ICF	ICF International
ICM	Indirect Cost Multiplier
ICP	Intake Cam Phasing
ICP-MS	nductively coupled plasma mass spectrometry
IMAC	Improved Mobile Air Conditioning
IMPROVE	Interagency Monitoring of Protected Visual Environments
IPCC	Intergovernmental Panel on Climate Change
IRAF	infrequent regeneration factor
IRFA	Initial Regulatory Flexibility Analysis
IRIS	Integrated Risk Information System
ISA	Integrated Science Assessment
JAMA	Journal of the American Medical Association
k	Thousand
K	Potassium
kg	Kilogram
KI	kinetic intensity
km	Kilometer
km/h	Kilometers per Hour
kW	Kilowatt
L	Liter
LA92	Inventory development dynamometer driving schedule
lb	Pound
LD	Light-Duty
LDT	Light-Duty Truck
LHD	Light Heavy-Duty
LIVC	Late Intake Valve Closing
LLC	Low Load Cycle
LLNL	Lawrence Livermore National Laboratory's
LPG	Liquified Petroleum Gas
LRR	Lower Rolling Resistance
LSC	Low Speed Cruise Duty Cycle
LT	Light Trucks
LTCCS	Large Truck Crash Causation Study
LUB	Low Friction Lubes
LUC	Land Use Change
m^2	Square Meters
m^3	Cubic Meters
MAGICC	Model for the Assessment of Greenhouse-gas Induced Climate Change
MCF	Mixed Conifer Forest
MD	Medium-Duty
MDPV	Medium-Duty Passenger Vehicle
MECA	Manufacturers of Emissions Control Association
mg	Milligram
Mg	Magnesium
Mg(OH)2	Magnesium Hydroxide
mg/hp-hr	Milligrams per horsepower-hour

MHD	Medium Heavy-Duty
MHEV	Mild Hybrid
mi	mile
min	Minute
MM	Million
MMBD	Million Barrels per Day
MMT	Million Metric Tons
Mn	Manganese
MOVES	MOtor Vehicle Emissions Simulator
MP-AES	Microwave Plasma Atomic Emission Spectroscopy
mpg	Miles per Gallon
mph	Miles per Hour
MRL	Minimal Risk Level
MSAT	Mobile Source Air Toxic
MT	Manual Transmission
MTS	Maximum Test Speed
MW	Megawatt
MY	Model Year
N2	Molecular Nitrogen
N2O	Nitrous Oxide
Na	Sodium
NA	Not Applicable
NAAQS	National Ambient Air Quality Standards
NACFE	North American Council for Clean Freight Efficiency
NAFA	National Association of Fleet Administrators
NAICS	North American Industry Classification System
NAS	National Academy of Sciences
NASTC	National Association of Small Trucking Companies
NATA	National Air Toxic Assessment
NCAR	National Center for Atmospheric Research
NCI	National Cancer Institute
NCLAN	National Crop Loss Assessment Network
NEC	Net Energy Change Tolerance
NEI	National Emissions Inventory
NEMS	National Energy Modeling System
NEPA	National Environmental Policy Act
NESCAUM	Northeastern States for Coordinated Air Use Management
NESCCAF	Northeast States Center for a Clean Air Future
NESHAP	National Emissions Standards for Hazardous Air Pollutants
NH3	Ammonia
NHS	National Highway System
NHTSA	National Highway Traffic Safety Administration
NiMH	Nickel Metal-Hydride
NIOSH	National Institute of Occupational Safety and Health
NIST	National Institute for Standards and Technology
Nm	Newton-meters
NMHC	Nonmethane Hydrocarbons
NMMAPS	National Morbidity, Mortality, and Air Pollution Study
NO	Nitric Oxide
NO_2	Nitrogen Dioxide
NOAA	National Oceanic and Atmospheric Administration
NO_X	Oxides of Nitrogen
NPRM	Notice of Proposed Rulemaking

NPV	Net Present Value
NRC	National Research Council
NRC-CAN	National Research Council of Canada
NREL	National Renewable Energy Laboratory
NTE	Not-to-exceed
NTEA	National Truck and Equipment Association
NTP	National Toxicology Program
NVH	Noise Vibration and Harshness
O&M	Operating and maintenance
O_3	Ozone
OAQPS	Office of Air Quality Planning and Standards
OBD	Onboard diagnostics
OC	Organic Carbon
OE	Original Equipment
OEHHA	Office of Environmental Health Hazard Assessment
OEM	Original Equipment Manufacturer
OHV	Overhead Valve
OMB	Office of Management and Budget
OOIDA	Owner-Operator Independent Drivers Association
OPEC	Organization of Petroleum Exporting Countries
ORD	EPA's Office of Research and Development
ORNL	Oak Ridge National Laboratory
ORVR	On-Board Refueling Vapor Recovery
ORVR	Onboard refueling vapor recovery
OTAQ	Office of Transportation and Air Quality
P	Phosphorus
Pa	Pascal
PAH	Polycyclic Aromatic Hydrocarbons
PCV	Positive Crankcase Ventilation
PEF	Peak Expiratory Flow
PEMFC	Proton-Exchange Membrane Fuel Cell
PEMS	Portable Emissions Monitoring System
PFI	Port Fuel Injection
PFI	Port Fuel Injection
PGM	Platinum Group Metal
PHEV	Plug-in Hybrid Electric Vehicles
PLT	Production-line testing
PM	Particulate Matter
PM_{10}	Coarse Particulate Matter (diameter of 10 µm or less)
$PM_{2.5}$	Fine Particulate Matter (diameter of 2.5 µm or less)
POM	Polycyclic Organic Matter
Ppb	Parts per Billion
Ppm	Parts per Million
Psi	Pounds per Square Inch
PTO	Power Take Off
R&D	Research and Development
RBM	Resisting Bending Moment
REL	Reference Exposure Level
RESS	Rechargeable Energy Storage System
RFA	Regulatory Flexibility Act
RfC	Reference Concentration
RFS2	Renewable Fuel Standard 2
RIA	Regulatory Impact Analysis

RMC-SET	Ramped Modal Cycle Supplementary Emissions Test
RPE	Retail Price Equivalent
RPM	Revolutions per Minute
RSWT	Reduced-Scale Wind Tunnel
S	Second
S	Sulfur
SAB	Science Advisory Board
SAB-HES	Science Advisory Board - Health Effects Subcommittee
SAE	SAE International, formerly Society of Automotive Engineers
SAR	Second Assessment Report
SAV	Submerged Aquatic Vegetation
SBA	Small Business Administration
SBREFA	Small Business Regulatory Enforcement Fairness Act
SCR	Selective Catalyst Reduction
SEA	Selective enforcement audit
SER	Small Entity Representation
SET	Supplemental Emission Test
SGDI	Stoichiometric Gasoline Direct Injection
SHED	Sealed Housing Evaporative Determination
SHEV	Strong Hybrid Vehicles
SI	Spark-Ignition
SiC	Silicon Carbide
SIDI	Spark Ignition Direct Injection
SO_2	Sulfur Dioxide
SOA	Secondary Organic Aerosol
SOC	State of Charge
SOFC	Solid Oxide Fuel Cells
SOHC	Single Overhead Cam
SOx	Sulfur Oxides
SO_X	Oxides of Sulfur
SPR	Strategic Petroleum Reserve
SSZ-13	A Chabazite-type Aluminosilicate ABC-6 Zeolite
STB	Surface Transportation Board
Std.	Standard
STP	Scaled Tractive Power
SUV	Sport Utility Vehicle
SVOC	Semi-Volatile Organic Compound
SwRI	Southwest Research Institute
TAR	Technical Assessment Report
TC	Total Costs
TCO	Total Cost of Ownership
TCp	Total Cost package
TDS	Turbocharging And Downsizing
THC	Total Hydrocarbon
TIAX	TIAX LLC
TMC	Technology & Maintenance Council
TOFC	Trailer-on-Flatcar
Ton-mile	One ton (2000 pounds) of payload over one mile
TRBDS	Turbocharging and Downsizing
TRU	Trailer Refrigeration Unit
TSD	Technical Support Document
TSS	Thermal Storage
TW	Test Weight

TWC	Three-Way Catalyst
U.S.	United States
U/DAF	Upward and Downward Adjustment Factor
UBE	Useable battery energy
UCT	Urban Creep and Transient Duty Cycle
UFP	Ultra Fine Particles
ULSD	Ultra-low sulfur diesel
URE	Unit Risk Estimate
USDA	United States Department of Agriculture
USGCRP	United States Global Change Research Program
UV	Ultraviolet
UV-b	Ultraviolet-b
VGT	Variable-geometry Turbine
VIN	Vehicle Identification Number
VIUS	Vehicle Inventory Use Survey
VMT	Vehicle Miles Traveled
VOC	Volatile Organic Compound
VSL	Vehicle Speed Limiter
VTEC-E	Variable Valve Timing & Lift Electronic Control-Economy
VTRIS	Vehicle Travel Information System
VVL	Variable Valve Lift
VVT	Variable Valve Timing
WACAP	Western Airborne Contaminants Assessment Project
WHR	Waste Heat Recovery
WHTC	World Harmonized Transient Cycle
WHVC	World Harmonized Vehicle Cycle
WRF	Weather Research Forecasting
WTP	Willingness-to-Pay
WTVC	World Wide Transient Vehicle Cycle
WVU	West Virginia University
Zn	Zinc
ZSM-5	Zeolite Socony Mobil–5, an Aluminosilicate Pentasil Zeolite within the family of zeolites

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Executive Summary

The Environmental Protection Agency (EPA) is finalizing new standards and changes to the heavy-duty highway engine and vehicle emissions control program in order to reduce emissions of oxides of nitrogen (NO_X), particulate matter (PM), hydrocarbons (HC), and carbon monoxide (CO). These emissions contribute to ozone and PM and their resulting threat to public health, which includes premature death, respiratory illness (including childhood asthma), cardiovascular problems, and other adverse health impacts.

This RIA is generally organized to provide overall background information, methodologies, and data inputs, followed by results of the various analyses. A summary of each chapter of the RIA follows.

Chapter 1 describes key technologies that manufacturers could use to meet more stringent emissions standards for NO_X, PM, HC, and CO. The chapter focuses on technologies specific to compression-ignition engines and spark-ignition engines, and also discusses fuel considerations.

Chapter 2 describes the existing test procedures as well as the development process for the final test procedures for spark- and compression-ignition engine compliance. This includes the determination of emissions from both engines and hybrid powertrains, as well as the development of new duty cycles.

Chapter 3 describes the technology feasibility demonstration programs, including engine technologies and emission control strategies for reducing NO_X, PM, NMHC, and CO. The technologies presented represent potential ways that the industry could meet the final stringency levels, and they provide the basis for the technology costs and benefits analyses.

Chapter 4 presents a discussion of the health effects associated with exposure ambient concentrations of ozone, PM, NO_2, CO, and air toxics. The discussion of health impacts is mainly focused on describing the effects of air pollution on the population in general. Additionally, children are recognized to have increased vulnerability and susceptibility related to air pollution and other environmental exposures; this and effects for other vulnerable and susceptible groups are discussed. The chapter also discusses the environmental effects associated with pollutants affected by this final rulemaking, specifically PM, ozone, NO_X and air toxics.

Chapter 5 presents our analysis of the national emissions impacts of the final program for calendar years 2027 through 2045. In this chapter we quantify emissions from NO_X, volatile organic compounds (VOC), $PM_{2.5}$, CO, and others. The onroad national emissions inventories were estimated using the latest public version of EPA's Motor Vehicle Emission Simulator (MOVES) model (MOVES3). Table ES-1 summarizes the projected reductions in heavy-duty emissions from the final rule in 2045. In addition to describing the national emission inventories, this chapter describes the methods used to estimate the spatially and temporally-resolved emission inventories used to support the air quality modeling analysis documented in Chapter 6.

Table ES-1: Projected Heavy-Duty Emission Reductions in 2045 from the Final Program

Pollutant	Percent Reduction in Highway Heavy-duty Emissions
NO_X	48%
Primary $PM_{2.5}$	8%
VOC	23%
CO	18%

Chapter 6 presents information on air quality, including a discussion of current air quality, details related to the methodology used for the air quality modeling analysis, and results from the air quality modeling analysis. When feasible, we conduct full-scale photochemical air quality modeling to accurately project levels of criteria and air toxic pollutants, because the atmospheric chemistry related to ambient concentrations of $PM_{2.5}$, ozone, and air toxics is very complex. Air quality modeling was performed for the proposed rule using emission reductions that compare well with the emission reductions estimated for the final rule, and it demonstrated improvements in concentrations of air pollutants. Given the similar structure of the proposed and final programs, we expect consistent geographic distribution of emissions reductions and modeled improvements in air quality, and that the air quality modeling conducted at the time of proposal adequately represents the final rule. Specifically, we expect this rule will decrease ambient concentrations of air pollutants, including significant improvements in ozone concentrations in 2045 as demonstrated in the air quality modeling analysis. Our analysis indicates that the largest predicted improvements in both ozone and $PM_{2.5}$ are estimated to occur in areas with the worst baseline air quality, and that a substantially larger number of people of color are expected to reside in these areas. An expanded analysis of the air quality impacts experienced by specific race and ethnic groups found that non-Hispanic Blacks will receive the greatest improvement in $PM_{2.5}$ and ozone concentrations as a result of the standards. The emission reductions provided by the final standards will be important in helping areas attain and maintain the National Ambient Air Quality Standards (NAAQS) and prevent future nonattainment. We also expect reductions in ambient $PM_{2.5}$, NO_2 and CO due to this rule. Although the spatial resolution of the air quality modeling is not sufficient to quantify it, this rule's emission reductions will also reduce air pollution in close proximity to major roadways, where concentrations of many air pollutants are elevated. In addition, the final standards are expected to result in improvements in nitrogen deposition and visibility.

Chapter 7 presents estimates of the costs associated with the emissions-reduction technologies that manufacturers could add in response to the final program. We present these not only in terms of the upfront technology costs per engine as presented in Chapter 3 of this RIA, but also how those costs would change in the years following implementation. We present the costs associated with the final program elements of extended regulatory useful life and warranty. These technology costs are presented in terms of direct manufacturing costs and associated indirect costs such as warranty and research and development (R&D). The analysis also includes estimates of the possible operating costs associated with the final program-- the addition of new technology and extension of warranty and useful life periods. All costs are presented in 2017 dollars unless noted otherwise. Table ES-2 presents the technology costs, operating costs and the sum of the two for the final program in 2045.

Table ES-2: Total Program Costs: Undiscounted Annual Costs in 2045 and Annualized Costs through 2045 at 3% and 7% Discount Rates (Billions of 2017 dollars)

	Total Technology Costs	Total Operating Costs	Sum
2045 Annual	$4.1	$0.62	$4.7
Present Value, 3%	$53	$1.4	$55
Present Value, 7%	$38	$0.6	$39
Annualized, 3%	$3.7	$0.099	$3.8
Annualized, 7%	$3.7	$0.058	$3.8

Chapter 8 describes the methods used to estimate health benefits from reducing concentrations of ozone and $PM_{2.5}$. For the final rulemaking, we have quantified and monetized health impacts in 2045, representing projected impacts associated with a year when the program will be fully implemented and when most of the regulated fleet will have turned over. We also discuss unquantified benefits associated with the standards that, if quantified and monetized, will increase the total monetized benefits. Overall, we estimate that the final program will lead to a substantial decrease in adverse $PM_{2.5}$- and ozone-related health impacts in 2045. Table ES-3 presents our estimates of total monetized benefits for the final program.

Table ES-3: 2045 Annual Value, Present Value and Equivalent Annualized Value of Benefits of the Final Program (billions, 2017$)[a,b]

	3% Discount	7% Discount
2045	$12 - $33	$10 - $30
Present Value (2027-2045)	$91 - $260	$53 - $150
Annualized Value	$6.3 - $18	$5.1 - $14

[a] All benefits estimates are rounded to two significant figures; numbers may not sum due to independent rounding. The range of benefits (and net benefits) in this table are two separate estimates and do not represent lower- and upper-bound estimates, though they do reflect a grouping of estimates that yield more and less conservative benefits totals. The costs and benefits in 2045 are presented in annual terms and are not discounted. However, all benefits in the table reflect a 3 percent and 7 percent discount rate used to account for cessation lag in the valuation of avoided premature deaths associated with long-term exposure.

[b] The benefits associated with the standards presented here do not include the full complement of health and environmental benefits that, if quantified and monetized, would increase the total monetized benefits.

Chapter 9 compares the estimated range of total monetized health benefits to total costs associated with the final program. This chapter also presents the range of monetized net benefits (benefits presented in Chapter 8 minus costs presented in Chapter 7) associated with the same scenarios (see Table ES-4).

The health- and environmental-related effects associated with heavy-duty vehicle and engine emissions are a classic example of an externality-related market failure. An externality occurs when one party's actions impose uncompensated costs on another party. The final standards will help correct this market failure. EPA expects that implementation of the final rule will provide society with a substantial net gain in welfare, notwithstanding the health and other benefits we were unable to quantify (see RIA Chapter 8.8 for more information about unquantified benefits).[A]

Table ES-4: 2045 Annual Value, Present Value and Equivalent Annualized Value of Costs, Benefits and Net Benefits of the Final Program (billions, 2017$)[a,b]

		3% Discount	7% Discount
2045	Benefits	$12 - $33	$10 - $30
	Costs	$4.7	$4.7
	Net Benefits	$6.9 - $29	$5.8 - $25
Present Value	Benefits	$91 - $260	$53 - $150
	Costs	$55	$39
	Net Benefits	$36 - $200	$14 - $110
Equivalent Annualized Value	Benefits	$6.3 - $18	$5.1 - $14
	Costs	$3.8	$3.8
	Net Benefits	$2.5 - $14	$1.3 - $11

EPA is required by Executive Order (E.O.) 12866 to estimate the benefits and costs of major new pollution control regulations. At the same time, EPA notes that this analysis is for purposes of Executive Order 12866, rather than for purposes of showing that the final rule satisfies the requirements of the Clean Air Act section 202(a). The Clean Air Act does not require a weighing of costs and benefits in determining what standards are achievable, and EPA did not do so in determining what standards to adopt.

Chapter 10 provides an economic analysis of the impacts of the final standards on vehicle sales and employment. This rulemaking is considered economically significant, because it is expected to have an annual impact on the economy of $100 million or more but is not expected to have measurable inflationary or recessionary effects. This chapter presents a peer-reviewed analysis to develop a relationship between estimated changes in vehicle price due to a new regulation and corresponding changes in vehicle sales (i.e., pre- and low-buy elasticities). In RIA Chapter 10.1 we outline an approach to quantify potential impacts on vehicle sales due to new emission standards; we also illustrate how this method could be used to estimate pre-and low-buy as a function of the estimated costs of this final rule. Our example results for the final standards suggest pre- and low-buy for Class 8 trucks may range from zero to approximately 2 percent increase in sales over a period of up to 8 months before the 2027 standards begin (pre-buy), and a decrease in sales from zero to just under three percent over a period of up to 12 months after the 2027 standards begin (low-buy). Our illustrative analysis suggests that if pre-buy and low-buy occur, the difference would be very slight and short-lived; we do not expect long-term fleet turnover impacts from pre-buy or low-buy, including effects on average fleet age. The employment assessment focuses on the motor vehicle manufacturing and the motor vehicle parts manufacturing sectors, with some assessment of impacts on additional sectors likely to be most affected by the standards. The employment assessment includes EPA's qualitative and quantitative estimates of the partial employment impacts of this rule on regulated industries and an examination of employment impacts in some closely related sectors.

[A] EPA does not expect the omission of unquantified benefits to impact the Agency's evaluation of the final program since unquantified benefits generally scale with the emissions impacts of the final program.

Chapter 11 presents our analysis of the potential impacts of the final rule on small entities that will be subject to the highway heavy-duty engine and vehicle provisions of this final rule. These are: heavy-duty alternative fuel engine converters and heavy-duty secondary vehicle manufacturers. Other entities that will be subject to the rule are either not small (e.g., engine and incomplete vehicle manufacturers) or are not expected to incur any burden from the final rule (e.g., in sectors other than highway heavy-duty engines and vehicles). Our analysis estimates that no small entities will experience an impact of 3% or more of their annual revenue as a result of our final rule.

Chapter 1 Technology to Control Emissions from Heavy-Duty Engines

This chapter describes key technologies that manufacturers are likely to use to meet more stringent emissions standards for oxides of nitrogen (NO_X), particulate matter (PM), hydrocarbons (HC), and carbon monoxide (CO). The chapter introduces technologies specific to compression-ignition engines and spark-ignition engines, and also discusses fuel considerations, advanced powertrain technologies, and emission monitoring technologies that may apply across engine types.

1.1 Compression-Ignition Engine Technologies

The following sections describe the compression-ignition engine technologies that we considered for reducing criteria pollutant emissions as part of this final rule. Many of the technologies are described with respect to diesel fuel, but they are expected to be broadly applicable to all fuels used in compression-ignition engines. Our compression-ignition engine feasibility demonstration for this final rule is based on some of the technologies presented in this section.

Chapter 2 of this RIA describes the updated test procedures for diesel-ignition engine certification. Chapter 3 describes the compression-ignition engine feasibility demonstration program, including a description of the specific technology packages we evaluated, the effectiveness of those technologies relative to the final standards and corresponding test procedures, and our projected direct manufacturing cost of those technologies.

1.1.1 **Current Catalyst Technologies**

This section addresses technologies that, based on our current understanding, are anticipated to be available in the 2024 to 2030 timeframe to reduce emissions and ensure robust in-use compliance. The following discussion introduces the technologies and emission reduction strategies we considered for the final rulemaking, including thermal management technologies that can be used to better achieve and maintain adequate catalyst temperatures, and the next generation of catalyst configurations and formulations that will improve catalyst performance across a broader range of engine operating conditions.

Modern diesel engines rely heavily upon catalytic exhaust aftertreatment systems (EAS) to meet exhaust emission standards. Current (MY2018-2022) heavy-duty diesel EAS consist of a diesel oxidation catalyst (DOC) followed by a catalyzed diesel particulate filter (CDPF), a urea injector, a urea mixer or other decomposition component, and then one or more selective catalytic reduction (SCR) monolithic substrates (Figure 1-1, Figure 1-2). Such systems are capable of reducing PM emissions by greater than 95% under most operating conditions and are capable of reducing NO_X emissions by 90 to 98% at exhaust temperatures above approximately 250 °C.

Unreacted ammonia downstream of the SCR is typically referred to as "ammonia slip". An ammonia slip catalyst (ASC) can be zone-coated onto the outlet of the rearmost SCR substrate (the case for most LHDDE and MHDDE and some HHDDE applications) or can be coated onto a separate catalyst substrate (some HHDDE applications) and uses platinum-group-metal (PGM) exchanged zeolites to promote reaction of ammonia remaining downstream of the SCR catalysts. Ammonia is an important air toxic compound and can also contribute to secondary PM formation. The use of closed-loop feedback electronic control of urea dosing using zirconia NO_X sensors for NO_X and ammonia feedback and the use of ASC together can reduce ammonia emissions from modern EAS-equipped heavy-duty diesel engines to less than 4 mg/bhp-hr.[1] Some LHDDE applications using chassis dynamometer certification place the urea injector and SCR between the DOC and CDPF or combine SCR and CDPF functionality into one catalyst, sometimes referred to as selective catalytic reduction on filter (SCRF).

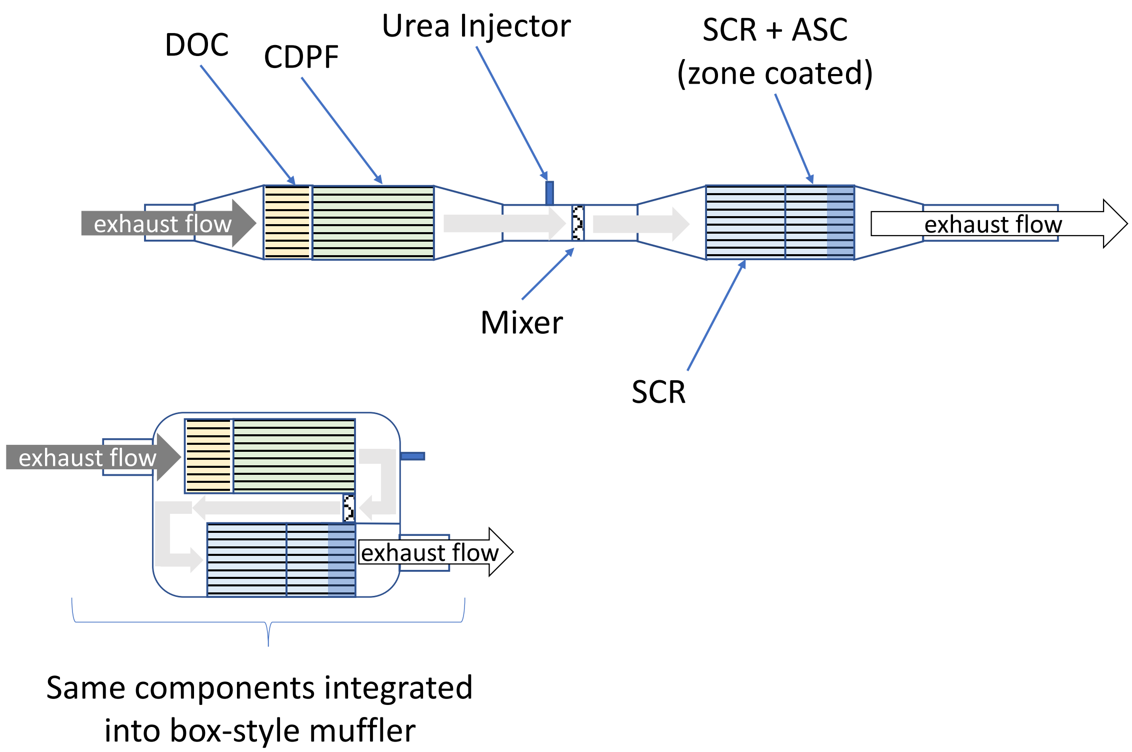

Figure 1-1: Functional schematic showing relative positioning of exhaust emission control components arranged within an in-line exhaust system (top) and integrated into a box-style system (bottom).

Figure 1-2: Integrated/series heavy-duty truck exhaust emission control system from Cummins Emission Solutions (top) and box-style system from Eberspächer (bottom), with cut-away showing some of the internal components (bottom right).[A]

The DOC, SCR, and ASC typically use cordierite ceramic flow-through monolithic substrates that are wash-coated with active materials. The CDPF uses a wall-flow substrate made of either cordierite, silicon carbide (SiC), or aluminum titanate (Al_2TiO_5) for exhaust filtration (or "trapping") of particulate matter that is coated with active materials. Alternating cells of the wall-flow substrate are blocked, forcing the exhaust to flow through the porous substrate wall. The particulate matter, consisting primarily of elemental carbon soot, is filtered from the exhaust

[A] Disclaimer: Any mention of trade names, manufacturers or products does not imply an endorsement by the United States Government or the U.S. Environmental Protection Agency. EPA and its employees do not endorse any commercial products, services, or enterprises.

flow onto and within the wall of the CDPF and can then be oxidized to CO_2 using either passive regeneration with nitrogen dioxide (NO_2) or active regeneration in the presence of excess oxygen in the exhaust. Passive regeneration of the CDPF depends on oxidation of a fraction of nitric oxide (NO) emissions in the exhaust to NO_2. Soot oxidation using NO_2 occurs at exhaust temperatures of approximately 250 °C, thus it does not require external heat addition to the exhaust under most operating conditions. Active regeneration using excess oxygen in the exhaust occurs at exhaust temperatures above 500 °C – 600 °C[B], and thus requires adding heat to the exhaust. This can be accomplished using one of several different approaches:

- Late, in-cylinder, post-injection of fuel after the primary combustion event and subsequent heat addition from the exothermic reaction of the excess fuel over the DOC and CDPF.
- Direct injection of diesel fuel into the exhaust, with exothermic reaction of the fuel over the DOC and CDPF.
- Use of an exhaust-integrated, external combustion burner system.

Selective catalytic reduction (SCR) reduces nitrogen oxides NO_X (consisting of both NO and NO_2) to N_2 and water by using ammonia (NH_3) as the reducing agent. The SCR catalyst coatings used for post-2010 model-year heavy-duty diesel applications in the U.S. are typically copper (Cu) exchanged or iron (Fe) exchanged zeolites, (e.g., Fe-ZSM-5), and most SCR coatings for recent (MY2018-2020) applications are one of several different Cu and/or Fe-exchanged chabazite zeolite structures (e.g., Cu-SSZ-13). The method for supplying ammonia to the SCR catalyst is to inject a mixture of 32.5% urea in water solution into the exhaust stream. In the presence of high temperature exhaust gasses (> 180 - 250 °C)[C], the urea decomposes to form both NH_3 and iso-cyanic acid (HNCO) by thermolysis, with subsequent hydrolysis of the HNCO to form additional NH_3:

$$CO(NH_2)_2 \longrightarrow NH_3 + HNCO$$

$$HNCO + H_2O \longrightarrow NH_3 + CO_2$$

The "standard SCR reaction" of NO (the predominant NO_X species from diesel combustion) over transition-metal zeolite or vanadium SCR catalysts can be represented as:

$$4NH_3 + 4NO + O_2 \longrightarrow 4N_2 + 6H_2O$$

Improved reaction kinetics can be achieved at low exhaust temperatures (<300 °C) by oxidizing a portion of the NO in exhaust using a platinum-group-metal (PGM) coated diesel oxidation catalyst (DOC) to achieve a 1:1 molar ratio of NO:NO_2. The resulting "fast SCR reaction" is:

[B] The temperature at which soot oxidation temperature occurs differs depending on the catalytic coating used on the CDPF.

[C] Note that the urea decomposition temperature is dependent upon spray atomization, exhaust flow turbulence, and exhaust flow rate.

$2NH_3 + NO + NO_2 \longrightarrow 2N_2 + 3H_2O$

An NO_2 SCR reaction with limited NO availability also occurs, but at a significantly slower reaction rate than the standard SCR reaction and is sometimes referred to as the "slow SCR reaction":

$8NH_3 + 6\ NO_2 \longrightarrow 7N_2 + 12H_2O$

The details of SCR reactions for NO_X reduction over transition-metal zeolite SCR can be better represented as a series of interrelated reactions that are part of a more complex redox cycle, such as the one proposed by Rudolf and Jacob.[2]

Urea dosing control takes into account a number of different factors, including:

- The stoichiometry of NO_X reduction by NH_3 (1:1 or 4/3:1 molar ratio)
- Molar ratio of $NO:NO_2$ at the inlet of the SCR catalyst
- The amount of NH_3 stored and released from the zeolites
  - Thermal desorption of stored NH_3 can allow NO_X reduction to occur at exhaust temperatures that are often too low for urea injection and decomposition
- The degree of urea/exhaust mixture preparation of the system design
  - Droplet formation and evaporation
  - Induced turbulent mixing of aqueous urea and exhaust to aid droplet breakup
- The efficiency of urea decomposition to NH_3 (> 95-98% at >250 °C is typical for modern injector/mixer designs)
- The probability forming solid deposits at low exhaust temperatures from partially decomposed urea
  - Urea injection at exhaust temperatures below approximately 180 to 200 °C can result in significantly increased deposit formation depending on mixture preparation and other factors
  - Urea injector fouling can occur from deposit build up on the urea injector tip and other exhaust system surfaces
  - Deposits can temporarily deactivate active catalytic surfaces, requiring higher temperature operation in order to remove the deposits

Copper (Cu) exchanged chabazite zeolites such as Cu-SSZ-13 have demonstrated good hydrothermal stability, good low temperature performance, and represent a large fraction of the transition-metal zeolite SCR catalysts used in heavy-duty diesel applications.[3] Improvements to both the coating processes and the substrates onto which the zeolites are coated have improved the low-temperature and high-temperature NO_X conversion, improved selectivity of NO_X reduction to N_2 (i.e., reduced selectivity to N_2O), and improved the hydrothermal stability. Improvements in SCR catalyst coatings over the past decade have included:[4,5,6,7,8]

- Increased washcoat thickness
- Optimization of Silicon/Aluminum (Al) and Cu/Al ratios
- Increased Cu content and Cu surface area
- Optimization of the relative positioning of Cu^{2+} ions within the zeolite structure
- The introduction of specific co-cations
- Co-exchanging of more than one type of metal ion into the zeolite structure

In the absence of more stringent NO_X standards, these improvements have been realized primarily as reductions in SCR system volume, reductions in system cost, and improvements in durability since the initial introduction of metal-exchanged zeolite SCR in MY2010. Sales-weighted average engine-displacement-specific catalyst volumes for MY2019 MHDDE and HHDDE are shown in Table 1-1.

Table 1-1: Engine-displacement specific catalyst substrate volume for MY2019 MHDDE and HHDDE

Component	MHDDE Specific Volume*	HHDDE Specific Volume*
DOC	0.61	0.74
CDPF	1.39	1.49
SCR	2.11	2.24
ASC	0.38	0.40

Notes:
*Specific Volume = (catalyst total substrate volume) / (engine's piston swept displacement)

NO_X reductions greater than 95% are possible with modern SCR systems over a broad range of operating conditions and at relatively high hours of operation, however SCR functionality is particularly reduced at lower exhaust temperatures due to difficulties with low-temperature urea decomposition and due to slower SCR reaction kinetics (Figure 1-3). In the figure, the initial data point at 140 °C SCR inlet temperature reflects NOX reduction with stored ammonia only (no urea injection). The hours in the legend represent hours of operation over an accelerated aging cycle that included both thermal and chemical effects. The 1,430 hours of aging represented approximately 8,000 hours of equivalent engine operation. Reduced oxidation of NO to NO_2 over the DOC and DPF at low exhaust temperatures (e.g., 200 to 250 °C) reduces the ability to take advantage of the "fast SCR reaction". As previously mentioned, current SCR systems limit urea injection to temperatures above 180 °C to 200 °C to prevent urea injector and catalyst deposits. NO_X reduction reactions at temperatures below approximately 200 °C are thus reliant on use of NH_3 stored within the zeolite structure. During extended operation at low exhaust temperatures, stored NH_3 is eventually depleted and if exhaust temperatures cannot be increased sufficiently to allow initiation of urea injection and effective decomposition to NH_3, then NO_X reduction eventually ceases.

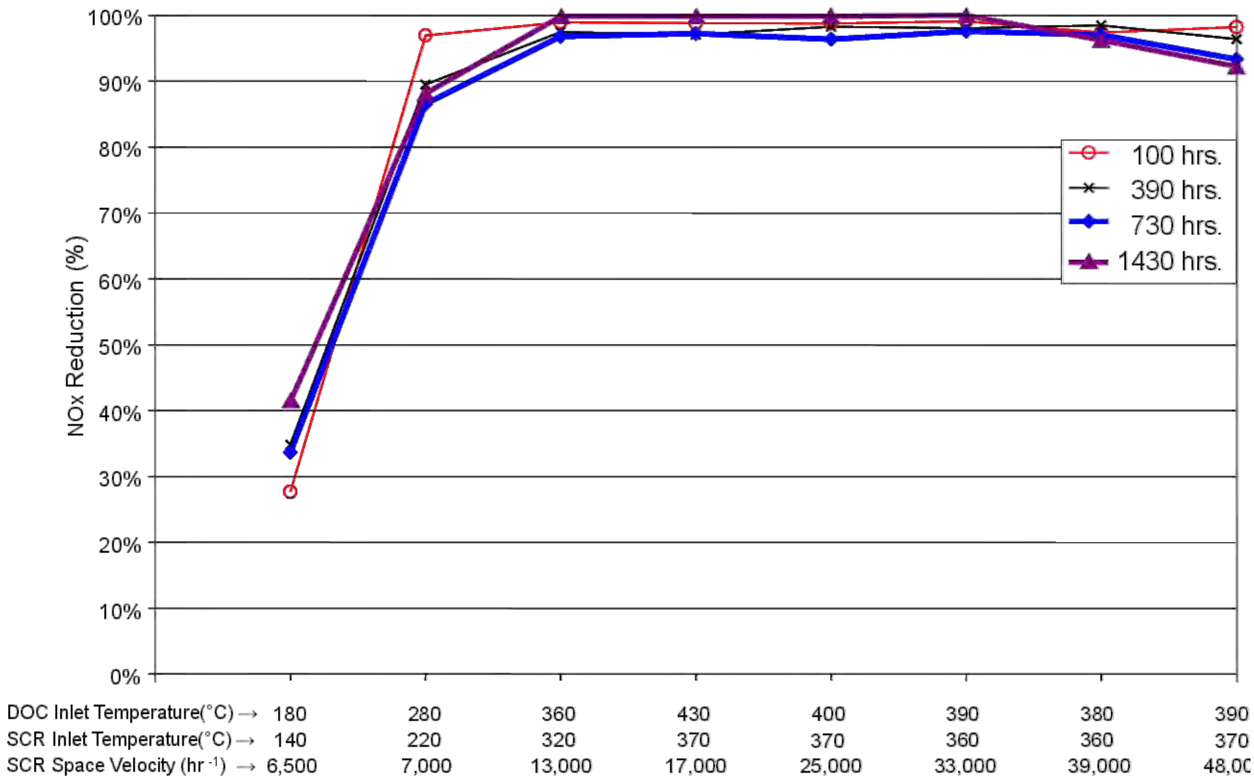

Figure 1-3: NO_X reduction efficiency of an early, developmental Cu-zeolite SCR formulation relative to DOC and SCR inlet temperatures and SCR space velocity (adapted from McDonald et al. 2011[9]).

Compression-ignition engine exhaust temperatures are relatively low following cold starts, during coasting downhill, during sustained idle, or at low vehicle speeds and during light load operation. Technologies that accelerate warm-up from a cold-start and maintain catalyst temperature above 200°C can help achieve further NO_X reduction from SCR systems under those types of operation. Technologies that improve urea decomposition to NH_3 at temperatures below 200°C can also be used to reduce NO_X emissions under cold start, coasting, light load, and low speed conditions.

1.1.2 <u>**Catalyst Durability**</u>

The regulatory full-useful-life for HHDDE emissions compliance with the fully-phased-in 2007 Heavy-duty Standards is 435,000 miles, ten years, or 22,000 hours of operation. Zeolite-based SCR systems have demonstrated high levels of NO_X reduction efficiency at the end of regulatory useful life.[D] The aging mechanisms of diesel exhaust aftertreatment systems are complex and include both chemical and hydrothermal changes. Aging mechanisms on a single component can also cascade into impacts on multiple catalysts and catalytic reactions within the system due to the interrelated nature of catalytic reactions over upstream components on other aftertreatment components further downstream. Some aging impacts are fully reversible (i.e., conditions occur that can fully mitigate the aging impact). Other aging impacts are only partially

[D] Note that this would not necessarily be the case for EAS subjected to misfuelling with a high sulfur distillate diesel fuel, poor maintenance and subsequent severe component failure, or tampering with or removal of key EAS system components.

reversible, irreversible, or can only be reversed with some form of intervention (e.g., changes to engine calibration to alter exhaust temperature and/or composition).

DOCs undergo reversible aging due to adsorption of hydrocarbons, soot and sulfur species; and irreversible aging due to phosphorus (P) poisoning[E] and thermal sintering[F] of platinum group metals (PGM) and other active materials. The catalytic materials on DPFs undergo similar aging impacts, and also continuously accumulate metallic ash, which typically accumulates towards the rear of the DPF channels. As accumulated ash migrates towards the front of the DPF channels, exhaust backpressure increases. Heavy-duty diesel exhaust systems, particularly those of HHDDE, are designed to allow CDPF removal for ash maintenance, often at approximately 250,000- to 350,000-mile intervals. Systems for LHDDE and MHDDE are typically designed with sufficient CDPF capacity to forgo ash maintenance within the current regulatory requirements for full useful life. Ash maintenance involves removal of the CDPF and application of a either a dry cleaning process (e.g., reverse flushing of the CDPF with compressed air) or a wet cleaning process (e.g., reverse flushing of the CDPF with water or with a specific aqueous cleaning solution).

Aging of zeolite SCR is more complex. Hydrothermal aging of Cu-SSZ-13 SCR catalysts impacts both catalyst acidity and NH_3 adsorption, transforms active Cu sites into less active species, and causes Cu migration from exchanged positions within the zeolite structure and subsequent formation of aggregated CuO.[10] The severity of hydrothermal aging increases in the presence of sulfur.[11]

Chemical poisoning of SCR can occur from fuel and lubricant contaminants, or via degradation of upstream components. Sources of chemical poisoning include:

- Lubricant consumption
  - Zinc dialkyldithiophosphate anti-wear, antioxidant, and corrosion inhibiting additives
    - Phosphorus (P)
    - Zinc (Zn)
  - Sulfur (S)
- Fuel
  - Sulfur
  - Trace contaminants from biodiesel (alkali metals, e.g., Na, K)
  - Adsorption of hydrocarbon species into zeolite structure and subsequent blockage of pores through soot formation[12]
- Migration of metals from upstream components
  - PGM from the DOC and CDPF
  - Transition metal (e.g., Fe, Cu) oxides from upstream SCR components or along SCR substrates in series

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[E] The sources of P poisoning are from lubricating oil consumption and P-containing lubricating oil additives, such as zinc dithiophosphate.

[F] Sintering is a solid-solid phase transition that occur at very high temperatures and can lead to the transformation of one crystalline phase into another. Phase transformations typically occur in the bulk washcoat, and they dramatically decrease the surface area of the catalyst.

Hydrothermal and chemical aging impacts on the DOC can also impact SCR NO_X reduction, particularly at low temperatures, via inhibition of NO to NO_2 oxidation necessary for the fast SCR reaction. The potential for future SCR durability improvements fall into the following categories:

- Designing excess capacity into the catalyst (e.g., increased catalyst volume, increased catalyst cell density, increased active material content and surface area)
- Use of a small-volume initial “sacrificial” substrate to adsorb chemical catalyst poisons upstream of the initial DOC or SCR substrate
- Continued improvements to zeolite materials
  - Further optimization of Silicon/Aluminum (Al) and Cu/Al ratios
  - Exchange of beneficial co-cations
  - Co-exchanging of more than one type of transition metal into the zeolite structure
  - Reducing pore size to inhibit HC adsorption and pyrolysis
- Direct hydrocarbon dosing downstream of the light-off SCR during active CDPF regeneration to reduce exposure of the light-off SCR to hydrocarbons and fuel contaminants
- Use of washcoat additives, changes to substrate porosity, and other improvements to increase PGM dispersion, reduce PGM particle size, reduce PGM mobility and reduce agglomeration within the DOC and CDPF washcoatings,
- Improvements to catalyst housings and substrate matting material to minimize vibration and prevent exhaust gas leakage around the substrate
- Reducing SCR and DOC sectional density, either through increased porosity or decreased cell wall thickness, thus lowering substrate mass and improving warm-up characteristics.
- Adjusting engine calibration and emissions control system design to minimize operation that would damage the catalyst (e.g., improved control of CDPF active regeneration, increased passive CDPF regeneration, HC dosing downstream of initial light-off SCR, direct temperature sensor feedback control of active regeneration and chemical deSOx)
- Use of specific engine calibration strategies for chemical deSOx of SCR (e.g., high temperature operation with urea dosing)[13] to remove strongly-bound sulfur compounds from zeolite SCR
- Diagnosis and prevention of upstream engine malfunctions that can potentially damage exhaust aftertreatment components

Increased SCR catalyst capacity, along with incremental improvements to current zeolite coatings would be primary strategies for improving NO_X control over a longer regulatory useful life requirement. SCR capacity can be increased by approximately 40 to 50% with the use of a light-off SCR substrate combined with a downstream substrate with a moderate volume increase and with moderately increased catalytic activity from continued incremental improvements to chabazite and other zeolite coatings used for SCR. Total SCR volume would thus increase by approximately 50% to 80% relative to today’s systems. SCR capacity can also be increased in the downstream SCR system using thin-wall (4 to 4.5 mil), high cell density (600 cells-per-square-inch) substrates.

Chemical aging of the DOC, CDPF, and SCR can be reduced by the presence of an upstream light-off SCR or use of a small "sacrificial" substrate to adsorb chemical poisons. Transport and adsorption of sulfur (S), P, calcium (Ca), zinc (Zn), sodium (Na), and potassium (K) compounds and other catalyst poisons are more severe for the initial catalyst within an emissions control system and tend to reduce in severity for catalysts positioned further downstream. Chemical deSOx strategies can be used to remove strongly-bound sulfur from zeolite SCR[13]. This involves creating a strongly reducing environment via dosing of urea in excess of the typical 1:1 NH_3 to NO_X ratio at temperatures of approximately 500 °C to 550 °C. Further evolutionary improvements to the DOC washcoating materials to increase PGM dispersion and reduce PGM mobility and agglomeration are also anticipated for meeting increased useful life requirements.

The primary strategy for maintaining CDPF function to a longer useful life is via design of integrated systems that facilitate easier removal of the CDPF for ash cleaning at regular maintenance intervals. Accommodation of CDPF removal for ash maintenance is already incorporated into many existing diesel exhaust system designs.[G] Incremental improvements to catalyst housings and substrate matting material are also expected to be necessary for all catalyst substrates within the system. Integration into a box-muffler type system is currently being used by a number of manufacturers and this approach is expected to continue for all catalyst components (except possibly for an initial close-coupled/light-off SCR) in order to improve passive thermal management and improve access to the DPF for ash maintenance.

1.1.3 <u>Improving SCR NO_X Reduction at Low Exhaust Temperatures</u>

The improvement of SCR NO_X reduction under low-speed (<1200 rpm) light-load (< 5-bar BMEP) conditions or immediately following cold starts will require improvements to both active and passive thermal management of the EAS.

1.1.3.1 Active Thermal Management

Active thermal management involves using engine hardware and associated control systems to maintain and/or increase exhaust temperatures. This can be accomplished through a variety of means, including engine throttling, heated aftertreatment systems, and exhaust flow bypass systems. Later combustion phasing can also be used for active thermal management.

Diesel engines operate at very low fuel-air ratios (i.e., with considerable excess air), and particularly so at low load (<5-bar BMEP) conditions. This causes relatively cool exhaust to flow through the exhaust system at low loads, which cools the catalyst substrates. This is particularly the case at idle. It is also significant at moderate-to-high engine speeds with little or no engine load, such as when a vehicle is coasting down a hill. Air flow through the engine can be reduced by induction and/or exhaust throttling. All modern heavy-duty diesel engines are equipped with an electronic throttle control (ETC) within the induction system and most are equipped with a variable-geometry-turbine (VGT) turbocharger, and these systems can be used to throttle the induction and exhaust system, respectively, at light-load conditions. However, throttling reduces volumetric efficiency[H], and thus has a trade-off relative to CO_2 emissions and fuel consumption.

[G] Video by Eberspaecher demonstrating DPF removal for ash cleaning maintenance: https://youtu.be/lf_vysKbfaA
[H] Relative efficiency of the air-exchange process in an internal combustion engine

Heat can be added to the exhaust and the EAS by burning fuel in the exhaust system or by using electrical heating, both of which can increase the SCR efficiency. Burner systems use an additional diesel fuel injector in the exhaust to combust fuel and create additional heat energy in the exhaust flow. Electrically heated catalysts (EHC) use electric current applied to a metal foil monolithic structure in the exhaust to add heat to the exhaust flow. At light-load conditions with relatively high flow/low temperature exhaust, considerable fuel energy or electric energy is needed for these systems. This would likely cause a considerable increase in CO_2 emissions and fuel consumption with conventional designs. Heated and higher-pressure urea dosing systems improve the decomposition of urea at low exhaust temperatures and thus allow urea injection to occur at lower exhaust temperature (i.e., at approximately 135 °C to 140 °C) and with considerably less energy-input when compared to burner systems or EHC.

Exhaust flow bypass systems can be used to manage the cooling of exhaust during cold start and low load operating conditions. For example, significant heat loss occurs as the exhaust gases flow through the turbocharger turbine. Turbine bypass valves allow exhaust gas to bypass the turbine and avoid this heat loss at low loads when turbocharger boost requirements are low. In addition, an EGR flow bypass valve would allow exhaust gases to bypass the EGR cooler when EGR cooling is not required, such as immediately following a cold start or under cold ambient conditions. EGR cooler bypass is currently used in light-duty diesel, Light HDE, and Medium HDE applications.[14,15]

Variable valve actuation (VVA) systems can also be used for active thermal management. VVA includes a family of valvetrain designs that alter the timing and/or lift of the intake and exhaust valves. Use of VVA can reduce pumping losses, increase specific power, and control the level of residual gases in the cylinder.

VVA has been adopted in light-duty vehicles to increase an engine’s efficiency and specific power. It has also been used as a thermal management technology to open exhaust valves early to increase heat rejection to the exhaust and heat up exhaust catalysts more quickly. This VVA strategy, called early exhaust valve opening (EEVO), has been applied to the Detroit DD816 to aid in CDPF regeneration, but a challenge with this strategy for maintaining aftertreatment temperature is that it reduces cycle thermal efficiency, and thus can contribute to increased CO2 emissions.

Cylinder deactivation (CDA), late intake valve closing (LIVC), and early intake valve closing (EIVC) are three VVA strategies that can also be used to reduce airflow through the exhaust system at light-load conditions and can reduce the CO_2 emissions and fuel consumption trade-off compared to use of the ETC and/or VGT for throttling.[17,18,19,20]

Since we are particularly concerned with catalyst performance at low loads, EPA evaluated two valvetrain-targeted thermal management strategies that reduce air aspiration of engines at light-load conditions (i.e., less than 3-4 bar BMEP): CDA and LIVC. Both strategies force engines to operate at a higher fuel-air ratio in the active cylinders for a given load demand, which increases exhaust temperatures, with the benefit of little or no fuel consumption increase and with potential for fuel consumption decreases under some operating conditions. The key difference between these two strategies is that CDA completely removes airflow from one or more deactivated cylinders with the potential for exhaust temperature increases of up to 80 °C at light loads, while LIVC reduces airflow from all cylinders with up to 40 °C hotter exhaust temperatures.[18,19,20]

One of the challenges of CDA is that it requires proper integration with the rest of the vehicle's driveline. This can be difficult in the vocational vehicle segment where an engine is often sold by the engine manufacturer (to a chassis manufacturer or body builder) without knowing the type of transmission or axle used in the vehicle or the precise duty cycle of the vehicle. The use of CDA requires fine tuning of the engine calibration as the engine moves into and out of deactivation to achieve acceptable noise, vibration, and harshness (NVH). Mitigation strategies include changes to driveline dampening, motor mount and/or chassis dampening, and the use of dynamic CDA with individual cylinder deactivation control. LIVC may provide emission reductions similar to fixed CDA, with the added benefits of no significant NVH concerns and some efficiency improvements under higher load conditions.

1.1.3.2 Passive Thermal Management

Passive thermal management involves changes or modifications to component designs to increase and maintain the exhaust gas temperatures without the use of active thermal management. It is done primarily through insulation and/or reducing the mass of EAS and other exhaust system components so that less exhaust energy input is required to reach catalyst light-off temperatures and/or the exhaust temperatures at which urea dosing can commence.[21] Passive thermal management strategies generally have little to no impact on CO_2 emissions or can improve CO_2 emissions if used to replace an active thermal management strategy. The use of passive thermal management strategies for improving catalyst light-off in light-duty gasoline applications has led to significant reductions in cold-start exhaust emissions.[22] Passive thermal management design elements can be equally applied to EAS systems used in heavy-duty applications.

More specifically, using a smaller sized, initial SCR catalyst within the EAS with a high-porosity, lower density substrate reduces its mass and reduces catalyst warmup time. Moving the SCR catalyst nearer to the turbocharger outlet effectively reduces the exhaust system mass and surface area prior to the SCR inlet, minimizing heat loss and reducing the amount of energy needed to warm components up to normal operating temperatures. Reducing the mass of the exhaust system and insulating between the turbocharger outlet and the inlet of the SCR system using an air-gap or other insulation can also reduce the amount of thermal energy lost through the exhaust system walls. Close coupling of catalysts is near ubiquitous in modern light-duty EAS. The use of air-gap construction is also common in light-duty applications.

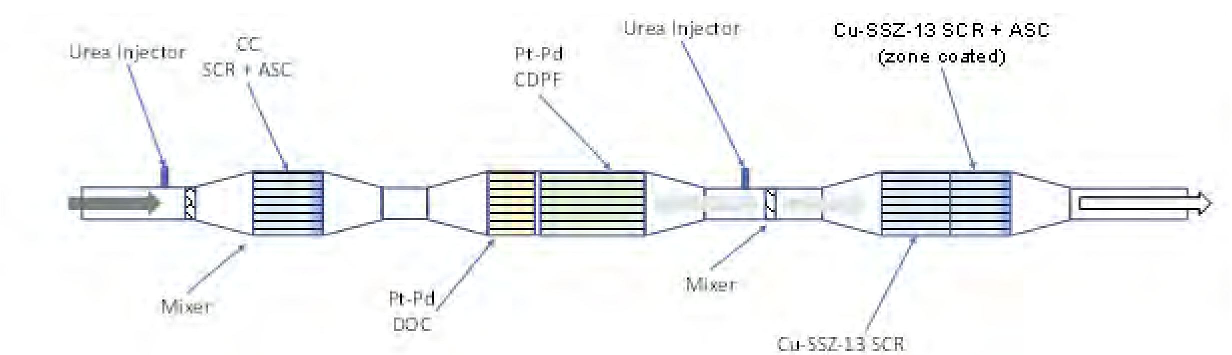
Dual-walled manifolds and exhaust pipes utilizing a thin inner wall and an air gap separating the inner and outer wall can be used to simultaneously insulate the exhaust system and reduce the thermal mass, minimizing heat lost to the walls and decreasing the time necessary to reach operational temperatures after a cold start. Mechanical insulation applied to the exterior of exhaust components, including exhaust catalysts, is readily available and can minimize heat loss to the environment and help retain heat within the catalyst as operation transitions to lighter loads and lower exhaust temperatures. Integrating the DOC, DPF, and SCR substrates into a single exhaust assembly can also assist with retaining heat energy.

EPA evaluated several passive thermal management strategies in the diesel technology feasibility demonstration program. See RIA Chapter 3.1 for detailed discussion of our diesel technology demonstration programs).

1.1.3.3 Advanced SCR System Development

A recent development in SCR system architecture is the development of light-off or dual SCR systems, which is a variation of passive thermal management.[18, 23, 24] This system maintains a layout similar to the conventional SCR configuration discussed earlier, but integrates an additional small-volume SCR catalyst, which is in some cases also close-coupled to the turbocharger's exhaust turbine outlet (Figure 1-4). This small SCR catalyst may be configured with or without an upstream DOC, and with or without a small sacrificial substrate to adsorb chemical poisons upstream of an initial SCR substrate. A recent example of this system's architecture was demonstrated as part of "Stage 3" of the California Air Resources Board (CARB) – Heavy-duty Low-NO_X Test Program.[25] The CARB Stage 3 research program is summarized within Chapter 3.1. EPA evaluated dual-SCR catalyst system technology similar to the CARB "Stage 3" system as part of a diesel technology feasibility demonstration program (see Chapter 3.1 for more detail).

The benefits of this design result from its ability to warm up the initial light-off SCR substrate faster as a result of it being relatively low mass and being the first catalyst downstream of the turbocharger with the EAS. Such light-off SCR catalysts can also be designed to have smaller substrates with lower bulk density. The reduced mass reduces thermal inertia and allows faster warmup. The design also positions the urea injection and mixing as the very first components in the system, thus allowing faster heat up of the urea injector and urea mixer when implementing active thermal management measures. These designs also require less input of heat energy into the exhaust to maintain exhaust temperatures during light-load operation. Urea injection to the close-coupled light-off SCR can also be reduced or terminated once the second, downstream SCR reaches operational temperature, thus allowing additional NO_X to reach the DOC and CDPF to promote passive regeneration (soot oxidation) on the CDPF, reducing fuel consumption and CO_2 emissions. Very close-coupling of the light-off SCR to the exhaust turbine is possible when using heated urea dosing system since such systems enable a relatively short mixing length between the urea dosing system and the inlet of the light-off SCR (see Chapter 3.1 for more detail).



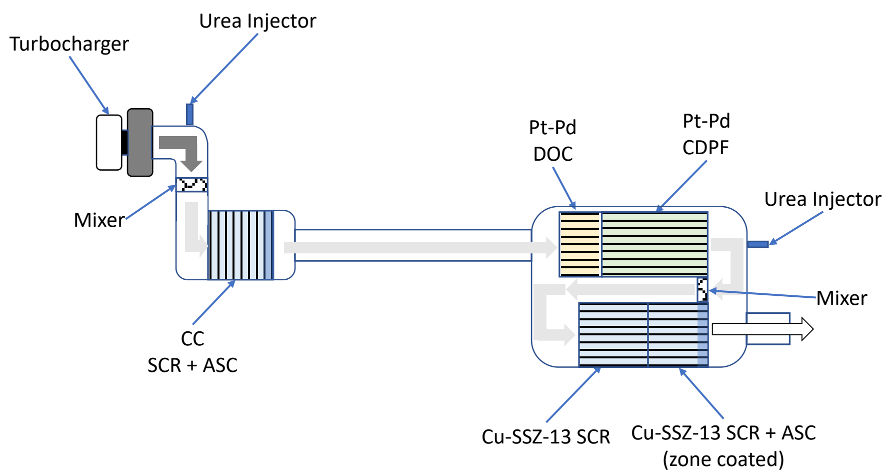


Figure 1-4: Potential layout of a 2027+ dual-SCR system in an in-line configuration (top) and comparable components integrated to improve passive thermal management (bottom).

One potential concern about this technology is the durability challenge associated with placing an SCR catalyst upstream of the CDPF. To address this concern, two light-off SCR system designs were hydrothermally and chemically aged to an equivalent of 850,000 miles as part of the EPA Heavy-duty Diesel Low NO_X Demonstration Program. Please refer to Chapter 3.1 for additional information regarding this test program.

1.1.4 <u>Crankcases</u>

During combustion, gases can leak past the piston rings sealing the cylinder and into the crankcase. These gases are called blowby gases and generally include unburned fuel and other combustion products. Blowby gases that escape from the crankcase are considered crankcase emissions (see 40 CFR 86.402-78). Current regulations restrict the discharge of crankcase emissions directly into the ambient air, and blowby gases from gasoline engine crankcases have been controlled for many years by sealing the crankcase and routing the gases into the intake air through a positive crankcase ventilation (PCV) valve. However, in the past there have been concerns about applying a similar technology for diesel engines. For example, high PM emissions venting into the intake system could foul turbocharger compressors. As a result of this concern, diesel-fueled and other compression-ignition engines equipped with turbochargers (or other equipment) were not required to have sealed crankcases (see 40 CFR 86.007-11(c)). For these engines, manufacturers are allowed to vent the crankcase emissions to ambient air as long as they are measured and added to the exhaust emissions during all emission testing to ensure compliance against the emission standards.

Because all new highway heavy-duty diesel engines on the market today are equipped with turbochargers, they are not required to have closed crankcases under the current regulations. Manufacturer compliance data show approximately one-third of current highway heavy-duty diesel engines have closed crankcases, indicating that some heavy-duty engine manufacturers have developed systems for controlling crankcase emissions that do not negatively impact the turbocharger. In the final rule associated with this RIA, EPA is finalizing a requirement for manufacturers to use one of two options for controlling crankcase emissions. One option is closing the crankcase, as proposed. These emissions could be routed upstream of the aftertreatment system or back into the intake system. The second option is an updated version of the current requirements for an open crankcase that includes accounting for total emissions during certification and off-cycle field testing through useful life including full accounting of crankcase emission deterioration (See Preamble Section III.B).

1.1.4.1 Emissions from Open Crankcases

EPA conducted emissions testing of open crankcase systems on two low mileage, modern heavy-duty diesel trucks.[26] The testing was conducted at EPA's National Vehicle and Fuel Emissions Laboratory. The two vehicles were tested on a heavy-duty chassis dynamometer where the crankcase flow and emissions were measured separately from the tailpipe exhaust emissions. The vehicles were tested over a variety of operating conditions. The cycles included the ARB Transient cycle with a cold start, repeat ARB Transient cycles, a 10-minute idle cycle, and a highway cycle at 55 mph and 65 mph.

The crankcase emission rates were calculated for THC, NMHC, CH_4, NO_X, CO_2, and CO using the densities found in 40 CFR 86.144-94. The average crankcase and tailpipe emission rates for each of the two trucks (NVFEL 1 and NVFEL 2) by test phase are show in Figure 1-5. The error bars represent the standard error of the mean. As shown, the crankcase THC and CO emissions are a notable fraction of the tailpipe exhaust emissions. Table 1-2 includes the average crankcase emission rates across the cycles for each truck.

Table 1-2: Average Crankcase Emission Rates (gram/hour)

	THC	**CH4**	**NOX**	**CO**
Truck 1	0.305	0.001	1.19	0.212
Truck 2	0.067	0.026	1.09	0.832

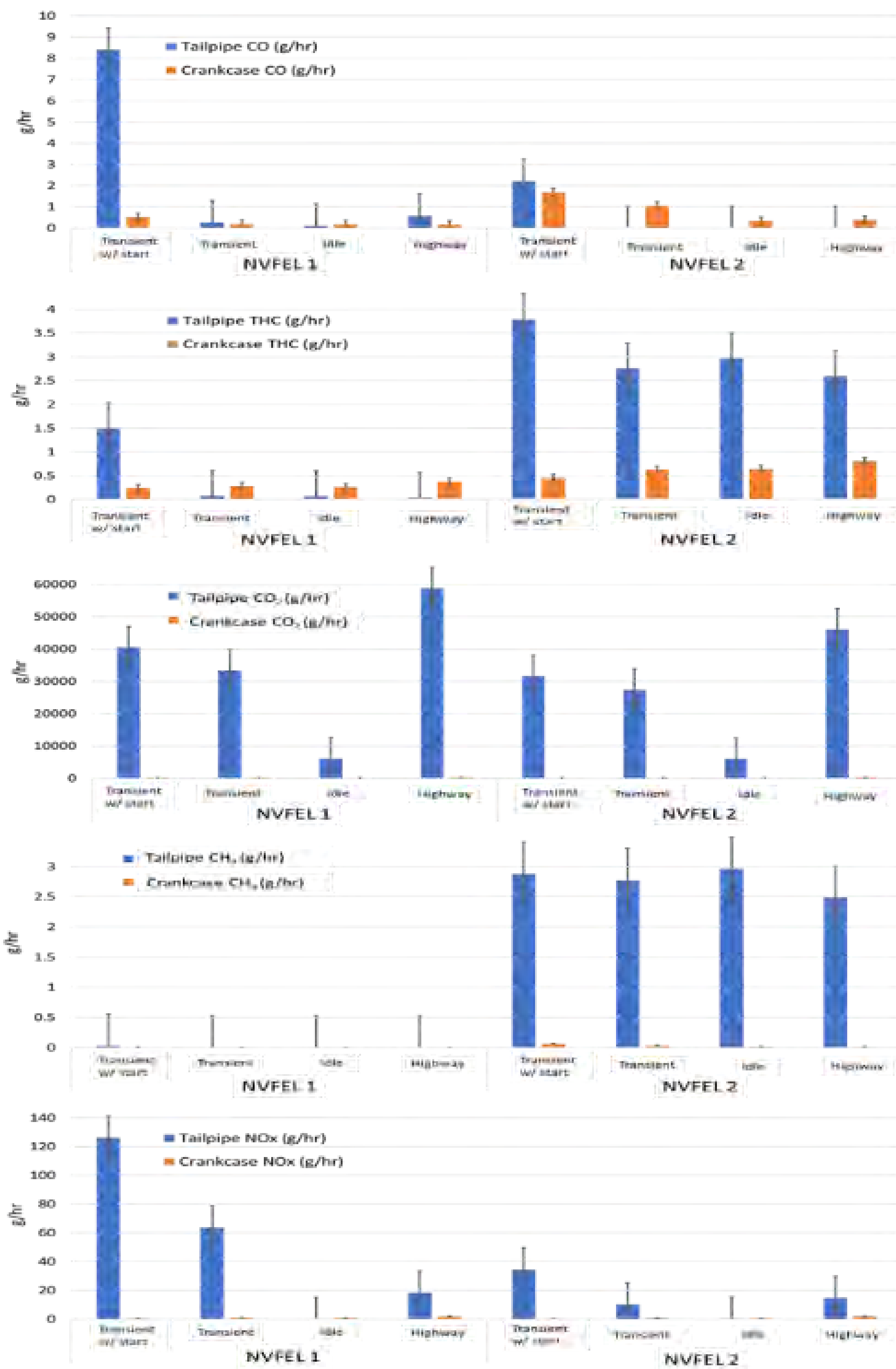


Figure 1-5: Tailpipe Exhaust and Crankcase Emission Rates from Two Heavy-Duty Diesel Trucks

We were unable to measure PM emissions from the crankcase as part of the EPA test program. Therefore, in our MOVES model we will continue to use the PM emission rates measured in the Advanced Collaborative Emissions Study (ACES) Phase 1 test program.[27] The average PM emission rate of the four 2007 MY heavy-duty diesel engines was 32.1 mg/hour.

1.1.4.2 Description of Closed Crankcase Technologies

If manufacturers choose to close the crankcase, crankcase emissions can be controlled through the use of closed crankcase filtration systems or by routing unfiltered blow-by gases directly into the exhaust system upstream of the emission control equipment. Closed crankcase filtration systems work by separating oil and particulate matter from the blow-by gases through single or dual stage filtration approaches, routing the blowby gases into the engine's intake manifold and returning the filtered oil to the oil sump. These systems were required for new heavy-duty diesel vehicles in Europe starting in 2000. Oil separation efficiencies in excess of 90 percent have been demonstrated with production ready prototypes of two stage filtration systems. By eliminating 90 percent of the oil that would normally be vented to the atmosphere, the system works to reduce oil consumption and to eliminate concerns over fouling of the intake system when the gases are routed through the turbocharger.

An alternative approach would be to route the blow-by gases into the exhaust system upstream of the catalyzed diesel particulate filter which would be expected to effectively trap and oxidize the engine oil and diesel PM. This approach may require the use of low sulfur engine oil to ensure that oil carried in the blow-by gases does not compromise the performance of the sulfur-sensitive emission control equipment.

Our feasibility analysis is based on the use of closed crankcase system that includes technologies designed to filter crankcase gases sending the clean gas to the engine intake for combustion and returning the oil filtered from the gases to the engine crankcase. These systems are proven in use.

1.1.5 Opposed-Piston Diesel Engines

While not part of EPA's planned technology demonstration program for this rulemaking, the agency is tracking ongoing work to develop opposed-piston diesel engine technology for heavy-duty on-highway vehicle applications.[28,29] One example of work with this technology is a project to develop and demonstrate a 10.6 liter, 450 hp opposed-piston diesel engine and related aftertreatment technologies for Class 8 line-haul tractors operating at certified emission performance levels of 0.02 g NO_X/bhp-h (90 percent below US 2010) and 432 g CO_2/bhp-h (2027 HD Phase 2 engine standard).[30] In addition to the emissions demonstration work, a high level cost study has been conducted by FEV that indicates the direct and indirect cost of an opposed-piston engine are less than that of a conventional HD diesel engine.31 This project is supported by a variety of public and private sector partners including: Achates Power, Aramco Services Company, BASF, CALSTART, CARB, Corning, Delphi, Eaton, Faurecia, Federal Mogul, PACCAR/Peterbilt, Sacramento Metropolitan Air Quality Management District, San Joaquin Valley Air Pollution Control District, South Coast Air Quality Management District, Southwest Research Institute, Tyson Foods, and Walmart.[32]

Opposed-piston engine technology has not yet been proven feasible in Class 8 on-highway applications, but if it becomes feasible, then the technology could provide another pathway to ultra-low NO_X, high efficiency engine technology for heavy-duty vehicle fleets. As such, it may be reasonable to anticipate commercialization of heavy-duty opposed-piston diesel engine technology by model year 2027.

1.2 Spark-Ignition Engine Technologies

The following sections describe the spark-ignition (SI) engine technologies we considered to reduce criteria pollutant emissions (NMHC, CO, NO_X, and PM) as part of this rulemaking. Many of the technologies are described with respect to gasoline fuel, but they are expected to be broadly applicable to all fuels used in spark-ignition engines. Our spark-ignition engine feasibility demonstration for this final rule is based on some of the technologies presented in this section.

Chapter 2.3 of this RIA describes the test procedures for spark-ignition engine certification. Chapter 3.2 describes the spark-ignition engine feasibility demonstration program, including a description of the specific technology packages we evaluated, the effectiveness of those technologies over our current and new test procedures, and our projected direct manufacturing cost of those technologies.

1.2.1 Technology Description for NMHC, CO, and NO_X Control

A range of technology options exist to reduce NMHC, CO, and NO_X emissions from both heavy-duty highway gasoline fueled spark ignition and diesel engines to levels below the current EPA 2007/2010 standards. Available options include modifications to the engine calibration, engine design, exhaust system, and aftertreatment system design. The different available options each contain specific benefits and limitations. This section describes the technical challenges to reducing emissions from current levels, describes available technologies for reducing emissions, estimates the potential emissions reduction of the different technologies, describes if there are other ancillary benefits to engine and vehicle performance with the technology, and reviews the limits of each technology. Except where noted, these technologies are applicable to all spark-ignition engines covered by this rule. Unique compression-ignition technologies are addressed in Section 1.1.

1.2.1.1 Summary of the Technology Challenge for NMHC, CO, and NO_X Control

Historically, heavy-duty spark-ignition engines running the EPA Federal Test Procedure (FTP) tests have shown that the majority of NMHC, CO, and NO_X emissions occur during the cold start phase. Real world emissions during warmed-up and hot operation, specifically during high-load operation that may not be fully exercised by the FTP tests, can significantly contribute to emissions. Additionally, as described in Chapter 3.2, in-vehicle testing has indicated that sustained low load conditions such as prolonged idling can result in emission increases due to reduced aftertreatment temperatures (i.e., cool-off). Prolonged idling is also a real world condition that is not thoroughly addressed by the FTP. For this rulemaking, we evaluated strategies that target prolonged idle and FTP and high-load NMHC, CO and NO_X emission control performance. Specifically, significant quantities of NMHC and CO emissions can be produced if enrichment events occur regularly during high-load operation. Control of NO_X emissions during high-load operation requires designs that provide sufficient catalyst volume to handle the higher exhaust gas flow rates and also precise control of closed loop fuel biasing for the catalyst to maintain peak NO_X efficiency.

In order to achieve significantly lower NMHC, CO and NO_X emissions over the FTP, manufacturers can change the design of their exhaust and catalyst systems, as well as adopt calibration strategies to reduce catalyst light-off times and reduce warmed-up and hot running emissions. Design changes to reduce catalyst light-off time (e.g., closer catalyst placement) can also result in higher catalyst temperatures during high-load operation. To achieve lower NMHC and NO_X levels, manufacturers will need to develop and implement technologies and calibration strategies to manage catalyst temperatures during high-load operation while minimizing fuel enrichment.

For the catalyst to effectively reduce NMHC, CO and NO_X emissions it must reach a light-off temperature of approximately 350 °C. Emissions during the catalyst warm up period can be reduced by reducing the emissions produced by the engine during the catalyst warm up phase. Emissions can also be reduced by shortening the time period required for the catalyst to reach the light-off temperature and maintaining sufficient catalyst temperature during low load and idle operation. Reducing warmed-up NO_X emissions requires improving the efficiency of the catalyst system using improved catalyst loading and washcoat technologies in addition to more precise calibration and software controls. NO_X emissions control system effectiveness has generally improved from a reduction in the sulfur content of the fuel in recent years.[I]

It is anticipated that engine manufacturers will focus on four areas to reduce emissions:

- Minimizing the emissions produced by the engine before the catalyst reaches the light-off temperature
- Reducing the time required for the catalyst to reach the light-off temperature and staying above the light-off temperature throughout all operation
- Improving the NO_X efficiency of the catalyst during warmed-up operation at medium and high loads
- Minimizing or eliminating enrichment in high-load operation.

We describe strategies to address these four areas in the following sections.

1.2.1.2 Reducing Engine-Out Emissions

When the three-way catalyst (TWC) of an SI engine is "cold" (i.e., the engine temperature matches ambient and is between 20 and 30 °C (68 and 86 °F)) and the engine is operating under low loads, the cold engine produces lower concentrations of NO_X than NMHC. As the engine warms up and as the load increases, the concentration of NO_X produced by the engine increases and the concentration of NMHC decreases. We have observed this NMHC-NO_X tradeoff during the cold start portion of the FTP duty cycle as the engine transitions from the first minutes of operation when the engine is either at idle or low speed and load to higher load with a warmed engine.

[I] Sulfur content in commercially available fuel was reduced starting in 2017 as part of the Tier 3 Motor Vehicle Emission and Fuel Standards rule (79 FR 23414); this test program was conducted using gasoline with Tier 3 sulfur content.

The design of the air induction system, combustion chamber, spark plug, and fuel injection system determine the quantity of fuel required for stable combustion to occur in the cold engine. Optimizing the performance of these components can provide reductions in the amount of fuel required to produce stable combustion during these cold operating conditions. Reductions in the amount of fuel required leads to reductions in cold start NMHC emissions.

The design considerations to minimize cold start emissions are also dependent on the fuel injection method. Port fuel injected (PFI) engines have different design constraints than gasoline direct injection (GDI) spark ignition engines. For both PFI and GDI engines, however, attention to the details affecting the in-cylinder air/fuel mixture can reduce cold start NMHC emissions.

For example, it has been shown that cold start NMHC emissions in PFI engines can be reduced by reducing the size of the fuel spray droplets and optimizing the spray targeting. Fuel impinging on cold engine surfaces in the cylinder does not readily vaporize and does not combust.[33] Improving injector targeting to reduce the amount of fuel reaching the cylinder walls reduces the amount of fuel needed to create a combustible air fuel mixture. Reducing the size of the spray droplets improves the vaporization of the fuel and the creation of a combustible mixture. Droplet size can be reduced by modifying the injector orifice plate and by increasing the fuel pressure. Reducing droplet size to improve fuel vaporization during cold start has been shown to reduce cold transient emissions by up to 40 percent during the cold start phase of the light-duty FTP emission test.

The mixture formation process in a GDI engine is different than a PFI engine. In a PFI engine the fuel can be injected prior to or during the intake valve opening to prepare the fuel in an optimal manner for emission controls. The fuel generally has time to evaporate during the intake stroke as the fuel and air are drawn into the cylinder and is mixed with the incoming air. In addition, as the engine combustion heat from the previous firing events warms the intake valve and other surfaces in the area, the fuel can be injected into the intake runner and engine heat can assist in evaporating the fuel prior to the intake valve opening. The GDI engine injects fuel at higher fuel pressures than PFI engines directly into the combustion chamber. In a GDI engine, the fuel droplets need to evaporate and mix with the air in the cylinder in order to form a flammable mixture. Injecting directly into the cylinder reduces the time available for the fuel to evaporate and mix with the intake air in a GDI engine compared to a PFI engine. An advantage of the GDI design is that the fuel spray does not impinge on the walls of the intake manifold or other surfaces in the cylinder.

GDI systems stagger the injection timing event. At least one study has indicated that significant reductions in hydrocarbon emissions can be achieved by splitting the injections during the cold start of a GDI engine. An initial injection occurs during the intake stroke and a second injection is timed to occur during the compression stroke. This injection method reduced unburned hydrocarbon emissions 30 percent compared to a compression stroke only injection method.[34]

These are two examples of specific engine design characteristics, fuel injector design and fuel system pressure on PFI engines and injection timing on GDI engines, which can be used to reduce cold start NMHC emissions significantly during the engine warm up prior to the catalyst reaching the light-off temperature.

Optimizing the fuel injection system design and calibration is anticipated to be used in all vehicle classes, including heavy-duty vehicles. It is anticipated that these described improvements, along with improvements to other engine design characteristics, will be used to reduce cold start emissions for passenger cars, LDTs, MDPVs, and HDTs in coming model years, which will pave the way for them to be applied to heavy-duty engines.

Because the engine is relatively cold and the operating loads are low during the first 50 seconds of operation, the engines typically do not produce significant quantities of NO_X emissions during this time. In addition, manufacturers tend to retard the combustion timing during the catalyst warm up phase. Retarding combustion timing has been shown also to reduce the concentration of NMHC in the exhaust. This calibration method further reduces peak combustion temperatures while increasing the exhaust gas temperature compared to optimized combustion timing. The increased exhaust gas temperature leads to improved heating of the catalyst and reduced catalyst light-off times. Retarding combustion and other technologies for reducing catalyst light-off time are discussed in the following section.

1.2.1.3 Reducing Catalyst Light-Off Time

The effectiveness of current engine emission control systems depends in large part on the time it takes for the catalyst to light-off, which is typically defined as the catalyst reaching a temperature of 350°C. In order to reduce catalyst light-off time, it is expected manufacturers will use technologies that will improve heat transfer to the catalyst during the cold start phase and improve catalyst efficiency at lower temperatures. Technologies to reduce catalyst light-off time include calibration changes, thermal management, close-coupled catalysts, catalyst PGM loading, and secondary air injection. The technologies are described in greater detail below.

1.2.1.3.1 *Calibration Changes*

Engine calibration changes may be employed to increase the temperature and mass flow of the exhaust with the goal of reducing the amount of time required for the catalyst to reach the critical light-off temperature. By reducing the time required for the catalyst to light-off, these changes can effectively reduce NMHC, CO and NO_X emissions. Since the catalyst system in an SI engine is the predominant method to control emissions and is responsible for over a 95% reduction from the engine out emissions, any acceleration in the warm-up of the catalyst system translates into immediate emission reductions at the tailpipe.

Retarding combustion in a cold engine by retarding the spark advance is a well-known method for reducing the concentration of NMHC emissions in the exhaust and increasing the exhaust gas temperature.[35,36] The reduction in NMHC concentrations is due to a large fraction of the unburned fuel within the cylinder combusting before the flame is extinguished at the cylinder wall. Reductions of total hydrocarbon mass of up to 40 percent have been reported from these studies evaluating the effect of spark retard on exhaust emissions.

In addition to reducing the NMHC exhaust concentration, retarding the spark advance reduces the torque produced by the engine. In order to produce the same torque and maintain the engine speed and load at the desired level when retarding the spark advance, the air flow into the engine is increased causing the manifold pressure to increase which can also improve combustion stability. Retarding the combustion process also results in an increase in the exhaust gas temperature. The retarded ignition timing during the cold start phase in addition to reducing the

NMHC emissions increases the exhaust mass flow and exhaust temperature. These changes lead to a reduction in the time required to heat the catalyst.

The torque produced by the engine will begin to vary as the spark retard amount reaches engine combustion limits. As the torque variations increase, the combustion process is deteriorating, and the engine performance begins to degrade due to the partial burning. It is the level of this variability which defines the absolute maximum reduction in spark advance that can be utilized to reduce NMHC emissions and reduce the catalyst light-off time.

Retarding combustion during cold start can be applied to spark-ignition engines in all vehicle classes. The exhaust temperatures and NMHC emission reductions will vary based on engine design. This calibration methodology is anticipated to be used to improve catalyst warm-up times and reduce cold start NMHC emissions for all vehicle classes.

With the penetration of variable valve timing technology increasing in gasoline-fueled engines, additional work is being performed to characterize the impact of valve timing on cold start emissions. Calibration changes to the valve timing during the cold start phase can lead to additional reductions in cold start NMHC emissions.[37]

1.2.1.3.2 <u>*Exhaust System Thermal Management*</u>

This category of technologies includes all design attributes meant to conduct combustion heat into the catalyst with minimal cooling. This includes insulating the exhaust piping between the engine and the catalyst, reducing the wetted area of the exhaust path, reducing the thermal mass of the exhaust system, and/or using close-coupled catalysts (i.e., the catalysts are packaged as close to the engine cylinder head as possible to mitigate the cooling effects of longer exhaust piping). By reducing the time required to achieve catalyst light-off, thermal management technologies reduce NMHC, CO and NO_X emissions.

Moving the catalyst closer to the cylinder head is a means that manufacturers have been using to reduce both thermal losses and the catalyst light-off time. Many vehicles today use close-coupled catalysts, a catalyst which is physically located as close as possible to the cylinder head. Moving the catalyst from an underbody location closer to the cylinder head reduces the light-off time significantly.

Another means for reducing heat losses is to replace cast exhaust manifolds with thin-wall stamped manifolds. Reducing the mass of the exhaust system reduces the heat losses of the system. In addition, an insulating air gap can be added to the exhaust system which further reduces the heat losses from the exhaust system. Insulating air gap manifolds are also known as dual-wall manifolds.

With thin- and dual-wall exhaust manifolds, close-coupled catalyst housings can be welded to the manifold. This reduces the needed for manifold to catalyst flanges which further reduces the thermal inertia of the exhaust system. Close coupling of the catalyst and reducing the thermal mass of the exhaust system significantly reduces the light-off time of the catalyst compared to an underbody catalyst with flanges and pipes connected to a cast exhaust manifold.

Using close-coupled catalysts reduces the heat losses between the cylinder head and catalyst. While reducing the time required to light-off the catalyst the close-coupled catalyst can be subject to higher temperatures than underbody catalysts during high-load operating conditions.

To ensure the catalyst does not degrade, manufacturers currently use fuel enrichment to maintain the exhaust temperatures below the levels which would damage the catalyst. It is anticipated that manufacturers will ensure that fuel enrichment is minimized on the FTP. Calibration measures, other than fuel enrichment, may therefore be needed to ensure the catalyst temperature does not exceed the maximum limits.

Another technology beginning to be used for both reducing heat loss in the exhaust and limiting exhaust gas temperatures under high-load conditions is integrating the exhaust manifold into the cylinder head. Honda utilized this technology on the Insight's 1.0 L VTEC-E engine. The advantage of this technology is that it minimizes exhaust system heat loss during warm-up. In addition, with the exhaust manifold integrated in the cylinder head, the cooling system can be used to reduce the exhaust temperatures during high-load operation. It is anticipated that manufacturers will further develop this technology as a means to both quickly light-off the catalyst and reduce high-load exhaust temperatures.

We expect thermal management to be an effective strategy for manufacturers to lower NMHC, CO and NO_X emission levels. Our feasibility demonstration described in Chapter 3.2 evaluates catalysts located closer to the engine as a method of thermal management. We expect that manufacturers will further optimize the thermal inertia of the exhaust system to minimize the time needed for the catalyst to achieve the light-off temperature, while ensuring the high-load performance does not cause thermal degradation of the catalyst system. It is expected that methods and technologies will be developed to reduce the need to use fuel enrichment to reduce high-load exhaust temperatures.

Optimizing the catalyst location and reducing the thermal inertia of the exhaust system are design options manufacturers can apply to all vehicle classes for improving engine cold start emission performance. It is not anticipated that heavy-duty vehicles with spark-ignition engines will utilize catalysts that are very close-coupled to the exhaust manifold (i.e., will not use a strategy similar to close-coupled catalyst locations found on passenger cars and light-duty trucks). The higher operating loads of these heavy-duty engines results in durability concerns due to high thermal loading. It is expected that manufacturers will work to optimize the thermal mass of the exhaust systems to reduce losses along with optimizing the underbody location of the catalyst. These changes are expected to improve the light-off time while not subjecting the catalysts to the higher thermal loadings from a close coupled location.

1.2.1.3.3 *Catalyst Design Changes*

There are several different catalyst design changes that can be implemented to reduce the time for the catalyst to light-off. Changes include modifying the substrate design, replacing a large volume catalyst with a cascade of two or more catalysts, and optimizing the loading and composition of the platinum group metals (PGM).

Progress continues to be made in the development of the catalyst substrates which provide the physical support for the catalyst components, which typically include a high surface area alumina carrier, ceria used for storing oxygen, PGM catalysts, and other components. A key design parameter for substrates is the cell density. Today, catalyst substrates can be fabricated with cell densities up 1,200 cells per square inch (cpsi) with wall thicknesses approaching 0.05 mm.

Increasing the surface area of the catalyst improves the performance of the catalyst. Higher substrate cell densities increase the surface area for a given catalyst volume. Higher surface areas improve the catalyst efficiency and durability reducing NMHC and NO_X emissions.

The key limitation of the higher cell density substrates is increased exhaust system pressures at high-load conditions. The cell density and substrate frontal area are significant factors that need to be considered to optimize the catalyst performance while limiting flow loss at high-load operation.

Engine speeds and load are low during the first 50 seconds of the FTP test and it is a challenge to achieve catalyst light-off during the cold start operation. One method for reducing the catalyst light-off time is to replace a single catalyst with two catalysts which when combined total the same volume as the single catalyst. Having a two-catalyst system that includes a close-coupled, front catalyst with comparatively reduced-volume reduces the heat needed to reach the light-off temperature for the front catalyst due to its location and reduced thermal mass. The front catalyst of the two-catalyst system will reach operating temperature before the individually larger volume single catalyst, reducing the light-off time of the system.

All other parameters held constant, increasing the PGM loading of the catalyst also improves the efficiency of the catalyst. The ratio of PGM metals is important as platinum, palladium, and rhodium have different levels of effectiveness promoting oxidation and reduction reactions. Therefore, as the loading levels and composition of the PGM changes, the light-off performance for both NMHC and NO_X need to be evaluated. Improved catalyst substrates and PGM loadings designs are additional effective approaches to reduce emissions and we anticipate manufacturers will incorporate advanced catalyst designs in their future emission strategies.[38] We used an advanced catalyst formulation in our HD SI feasibility demonstration for this rule (see Chapter 3.2).

1.2.1.3.4 *Secondary Air Injection*

By injecting air directly into the exhaust stream, close to the exhaust valve, combustion (hydrocarbon oxidation) can be maintained within the exhaust, creating additional heat and thereby further increasing the catalyst temperature. The air/fuel mixture must be adjusted to provide a richer exhaust gas for the secondary air to be effective.

Secondary air injection systems are used after the engine has started and once exhaust port temperatures are sufficiently high to sustain combustion in the exhaust port. When the secondary air pump is turned on, the engine control module increases the amount of fuel being injected into the engine. This fuel is added to decrease the air/fuel ratio in the cylinder to a ratio that is rich of stoichiometric. The exhaust contains significant quantities of CO and hydrocarbons. The rich exhaust gas mixes with the secondary air in the exhaust port and the combustion process continues, increasing the temperature of the exhaust and rapidly heating the manifold and close-coupled catalyst.[39,40]

Engines which do not use secondary air injection can only operate rich of stoichiometry for a minimal amount of time after a cold start as the added enrichment will cause increased NMHC emissions. The richer cold start calibration used with vehicles that have a secondary air injection system provides a benefit, as combustion stability is improved. In addition, the richer calibration is not as sensitive to changes in fuel volatility. Less volatile fuels found in the market may result in poor start and idle performance on engines calibrated to run lean during the cold operation. Engines which use secondary air and have a richer warm up calibration will have a greater combustion stability margin. Manufacturers may perceive this to be a benefit for the operation of their vehicles during the cold start and warm up phase.

Historically, secondary air injection has also been used to control CO and NMHC emissions during high load rich operation. These designs incorporated a mechanically driven air pump that continuously injects air into part of the catalyst to oxidize the CO and NMHC emissions, a particularly important emission control technology used during enrichment operation. With the improvements in electronic engine controls, all manufacturers discontinued the use of secondary air injection for the purpose of high load enrichment and instead used software models and other algorithms that maintain stoichiometric operation for slightly longer periods of operation than previously. These newer strategies however were designed to only provided a temporary emission control benefit of stoichiometric operation since the engines eventually go into enrichment modes for either power improvement or thermal protection at which point the CO and NMHC emissions are no longer controlled.

1.2.1.4 Improving Catalyst NO_X Efficiency during Fully Warmed-up Operation

For engines certified to the EPA 2007/2010 emission standards, significant quantities of NO_X emissions are produced by engines during warmed-up engine operation on the FTP. The stabilized NO_X emission levels will need to be reduced. Improving the NO_X performance of the engine can be achieved by improving the catalyst efficiency during warmed-up operation. As previously described, the performance of the catalyst can be improved by modifications to the catalyst substrate, increasing cell density, increasing PGM loadings, and, particularly important, reducing the sulfur level of gasoline. Three-way catalyst efficiency is also affected by frequency and amplitude of the air/fuel ratio. For some engines warmed-up catalyst NO_X efficiency can be improved by optimizing the air/fuel ratio control and limiting the amplitude of the air fuel ratio excursions. It is anticipated that a combination of changes will be made by manufacturers, including further improvements to air/fuel ratio calibration and catalyst changes including cell density and PGM loadings.

1.2.1.5 Reducing Enrichment

Heavy-duty vehicles tend to operate at high loads and catalyst durability can be a concern due to the increased thermal loading as the catalyst is moved closer to the cylinder head. Moving the catalyst closer to the exhaust manifold could result in increasing the time spent in fuel enrichment modes to ensure the catalyst temperatures are maintained below design thresholds, which if allowed to operate too hot could reduce the durability of the catalyst. Using fuel enrichment to control catalyst temperature while effective, causes significant increases in criteria pollutant emissions and also significant increases in fuel consumption.

1.2.1.5.1 *Exhaust Gas Temperature Measurement*

The methodology for determining when temperatures in the exhaust and in the catalyst are high enough to initiate thermal protection (i.e., enrichment) is almost exclusively done using software modeling of the thermally-limited components. This methodology can be effective at triggering enrichment when needed; however, if it is implemented in an excessively conservative manner where the temperature prediction is higher than the actual critical component temperature, it can result in unnecessary enrichment episodes which lead to substantial increases in all the emissions. Since the gasoline heavy-duty engines are designed for work and are expected to operate regularly at high loads where the exhaust temperatures become important concerns for component durability, any improvement in the accuracy of the methodology to provide enrichment protection will result in both reductions in emissions and improvements to durability.

A potential improvement over the current practice of using temperature modeling is use of hardware in the form of temperature measurement sensors in the exhaust at the most critical locations, generally at the catalyst inlet. While some light-duty variations are available, currently, only one gasoline HD product, the Nissan NV series of cargo vans with the 5.6L V8 has implemented temperature sensors.[41] However, temperature measurement sensors are very common in all diesel applications. While this improvement to the accuracy of temperature measurements in the exhaust may not result in emission reductions during the limited operation range of today's FTP and even SET certification test cycles, it will likely provide "real world" emission reductions compared to modeled temperature strategies that may conservatively trigger enrichment episodes prematurely.

1.2.1.5.2 *Continuous Stoichiometric Operation*

One concept that has been used in other non-road sectors (e.g., large SI engines operating indoors or confined areas) is requiring the engine to always operate at stoichiometry where the three-way catalyst is generally at peak efficiency for all emissions. To apply this strategy to heavy-duty highway SI engines, manufacturers must prevent the engine from entering enrichment. Enrichment can be avoided with upgrades to materials of specific components that are currently limited by high temperature constraints, or using the large degree of modern engine control authority to prevent the engine from entering areas of operation that require enrichment. Modern gasoline engines have several engine hardware components and calibration strategies that can reduce temperatures by modifying combustion or load characteristics, such as EGR, valve timing, electronic throttle airflow, cylinder deactivation, and other available methods. Some of these methods to remain in stoichiometric air-fuel control may result in a governing or detuning of the engine after a period of time to avoid prolonged high-power operation where stochiometric operation cannot be maintained due to increasing exhaust component temperatures; however, this may be an acceptable approach for a sector where sustained absolute engine power may not be necessary or as important as lower emissions and better fuel economy.

While we did not have the hardware and/or software required to directly control enrichment in our HD SI feasibility demonstration, in Chapter 3.2 we show that emissions can be well-controlled when an engine maintains stoichiometric operation. We expect engine manufacturers will continue to optimize their engine calibrations and limit enrichment as an effective means of reducing NMHC and CO.

1.2.1.6 Additional Emission Control Strategies

A strategy that may provide some degree of emission reductions involves down-speeding or governing of the engine operating range to keep the engine speeds and loads in areas where engine hardware and exhaust temperatures minimize needed enrichment for thermal protection. This strategy will allow the emission controls to remain in stoichiometric air-fuel control (i.e., closed loop) where the catalysts can maintain peak efficiency for NMHC, CO and NO_X for a broader range of operation.

The down-speeding approach was a technology discussed to reduce CO_2 emissions for the HD Phase 2 GHG rule[42] and has been successfully implemented in both gasoline and diesel applications to reduce GHG emissions. With the advent of modern electronic controls of both engine and transmission, the opportunity exists to precisely keep the engine in the optimal operating areas for reduced GHG and criteria emissions. Multi-speed automatic transmissions available in recent years in HD applications with wide ratios containing 6 or more forward gears have provided the opportunity to operate the engine in a more optimal fashion with little or no loss of vehicle performance and capabilities. This strategy is currently used by at least one HD gasoline engine manufacturer as indicated by its advertised maximum rated test speed (i.e., peak horsepower) of approximately 4000 RPM compared to the much higher speeds, over 4700 RPM of other HD applications.[43] This lower speed is made possible by transmission strategies preventing over-speeding, which allows the emission controls to operate in a much more desirable and lower emitting area of engine operation. We evaluated engine down-speeding as part of the HD SI feasibility demonstration presented in Chapter 3.2.

1.2.2 **Technology Description for PM Control**

Particulate matter emitted from internal combustion engines is a multi-component mixture composed of elemental carbon (or soot), semi-volatile organic compounds (SVOC), sulfate compounds (primarily sulfuric acid) with associated water, nitrate compounds and trace quantities of metallic ash. At temperatures above 1,300 K, fuel hydrocarbons without access to oxidants can pyrolyze to form particles of elemental carbon. Fuel pyrolysis can occur as the result of operation at richer than stoichiometric air-to-fuel ratio (primarily gasoline PFI engines and GDI engines), direct fuel impingement onto surfaces exposed to combustion (primarily GDI and diesel engines), and non-homogeneity of the air-fuel mixture during combustion (primarily diesel engines). Elemental carbon particles that are formed can be oxidized during later stages of combustion via in-cylinder charge motion and reaction with oxidants.

SVOCs are composed primarily of organic compounds from lubricant and partial combustion products from fuel. PM emissions from SVOC are typically gas phase when emitted from the engine and contribute to PM emissions via particle adsorption and nucleation after mixing with air and cooling. Essentially, PM-associated SVOC represent the condensable fraction of NMHC emissions. Sulfur and nitrogen compounds are emitted primarily as gaseous species (SO_2, NO and NO_2). Sulfate compounds can be a significant contributor to PM emissions from stratified lean-burn gasoline engines and diesel engines, particularly under conditions where PGM containing exhaust catalysts used for control of gaseous and PM emissions oxidize a large fraction of the SO_2 emissions to sulfate (primarily sulfuric acid). Sulfate compounds do not significantly contribute to PM emissions from spark-ignition engines operated at near stoichiometric air-fuel ratios due to insufficient availability of oxygen in the exhaust for oxidation of SO_2 over PGM catalysts.

Elemental carbon PM emissions can be controlled by:

- Reducing fuel impingement on piston and cylinder surfaces
- Inducing charge motion and air-fuel mixing via charge motion (e.g., tumble and swirl) or via multiple injection (e.g., GDI and diesel/common rail applications)
- Injection strategies that eliminate opportunity for PM forming conditions (open valve injection on PFI)
- Reducing or eliminating operation at net-fuel-rich air-to-fuel ratios (PFI gasoline and GDI applications)
- Use of wall-flow or partial-wall-flow exhaust filters (GPF)

SVOC PM emissions can be controlled by:

- Reducing lubricating oil consumption
- Improvements in exhaust catalyst systems used to control gaseous NMHC emissions (e.g., increased PGM surface area in the catalyst, improvements in achieving catalyst light-off following cold-starts, etc.)

1.2.3 Technologies to Address Evaporative and Refueling Emissions

As exhaust emissions from gasoline engines continue to decrease, evaporative emissions become an increasingly significant contribution to overall HC emissions from gasoline-fueled vehicles. To evaluate the evaporative emission performance of current production heavy-duty gasoline vehicles, EPA tested two heavy-duty vehicles over running loss, hot soak, three-day diurnal, on-board refueling vapor recovery (ORVR), and static test procedures. These engine-certified "incomplete" vehicles meet the current heavy-duty evaporative running loss, hot soak, and three-day diurnal emission requirements. However, as they are certified as incomplete vehicles, they are not required to control refueling emissions and do not have ORVR systems. Results from the refueling testing confirm that these vehicles have much higher refueling emissions than gasoline vehicles with ORVR controls.[44,45] The results for the ORVR tests are shown in Table 1-3. A discussion on the test procedure limitations and estimated modeled results from this test program is in Section 2.3.2.

Table 1-3: ORVR results with modeled values for test procedure limitations

	SHED (grams)	Modeled SHED (grams)	Average (g/gal)	Modeled Average (g/gal)	Current Refueling Standard (g/gal)
Ford E-450	114	168	2.3	3.34	0.2
	108		2.2		
Isuzu NPR	72	86	2.8		
	71		2.8		

Opportunity exists to extend the usage of the refueling evaporative emission control technologies already implemented in complete heavy-duty gasoline vehicles to the engine-certified incomplete gasoline vehicles in the over-14,000 lb. GVWR category. The primary technology we are considering is the addition of ORVR, which was first introduced to the chassis-certified light-duty and heavy-duty applications beginning in MY 2000 (65 FR 6698, February 10, 2000). An ORVR system includes a carbon canister, which is an effective technology designed to capture HC emissions during refueling events when liquid gasoline displaces HC vapors present in the vehicle's fuel tank as the tank is filled. Instead of releasing the HC vapors into the ambient air, ORVR systems recover these HC vapors and store them for later use as fuel to operate the engine.

The fuel systems on these over-14,000 pound GVWR incomplete heavy-duty gasoline vehicles are similar to complete heavy-duty vehicles that are already required to incorporate ORVR. These incomplete vehicles may have slightly larger fuel tanks than most chassis-certified (complete) heavy-duty gasoline vehicles and are somewhat more likely to have dual fuel tanks. These differences may require a greater ORVR system storage capacity and possibly some unique accommodations for dual tanks (e.g., separate fuel filler locations), but we expect they will maintain a similar design. Figure 1-6 presents a schematic of a standard ORVR system.

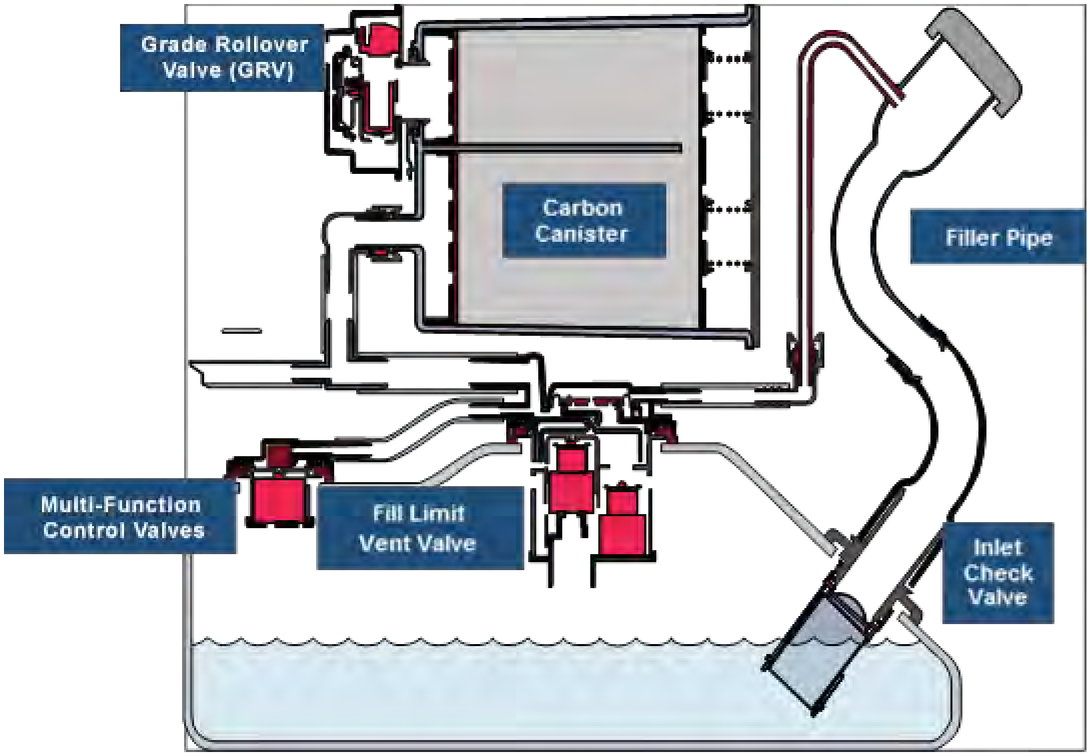


Figure 1-6: Schematic of an ORVR system[J]

1.2.3.1 Filler Pipe and Seal

In an ORVR system, the design of the filler pipe, the section of line connecting the point at which the fuel nozzle introduces fuel into the system to the gas tank, is integral to how fuel vapors displaced during a fuel fill will be handled. The filler pipe is typically sized to handle the maximum fill rate of liquid fuel allowed by law while also integrating one of two methods to prevent fuel vapors from exiting through the filler pipe to the atmosphere: a mechanical seal or a liquid seal approach. A dual fuel tank chassis configuration may require a separate filler pipe and seal for each fuel tank.

The mechanical seal is typically located at the top of the filler neck at the location where the fuel nozzle is inserted into fuel neck. The hardware piece forms a seal against the fuel nozzle by using some form of a flexible material (usually a plastic material) that makes direct contact with the fuel station fuel-filling nozzle to prevent fuel vapors from exiting the filler pipe as liquid fuel is pumped into the fuel tank. In the case of capless systems, this seal may be integrated into the spring-loaded seal door that opens when the nozzle is inserted into the filler pipe receptacle. There are concerns with a mechanical seal's durability due to wear over time, and its ability to maintain a proper seal with unknown service station fill nozzle integrity and variations beyond design tolerances.

The liquid seal approach uses the size and bends of the filler pipe to cause a condition where the entire cross-section of the filler pipe is located in the fuel tank or close to the entry into the fuel tank and is full of the incoming liquid fuel preventing fuel vapors from escaping up and out

[J] Stant ORVR System http://stant.com/orvr/orvr-systems/

through the filler pipe. By creating a solid column of liquid fuel in the filler pipe, the liquid seal approach does not require a mechanical contact point with the fill nozzle to prevent escape of vapors. The liquid seal has been the predominant sealing method implemented in the regulated fleet in response to the ORVR requirements.

1.2.3.2 ORVR Flow Control Valve

As described above, the sealing of the filler pipe prevents the fuel vapors from escaping into the ambient air; however, the fuel vapors that are displaced by the incoming liquid fuel need to be routed to the canister. In order to properly manage the large volume of vapors during refueling that need to be controlled, most ORVR systems have implemented a flow control valve that senses that the fuel tank is getting filled with fuel and triggers a unique low-restriction flow path to the canister. This flow path is specifically used only during the refueling operation and is unique in that it provides the ability to quickly move larger volumes of fuel vapors into the tank than normally required under other operation outside of refueling events. The flow control valve will allow this larger flow volume path while refueling but then return to a more restrictive vapor flow path under all other conditions, including while driving and while parked for overnight diurnals.

The flow control valve is generally a fully-mechanical valve system that utilizes connections to the fuel tank and filler pipe to open and close vapor pathways with check valves and check balls and pressure switches via diaphragms. The valve may be integrated into the fuel tank and incorporate other aspects of the fuel handling system ("multi-function control valve" in Figure 1-6) including roll-over valve, fuel and vapor separators to prevent liquid fuel from reaching the canister, and other fuel tank vapor control hardware. Depending on the design, the filler pipe may also be integrated with the flow control valve to provide the necessary pressure signals. A dual fuel tank chassis configuration may require a separate flow control valve for each fuel tank.

1.2.3.3 Canister

The proven technology to capture and store fuel vapors has been activated charcoal. This technology has been used in vehicles for over 50 years to reduce evaporative emissions from sources such as fuel tanks and carburetors. When ORVR was originally discussed, existing activated charcoal technology was determined to be the appropriate technology for the capture and storage of refueling related fuel vapors. This continues to be the case today, as all known ORVR-equipped vehicles utilize some type of activated charcoal.

The activated charcoal is contained in a canister, which is made from a durable material that can withstand the fuel vapor pressures, vibration, and other durability concerns. For vehicles without ORVR systems, canisters are sized to handle evaporative emissions for the three-day diurnal test with the canister volume based on the fuel tank capacity. A dual fuel tank chassis configuration may require a separate canister for each fuel tank.

1.2.3.4 Purge Valve

The purge valve is the electro-mechanical device used to remove fuel vapors from the fuel tank and canister by routing the vapors to the running engine where they are burnt in the combustion chamber. This process displaces some amount of the liquid fuel required from the fuel tank to operate the engine and results in a small fuel savings. The purge valve is controlled by the engine or emission control electronics with the goal of removing the necessary amount of captured fuel vapors from the canister in order to prepare the canister for subsequent fuel vapor handling needs of either the next refueling event or vapors generated from a diurnal event. All on-road vehicles equipped with a canister for evaporative emissions control utilize a purge valve. Depending on the design, a dual fuel tank chassis configuration may require a separate purge valve for each fuel tank.

1.2.3.5 Design considerations for Unique Fuel Tanks

The commercial truck market gasoline applications incorporate several fuel tank options that may require unique ORVR design considerations. While most commercial vehicle fuel tanks are similar to the already ORVR-compliant complete vehicles in the 8500 to 14,000 GVWR class, some of the commercial vehicles include larger tank sizes (50 to 70 gallons) or may have a dual tank option. As described above, the canister sizing will be a function of the required amount of fuel vapor handling during refueling. Larger fuel tanks will require larger canisters with more activated charcoal than historically found in other gasoline vehicles. Some design challenges will likely exist in designing the canister system to handle the large vapor volumes while balancing the restriction to flow through the larger activated charcoal containing canisters.

Dual fuel tank systems, which have very limited availability, may also require some unique design considerations. Typically, the canister is located in very close proximity to the fuel tank to properly manage the refueling fuel vapors efficiently with minimal distance between the tank and canister. Dual fuel tanks may require duplicate ORVR systems to have the necessary flexibility to manage the refueling vapors, particularly since the fuel tanks are filled independently through separate filler pipe assemblies.

A small portion of the commercial truck market gasoline applications have fuel tanks that are similar in design to diesel fuel tanks located on the outside of the frame. These tanks are typically cylindrical or rectangular in shape with the gas cap directly on the top of the tank and do not have a fill neck. These type of fuel tanks may require unique approaches such as a mechanical seal built into the fuel tank filling location where the fuel cap is normally located, or they may require a design that adds a filler pipe for a liquid seal approach.

1.3 Fuels Considerations

Both the compression- and spark-ignition engine technologies discussed above are capable of running on alternative fuels (e.g., natural gas, biodiesel). We have typically applied the gasoline- and diesel-fueled engine standards to alternatively-fueled engines based on the combustion cycle of the alternatively-fueled engine: applying the gasoline-fueled engine standards to spark-ignition engines and the diesel-fueled engine standards to compression-ignition engines. The sections below discuss some of the available alternative fuels in more depth.

1.3.1 <u>**Natural gas**</u>

With relatively low natural gas prices (compared to their peak values) in recent years, the heavy-duty industry has become increasingly interested in engines that are fueled with natural gas. It has some emission advantages over diesel, with lower engine-out levels of both NO_X and PM. Several heavy-duty CNG engines have been certified with NO_X levels better than 90 percent below US 2010 standards. However, because natural gas must be distributed and stored under pressure, there are additional challenges to using it as a heavy-duty fuel.

Liquified Petroleum Gas (LPG) is also used in certain lower weight-class urban applications, such as airport shuttle buses, school buses, and emergency response vehicles. LPG use is not extensive, nor do we project it to grow significantly in the timeframe of this rulemaking.

1.3.2 **Biodiesel**[K]

Over the last decade, biodiesel content in diesel fuel has increased under the Renewable Fuels Standard. In 2010, less than 400 million gallons of biodiesel were consumed in the U.S., whereas in 2018, over 2 billion gallons of biodiesel were being blended into U.S. diesel fuel. While the biodiesel content in diesel fuel averaged around 3.5 percent in 2018, biodiesel levels range from 0 to 20 volume percent in highway diesel fuel. As discussed further below, with increasing volumes of biodiesel in the fuel stream, greater attention has been given to the influence of biodiesel on highway diesel fuel quality. The following sub-sections discuss how biodiesel is produced and the importance of purification processes for removing potential metal containments, standards for biodiesel fuel quality, potential impacts of metals in biodiesel fuel on diesel engines, aftertreatment systems, and emissions, as well as efforts by EPA and others to test the metal content of biodiesel samples across the country.

1.3.2.1 Biodiesel Production and a Potential for Metals

Biodiesel can be made from various renewable sources, such as vegetable oil, animal fat, or waste cooking oil. It is produced through transesterification of the oil or fat with methanol, which results in mono-alkyl esters and the co-product glycerin. The process occurs in the presence of a catalyst, typically sodium or potassium methoxide or hydroxide.[46] Following transesterification and separation of the glycerin, the biodiesel must be purified, which is usually done by extracting, distilling, or filtering the impurities into water.

The purification process is essential to address the potential for metal and other impurities in biofuel. There are number of potential sources of metal contamination in biofuel production. These include:

1) Vegetable oil seeds used to produce feedstock contain high concentrations of sodium (Na), potassium (K), calcium (Ca), magnesium (Mg), and potassium (P), as well as aluminum (Al), iron (Fe), manganese (Mn), zinc (Zn), and smaller concentrations of other metals.[47]

2) The potassium and sodium methoxide catalysts which break down triglycerides to methyl esters (NaOH and KOH can also be used) can contribute metals to biodiesel. These metals can form soaps with free fatty acids, and the soaps in both the metal esters and glycerine forms are reacted with acid (hydrochloric acid) to convert the soaps to free fatty acids so they can be more easily removed. Sodium hydroxide is added to neutralize any acid added to eliminate soaps.

3) Methyl esters are washed, distilled or filtered to remove the metals added as catalysts. The wash water is recycled, and metal ions can accumulate in the wash water. Hard wash water containing $CaCO_3$, $Mg(OH)_2$, $CaSO_4$ is found in Rocky Mountain states and the Midwest, and these water-soluble compounds can accumulate in the residual water found in biodiesel.

[K] Biodiesel is different from renewable diesel which is much more similar to petroleum refined diesel than it is to biodiesel.

4) The medium used to filter methyl esters could also contribute to metals in the biodiesel. The filter material is typically made up of diatomaceous earth which is primarily silica containing alumina, iron oxide and calcium oxide. In addition, small amounts of calcium or magnesium can be added to the fuel from the purification process.[48,49]

1.3.2.2 Standards for Biodiesel Fuel Quality

Biodiesel quality, including metal content, is specified by ASTM D6751-19 for B100 fuels. ASTM D6751-19 specifies a limit of 5 ppm for combined Na and K (group 1A metals) and a limit of 5 ppm for combined Ca and Mg (group 2A metals) using the EN14538 inductively coupled plasma optical emission spectroscopy (ICP-OES) measurement method.[50] ASTM D6751-18 also specifies a 10-ppm limit on P (group 5 metal) using the ASTM D4951 inductively coupled plasma atomic emission spectroscopy (ICP-AES) measurement method.[51] The limits on metals in ASTM D6751 are meant to be protective when biodiesel is used in blends (e.g., B20, B10). Fuel quality for biodiesel blends in the B6 to B20 range is specified by ASTM D7467-19.[52] This specification does not contain a metal limit for these biofuel blends because, as the method states, the concentration would likely be too low to measure using the ICP-OES method specified (EN 14538). Similarly, D975 addresses B0 to B5 and does not have a metals specification (just a total ash % limit of 0.01%).[53] Thus, the basis for control of metals in biodiesel blends is control of the B100 blend stock. This is because if the B100 fuel is under the ASTM D6751-19 limit, the combined Na + K and Mg + Ca will likely be below 1 ppm respectively for B20 and lower blends. Yet, the actual metal content of today's fuels can be challenging to quantify when it is lower than the 1 ppm level specified for B20 and lower blends, because of the detection limit of the current test methods. The detection limit of the EN14538 is 1 ppm for each metal, and the method includes a statement if the metal is below the limit of detection of the method, then it is not included in the reporting calculation. Efforts to quantify biodiesel metal contents below 1 ppm are discussed in 1.3.2.5 below.

1.3.2.3 Potential Impacts of Metals on Engine and Emission Control Devices

Across a range of concentrations, metals in biodiesel can be present as ions, abrasive solids or soluble metallic soaps. Abrasive solids can contribute to wear of fuel system components, pistons and rings, as well as contribute to engine deposits. Soluble metallic soaps have little impact on wear but may contribute to diesel particulate filter plugging and engine deposits. Metal accumulation in diesel particulate filters can increase pressure drops and result in shorter times between maintenance intervals.[54,55] A level of 1 mg/kg (1 part per million) of trace metal in the fuel result in an estimated accumulation of about 22 g of trace metal in diesel particulate filters per 100,000 miles (assuming a fuel economy of 15 mpg and 100% trapping efficiency).[54]

Metallic fuel contaminants can also accumulate on fuel injectors, or be converted to oxides, sulfates, hydroxides or carbonates in the combustion process, which forms an inorganic ash that can deposit onto the exhaust emission control devices found in modern diesel engines.[56] Alkali metals are well known poisons for catalysts used in emission control devices, and have been shown to negatively impact the mechanical properties of ceramic substrates.[57,58] Alkali metal hydroxides such as Na and K are volatilized in the presence of steam and can, therefore, penetrate the catalyst washcoat or substrate.

1.3.2.4 Potential for Emissions Impacts of Metals in Biodiesel

Numerous studies have collected and analyzed emission data from diesel engines operated on biodiesel blended diesel fuel with controlled amounts of metal content.[55,59,60,61,62,63,64,65,66] Some of these studies show an impact on emissions, while others do not. However, four factors need to be considered when reviewing these studies:

1. These studies were conducted using accelerated aging protocols and exposure to these metals from the fuel consumed in a more conventional manner could cause different effects (larger or smaller) than what these studies show,
2. The emissions testing studies were designed to test the effect of metal content in biodiesel if the metal content was at the ASTM limit, however, as shown below, biodiesel likely contains a lower metal content than the standard.
3. Different manufactures use different catalyst formulations and different physical layouts for emission aftertreatment systems, and while one manufacturer might be less susceptible to metals contamination, others may be more affected. This issue relates to factor #4:
4. When these studies examined the effect of metals on heavy-duty engines, they studied the impact of these metals based on current engine and aftertreatment configurations over the current regulatory useful life. This rule requires heavy-duty engines to comply with a more stringent NO_X standard and a longer useful life. The longer useful life will expose the aftertreatment devices to increased amounts of metals during the useful life (compared to many of today's engines that often operate beyond the current regulatory useful life and are already exposed to more metals after their regulatory useful life). Also, the engine manufacturers may change the composition and configuration of their aftertreatment devices, which could affect how fuel metals affect the aftertreatment devices.

Brookshear et al. 2012 studied the impact of Na on heavy-duty diesel engine aftertreatment devices.[61,L] In this accelerated aging study, they doped a B20 fuel to 5,000 ppm each of Na and S and aged to an equivalent 435,000 miles. They found impacts on SCR function if the SCR was positioned before the DPF. There was no impact on the DOC or DPF.

Lance et al. 2016 also studied the effect of Na on heavy-duty diesel engine aftertreatment.[63,M] They doped their B20 fuel with Na to a level of 14 ppm or 14 times the pseudo 1 ppm limit of a B20 fuel and accelerated aged the aftertreatment out to 435,000 miles. The results indicated an acceleration of DPF ash buildup and platinum group metal migration from the DOC/CDPF to the SCR. The results of the system performance, including degradation in performance, are shown in Figure 1-7. The results indicated that the degradation in NO_X performance can be attributed to degradation of all aftertreatment components.

[L] Author affiliations: University of Tennessee and Oak Ridge National Lab
[M] Author affiliations: Oak Ridge National Lab, Cummins, and National Renewable Energy Lab

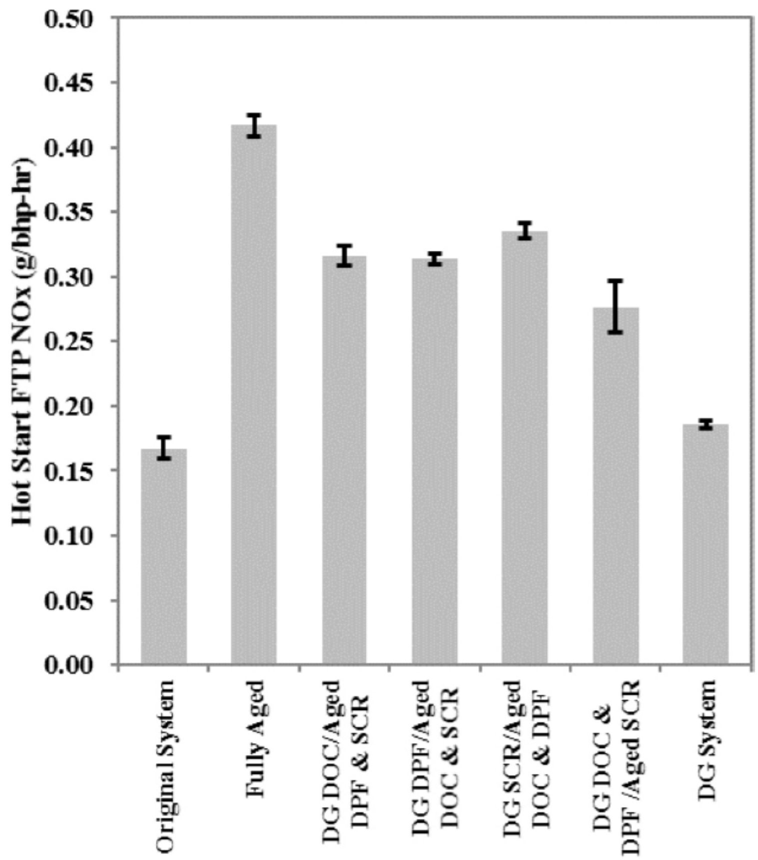


Figure 1-7: SCR performance over the hot start HDDE FTP[63, N]

Williams et al. 2011 studied the effect of Na and Ca on a 2008 non-road 8.8L Caterpillar diesel engine, a MAN D2066 10.5 L diesel engine, and a 2008 Cummins 8.3L diesel engine.[66,O] They doped their B20 fuel to 27 times the pseudo 1 ppm Na and Ca limit of a B20 fuel and accelerated aging of the emission control systems out to 150,000 and 435,000 miles. The results showed no significant degradation in the thermo-mechanical properties of cordierite, aluminum titanate, or silicon carbide DPFs after exposure to 150,000-mile equivalent biodiesel ash and thermal aging. It is estimated that the additional ash from 150,000 miles of biodiesel use would also result in moderate increases in exhaust backpressure for a DPF. A decrease in DOC activity was seen after exposure to 150,000-mile equivalent aging, resulting in higher HC slip and a reduction in NO_2 formation. The exposure of a cordierite DPF to 435,000-mile equivalent aging resulted in a 69% decrease in the thermal shock resistance parameter. The metal-zeolite SCR catalyst experienced a slight loss in activity after exposure to 435,000-mile equivalent aging. This catalyst, placed downstream of the DPF, showed a 5% reduction in overall NO_X conversion activity over the HDDT test cycle.

[N] Note that DG indicates that the specific component is not an aged part, but a new degreened part
[O] Author affiliations: National Renewable Energy Lab, Manufacturers of Emission Controls, BASF, Caterpillar, and Umicore AG

Williams et al. 2013 studied the effect of Na, K and Ca on a 2011 LD 6.7L diesel engine aftertreatment.[56,P] They doped their B20 fuel to 14 times the pseudo 1 ppm Na and Ca limit of a B20 fuel and accelerated aged the emission control systems out to 150,000 miles. The authors aged sets of production exhaust systems that included a DOC, SCR catalyst, and DPF. Four separate exhaust systems were aged, each with a different fuel: ULSD containing no measurable metals, B20 containing sodium, B20 containing potassium, and B20 containing calcium. Analysis of the aged catalysts included Federal Test Procedure emissions testing with the systems installed on a Ford F250 pickup, bench flow reactor testing of catalyst cores, and electron probe microanalysis (EPMA). The thermo-mechanical properties of the aged DPFs were also measured.

EPMA imaging of aged catalyst parts found that both the Na and K penetrated into the washcoat of the DOC and SCR catalysts, while Ca remained on the surface of the washcoat. Bench flow reactor experiments were used to measure the standard NO_X conversion, NH_3 storage, and NH_3 oxidation for each of the aged SCR catalysts. Flow reactor results showed that the first inch of the SCR catalysts exposed to Na and K had reduced NO_X conversion through a range of temperatures (Figure 1-8 and Figure 1-9) and also had reduced NH_3 storage capacity. The SCR catalyst exposed to Ca had similar NO_X conversion and NH_3 storage performance compared to the catalyst aged with ULSD.

Chassis dynamometer vehicle emissions tests were conducted with each of the aged catalyst systems installed onto a Ford F250 pickup. Regardless of the evidence of catalyst deactivation seen in flow reactor experiments and EPMA imaging, the vehicle successfully passed the 0.2 gram/ mile NO_X emission standard with each of the four aged exhaust systems. This indicates that if catalyst volumes are chosen to account for degradation, the emission control system can accommodate some loss in catalyst activity since deactivation occurred only in the first inch of the catalyst and did not affect overall NO_X emissions.[56]

[P] Author affiliations: National Renewable Energy Lab, Oak Ridge National Lab, Manufacturers of Emission Controls, BASF, Ford, and University of Tennessee

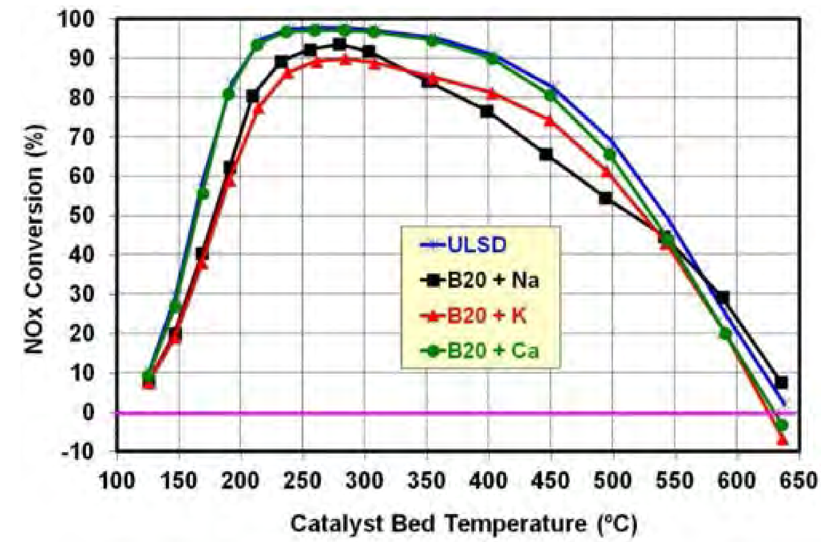


Figure 1-8: SCR NOx conversion for the first inch of aged SCR catalysts.[56]

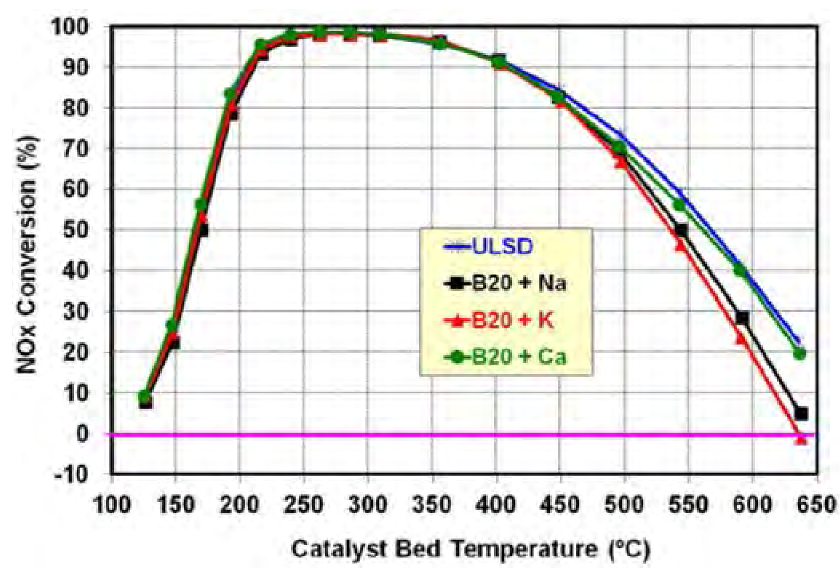


Figure 1-9: SCR NOx conversion for the seventh inch of aged SCR catalysts. [56]

1.3.2.5 Testing for Metals in Biodiesel

The National Renewable Energy Laboratory (NREL) has conducted several studies on the metal content of biodiesel. The NREL studies generally look at the fuel quality in fuel samples taken across from the country. In some cases, samples are taken at the refinery where B100 was sampled and in other cases they are taken at the pump. These studies provided analysis of metal content for fuel samples collected and analyzed in the 2006, 2007, 2008, 2011, and 2016 timeframes. The 2006, 2007, and 2011 studies analyzed the metal content of B100 fuel, while the 2008 and 2016 studies analyzed the metal content of biodiesel blends up to approximately B20. Some samples taken during the 2006 study were acquired prior to the finalization of the combined Na + K limit in ASTM D6751 (May 2006). [48,49,67,68,69,70,71]

These results indicate that over the 2006 to 2018 time frame the incidence of off specification biofuel decreased over time. A summary of the off-specification samples can be found in Table 1-4.

Table 1-4: NREL Fuel Samples off Specification for ASTM D6751 (or equivalent B20) limit for Na + K and Ca + Mg.

NREL Fuel Study Year	Biodiesel Content	Number of Samples off spec for Na + K	Number of Samples off spec for Ca + Mg	Total Number of Samples
2006 (pre-D6751)	B100	7	1	24
2006 (post-D6751)	B100	0	1	15
2007	B100	3	3	55
2008	B0.2 to B90	6	0	34
2011	B100	1	0	67
2016	B0 to B22	0	1	35
2017	B100	1	0	459
2018	B100	0	0	491

The NREL studies prior to 2016 focused on identifying gross exceedances of the blends and blend stocks. As noted above, the analytical method specified in ASTM D6751-18 (EN 14538) affords a detection limit of 1 ppm, which is adequate to ensure whether or not biofuel blend stocks (B100) are compliant with the Na + K and Ca + Mg limits. In addition to determining compliance with the ASTM D6751-18 limit, it is also important to determine the actual metal content of these fuels in order to assess the levels that aftertreatment systems will be exposed to over their full useful life. The 2016 NREL study used measurement equipment and procedures capable of testing down to lower metal levels, which is useful for understanding the actual metal content of biodiesel blends. Table 1-5 summarizes the test procedures and level of detection (LOD) levels for each year of the NREL studies.

Table 1-5: Test procedure LODs for NREL studies.

Year of NREL Study	Test Procedure	Level of Detection (ppm)			
		Na	K	Ca	Mg
2006	ASTM D5185	?	?	?	?
2007	ASTM D7111	1	1	0.1	0.1
2008	ASTM D7111	0.5	0.5	0.5	0.1
2011	ASTM D7111	1	1	0.1	0.1
2016	UOP-389	0.1	0.1	0.1	0.1
2016	ICP-MS	0.029	0.001	0.0005	0.001
2017	EN 14538	1	1	1	1
2018	EN 14538	1	1	1	1

In the 2016 study, NREL took steps to improve their understanding of testing accuracy and improve the testing resolution of the analysis by utilizing three different measurement methods, two of which afforded very low detection limits. In this study, biodiesel blends were analyzed for metal content using the Universal Oil Products Method 389 (UOP-389), Microwave Plasma Atomic Emission Spectroscopy (MP-AES), and Inductively Coupled Plasma Mass Spectroscopy (ICP-MS) methods.[72] This allowed for comparisons of metal content from the three different testing methods. Almost all the results for the MP-AES testing method were below the LOD, though, so this method will not be further discussed. The results of the study for UOP-389 and the ICM-MS are presented in Table 1-6.

Table 1-6: NREL 2016 Metals results for UOP-389 and ICP-MS[72]

State	Biodiesel Content	Error	UOP-389 ICP-AES				ICP-MS				
			Na	K	Ca	Mg	Na	K	Ca	Mg	Fe
	Vol %	Vol %	ppm	ppm	ppm	ppm	ppm	ppm	ppm	ppm	ppm
FL-1 (fleet)	22.01	1.76	0.14	0.04	0.08	0.01	(0.029)	0.021	0.008	0.026	0.065
FL-2	18.38	1.59	0.04	0.01	0.05	0.01	(0.029)	0.033	0.002	0.014	0.146
GA	19.36	1.63	0.06	0.02	0.06	0.01	(0.029)	0.034	0.002	0.011	1.44
NC-1 (fleet)	20.4	1.68	0.06	0.01	0.09	0.02	(0.029)	0.02	0.002	0.01	0.004
NC-2	0	0.71	0.02	0.01	0.02	0.01	No Data	No Data	No Data	No Data	No Data
PA-1	17.73	1.56	0.09	0.01	0.05	0.02	(0.029)	0.014	0.002	0.009	0.012
PA-2 (fleet)	20.31	1.68	0.08	0.02	0.04	0.03	(0.029)	0.031	0.002	0.019	0.017
MA	21.35	1.73	0.04	0.01	0.04	0.01	(0.029)	0.02	0.002	0.015	0.016
VA (fleet)	18.56	1.6	0.08	0.02	0.06	0.02	(0.029)	0.028	0.012	0.011	0.086
IA (fleet)	16.1	1.48	0.09	0.03	0.04	0.01	(0.029)	0.1	0.017	0.014	0.249
IL	15.3	1.44	0.07	0.01	0.03	0.01	(0.029)	0.02	0.007	0.019	0.005
IN	20.34	1.68	0.2	0.07	0.1	0.03	(0.029)	0.237	0.106	0.031	0.075
KS	17.27	1.53	0.09	0.02	0.53	0.06	(0.029)	0.11	1.366	0.166	1.557
KY	20.11	1.67	0.05	0.01	0.03	0.01	(0.029)	0.022	0.002	0.015	0.069
MI	20.51	1.69	0.11	0.02	0.06	0.01	(0.029)	0.045	0.122	0.025	3.117
MN-1	10.68	1.22	0.04	0.01	0.04	0.01	(0.029)	0.016	0.002	0.025	0.172
MN-2	10.61	1.22	0.02	0.01	0.03	0.01	(0.029)	0.016	0.002	0.048	0.003
MO	19.44	1.64	0.09	0.01	0.05	0.01	(0.029)	0.034	0.01	0.012	0.019
OH-1	20.4	1.68	0.05	0.01	0.05	0.01	(0.029)	0.013	0.002	0.011	0.033
OH-2	8.84	1.13	0.01	0.01	0.03	0.01	(0.029)	0.015	0.002	0.007	0.003
TN-1	20.24	1.68	0.1	0.01	0.06	0.01	(0.029)	0.019	0.003	0.006	0.085
TN-2 (fleet)	20.02	1.67	0.24	0.02	0.08	0.01	(0.029)	0.029	0.012	0.009	0.361
LA	12.26	1.3	0.04	0.01	0.03	0.01	No Data	No Data	No Data	No Data	No Data
TX-1	17.14	1.53	0.06	0.01	0.08	0.01	(0.029)	0.01	0.002	0.005	0.003
TX-2	1.4	0.78	0.07	0.01	0.04	0.01	(0.029)	0.012	0.002	0.006	0.004
TX-3	1.2	0.77	0.04	0.01	0.03	0.01	(0.029)	0.011	0.002	0.005	0.004
CO	21.99	1.76	0.09	0.01	0.05	0.01	(0.029)	0.023	0.094	0.035	0.037
ID (fleet)	19.64	1.65	0.22	0.03	0.12	0.03	(0.029)	0.085	0.065	0.034	0.044
AZ (fleet)	20.39	1.68	0.07	0.01	0.04	0.01	(0.029)	0.085	0.065	0.034	0.044
CA-1	18.84	1.61	0.05	0.02	0.03	0.01	(0.029)	0.016	0.002	0.019	0.009
CA-2	22.12	1.77	0.07	0.01	0.04	0.01	(0.029)	0.016	0.002	0.011	0.03
CA-3 (fleet)	19.9	1.66	0.06	0.01	0.04	0.01	(0.029)	0.033	0.002	0.018	0.005
NM	19.61	1.65	0.1	0.02	0.05	0.01	(0.029)	0.025	0.002	0.005	0.284
OR-1	20.1	1.67	0.06	0.02	0.03	0.02	(0.029)	0.012	0.002	0.005	0.009
OR-2	20.15	1.67	0.05	0.01	0.05	0.01	(0.029)	0.021	0.002	0.005	0.011
		Average	0.079	0.016	0.064	0.014	(0.029)	0.037	0.058	0.021	0.243
		Average Na + K and Ca + Mg	0.047		0.039		0.033		0.040		

Note: Values in parenthesis are below the detection limit and are reported at the detection limit.

Differences in the results between the two methods could be due to interferences by other metals, or other aspects of the molecules, with the measuring method. The measurement differences could also be due to the unique ionization efficiency of each element and how well the instrument ionizes the element of interest. Even small differences could impact the results at sub 1-ppm levels. The difference in sample preparation techniques can also have a significant effect on the results. The UOP-389 method uses acid digestion followed by ashing, while the ICP-MS method used a simpler preparation of sample dilution and direct analysis. The UOP-389 method was developed for the analysis of petroleum products and blending components, including biodiesel blends, and uses a wet ashing method that is unique to this procedure, which is why it was selected for this project.

Although several samples contained elevated amounts of Ca and Mg (KS, MI, IN, and CO) that were well above the level of other samples, these trace levels would still be very low in the B100 blend stock, with the exception of one sample (KS). Overall, the NREL data suggest that metal contents in biodiesel have decreased over time and, as of 2018, are generally very low across samples. Nevertheless, small sample sizes could be biasing the results.[46,47,67,68,69,70,71]

To that end, in 2019 an engine manufacturer raised concerns to EPA that biodiesel is the source of high metal content in highway diesel fuel, and that higher biodiesel blends, such as B20, are the principal problem.[73] The engine manufacturer observed higher than normal concentrations of alkali and alkaline earth metals (Na, K, Ca, and Mg) in their highway diesel fuel samples, and observed fouling of the aftertreatment control systems of their engines, which caused an associated increase in emissions. The engine manufacturer sampled the ash that was fouling their fuel injectors and aftertreatment devices and determined the ash to be composed of sodium sulfate, sodium carboxylates, and sodium chloride, which they claimed were from biodiesel. The engine manufacturer recommends limiting biodiesel blends to 5 percent biodiesel (B5). After hearing engine manufacturer concerns about the metal content in biodiesel in early 2019, EPA began to focus a previously developed fuel sampling program on biodiesel metal content. Below, we summarize the information that we obtained through that sampling program.

Separate from hearing about engine manufacturer concerns with biodiesel metal content, EPA began a process to determine the metal content of different fuels in early 2016. The most prominent concern to EPA at that time was the blending of less refined natural gas liquids with ethanol to produce E85 blends; however, EPA recognized the need to understand the metal content of all fuels. EPA initiated a sampling effort in late 2016 to obtain samples of diesel fuel, gasoline, natural gas liquids, jet fuel, biodiesel and ethanol. The samples were collected in acid-washed glass bottles using a clean hands/dirty hands sampling procedure to reduce the chance for contamination during the sampling process. Fuel samples were collected from both small and large fuel production plants, in case facility size plays a role in the metal content of the fuel. Samples were also taken in different geographical regions. Fuel production facilities likely use different feedstocks based on their geographical region, and the different feedstock types could affect the fuels metal content. Because of the cost and effort involved in obtaining these fuel samples, a limited number of samples of each fuel type were obtained. This screening study could be expanded upon the detection of high metal content in any of the fuels.

Approximately 100 samples were obtained across the various fuel types. A subset of these samples (27 B100 samples) were recently sent to the California Department of Food and Agriculture (CDFA) laboratory. These samples were analyzed for biodiesel regulated metals, including Na, K, Ca, Mg and P, and also tested for Molybdenum (Mo), Boron (B), Barium (Ba), Copper (Cu), Manganese (Mn), Silica (Si), Titanium (Ti), Vanadium (V) and Zinc (Zn). The

California lab utilized the ASTM D7111-16 ICP-AES method that returned detection limits of 0.023 (Na), 0.052 (K), 0.013 (Ca), 0.004 (Mg), 0.001 (P), 0.006 (Mo), 0.013 (B), 0.001 (Ba), 0.005 (Cu), 0.001 (Mn), 0.017 (Si), 0.003 (Ti), 0.002 (V), and 0.005 (Zn) ppm.[74] The results of the analysis are shown in Table 1-7 and Table 1-8.

Na was above the detection limit for 22 of the samples, with the highest result at 564 ppb. K was only above the detection limit for 3 of the samples, with the highest result at 660 ppb. Ca was above the detection limit for 9 of the samples, with the highest result at 551 ppb. Mg was only above the detection limit for 5 of the samples, with the highest result at 133 ppb. The highest result for combined Na and K was 744 ppb, while the highest result for combined Ca and Mg was 662 ppb.

All of the 27 B100 fuel samples from this test program were compliant with the ASTM D6751-18 limit of 5 ppm for Na + K and Ca + Mg respectively, and the results showed that levels were at less than 20% of the limit for two of the samples, while the rest were at less than 10% (and in most cases well below that) of the limit. A reduction of 80% in metal content for B20 and a reduction of 95% in metal content for B5 fuel blends would result in a maximum Na + K content of 149 ppb and 37 ppb respectively for the B100 fuel with the highest Na + K content. Ca + Mg would be 132 ppb and 33 ppb respectively.[Q]

[Q] This assumes no contribution from the diesel fuel used to formulate the blends.

Table 1-7: EPA 2017 Metals results for ICP-AES analysis of Na, K, Ca, and Mg performed by CDFA.

Sample ID	Area of US	Na (ppm)	K (ppm)	Ca (ppm)	Mg (ppm)	Na + K (ppm)	Ca + Mg (ppm)
25982	East	0.171	0.211	0.131	0.133	0.382	0.264
25988	East	0.084	0.660	(0.013)	0.018	0.744	[0.031]
25998	Midwest	0.241	(0.052)	(0.013)	0.005	[0.293]	[0.018]
26004	Midwest	(0.023)	(0.052)	(0.013)	(0.004)	(0.075)	(0.017)
26006	Midwest	0.081	(0.052)	(0.013)	(0.004)	[0.133]	(0.017)
26083	Midwest	0.181	(0.052)	0.026	(0.004)	[0.233]	[0.030]
26084	Midwest	0.188	(0.052)	0.025	(0.004)	[0.240]	[0.029]
26088	Midwest	0.111	(0.052)	(0.013)	(0.004)	[0.163]	(0.017)
26090	Midwest	0.135	(0.052)	(0.013)	(0.004)	[0.187]	(0.017)
26092	Midwest	0.201	(0.052)	(0.013)	(0.004)	[0.253]	(0.017)
26095	Midwest	0.378	(0.052)	(0.013)	(0.004)	[0.430]	(0.017)
26164	South	0.193	(0.052)	0.040	(0.004)	[0.245]	[0.044]
26165	South	0.044	(0.052)	(0.013)	(0.004)	[0.096]	(0.017)
26166	South	(0.023)	(0.052)	(0.013)	(0.004)	(0.075)	(0.017)
26217	West	0.108	(0.052)	(0.013)	(0.004)	[0.160]	(0.017)
26218	West	0.564	(0.052)	0.027	(0.004)	[0.616]	[0.031]
26219	West	0.278	(0.052)	0.143	0.031	[0.330]	0.174
26248	West	0.303	(0.052)	0.551	0.111	[0.355]	0.662
26250	South	0.031	0.186	(0.013)	(0.004)	0.217	(0.017)
26253	South	(0.023)	(0.052)	(0.013)	(0.004)	(0.075)	(0.017)
26254	South	(0.023)	(0.052)	(0.013)	(0.004)	(0.075)	(0.017)
26256	South	(0.023)	(0.052)	(0.013)	(0.004)	(0.075)	(0.017)
26283	South	0.331	(0.052)	(0.013)	(0.004)	[0.383]	(0.017)
26830	South	0.379	(0.052)	0.016	(0.004)	[0.431]	[0.020]
26833	East	0.290	(0.052)	(0.013)	(0.004)	[0.342]	(0.017)
27581	East	0.226	(0.052)	0.044	(0.004)	[0.278]	[0.048]
27955	West	0.270	(0.052)	(0.013)	(0.004)	[0.322]	(0.017)
Average		0.182	0.085	0.046	0.014	0.267	0.060

*Values in (parenthesis) are below the detection limit and are reported at the detection limit.
**Na + K and Ca + Mg values in [square brackets] include one element that is below the detection limit and is included in the calculation at the detection limit.

Table 1-8: EPA 2017 Metals results for ICP-AES analysis of Mo, P, B, Ba, Cu, Mn, Si, Ti, V, and Zn performed by CDFA.

Sample ID	Area of US	Mo (ppm)	P (ppm)	B (ppm)	Ba (ppm)	Cu (ppm)	Mn (ppm)	Si (ppm)	Ti (ppm)	V (ppm)	Zn (ppm)
25982	East	0.070	0.301	2.577	0.057	0.157	0.141	2.364	0.138	0.134	0.041
25988	East	0.012	0.305	1.220	0.006	0.032	0.014	2.896	0.004	0.010	(0.005)
25998	Midwest	(0.006)	0.207	0.126	0.002	0.006	0.002	0.927	(0.003)	0.003	(0.005)
26004	Midwest	(0.006)	0.073	0.154	(0.001)	(0.005)	(0.001)	0.034	(0.003)	(0.002)	(0.005)
26006	Midwest	0.008	0.138	0.081	(0.001)	(0.005)	(0.001)	0.100	(0.003)	(0.002)	(0.005)
26083	Midwest	(0.006)	0.188	0.062	0.003	0.022	0.004	0.241	(0.003)	(0.002)	(0.005)
26084	Midwest	(0.006)	0.182	0.053	0.002	0.023	0.004	0.233	(0.003)	(0.002)	(0.005)
26088	Midwest	(0.006)	0.735	0.029	(0.001)	(0.005)	(0.001)	0.020	(0.003)	(0.002)	(0.005)
26090	Midwest	0.008	0.226	0.017	(0.001)	0.019	(0.001)	0.172	(0.003)	(0.002)	(0.005)
26092	Midwest	0.010	0.165	0.038	(0.001)	0.046	(0.001)	1.168	(0.003)	0.004	(0.005)
26095	Midwest	(0.006)	0.100	0.029	(0.001)	0.054	(0.001)	1.043	(0.003)	0.004	(0.005)
26164	South	0.022	0.232	0.022	(0.001)	0.024	0.003	0.220	(0.003)	(0.002)	(0.005)
26165	South	(0.006)	0.949	(0.013)	(0.001)	(0.005)	(0.001)	0.046	(0.003)	(0.002)	(0.005)
26166	South	(0.006)	0.019	0.014	(0.001)	(0.005)	(0.001)	0.032	(0.003)	(0.002)	(0.005)
26217	West	0.011	0.101	0.024	(0.001)	0.035	(0.001)	0.350	(0.003)	0.021	(0.005)
26218	West	(0.006)	(0.001)	(0.013)	(0.001)	0.025	(0.001)	(0.017)	(0.003)	(0.002)	0.049
26219	West	(0.006)	0.302	0.014	(0.001)	0.035	(0.001)	0.104	(0.003)	(0.002)	(0.005)
26248	West	(0.006)	0.365	(0.013)	0.004	0.037	(0.001)	1.776	(0.003)	(0.002)	(0.005)
26250	South	(0.006)	(0.001)	(0.013)	(0.001)	(0.005)	(0.001)	0.130	(0.003)	(0.002)	(0.005)
26253	South	(0.006)	0.025	0.023	(0.001)	(0.005)	(0.001)	1.077	(0.003)	(0.002)	(0.005)
26254	South	(0.006)	0.021	0.027	(0.001)	(0.005)	(0.001)	1.239	(0.003)	(0.002)	(0.005)
26256	South	(0.006)	(0.001)	(0.013)	(0.001)	(0.005)	(0.001)	0.306	(0.003)	(0.002)	(0.005)
26283	South	(0.006)	0.133	(0.013)	(0.001)	0.193	(0.001)	0.592	(0.003)	0.020	(0.005)
26830	South	(0.006)	0.193	(0.013)	(0.001)	0.066	(0.001)	1.516	(0.003)	0.004	(0.005)
26833	East	0.012	0.237	(0.013)	(0.001)	0.035	0.007	0.397	(0.003)	0.013	(0.005)
27581	East	(0.006)	0.038	(0.013)	(0.001)	(0.005)	(0.001)	0.461	(0.003)	0.004	(0.005)
27955	West	(0.006)	(0.001)	(0.013)	(0.001)	(0.005)	(0.001)	0.115	(0.003)	(0.002)	(0.005)
Average		0.010	0.194	0.172	0.004	0.032	0.007	0.651	0.008	0.009	0.008

*Values in parenthesis are below the detection limit and are reported at the detection limit.

The California Air Resources Board (ARB) CDFA inspectors carried out a biodiesel sampling campaign throughout California during the spring and fall of 2019 collecting three hundred fifty-five (355) biodiesel and diesel fuel samples from both #2 diesel labeled pumps and biodiesel labeled pumps in the state of California.[75]

These samples were analyzed by the same lab as the 27 EPA samples mentioned above and afforded the same detection limits. The primary focus of analysis was to examine the average and observed range of concentration for Na, K, Ca, Mg and P of the biodiesel samples and the diesel samples.

Statistical analysis of the samples showed that the Na, K, Ca, Mg and P concentrations in all of the 355 collected fuel samples across California were significantly lower than the worst case expected concentrations for a B20 fuel blended from B100 blend stock that is at the ASTM

D6751-18 limit. Only three P samples, one Mg + Ca sample, and thirteen Na + K samples across the entire sample set exceeded worst case expected absolute concentrations for a B5 blended from B100 blend stock that is at the ASTM D6751-18 limit.

Na was the most abundant metal observed and was above the detection limit for 273 of 355 samples with sample 30077 exhibiting the highest result at 837 ppb. The rest of the metals were largely below detection limits. K was only above the detection limit for 14 of 355 samples with sample 15162 exhibiting the highest result at 172 ppb. Ca was above the detection limit for 24 of 355 samples with sample 30062 exhibiting the highest result at 168 ppb. Mg was only above the detection limit for 32 of 355 samples with sample 15162 exhibiting the highest result at 238 ppb. Sample 30077 exhibited the highest result for combined Na and K at 889 ppb, while sample 30062 exhibited the highest result for combined Ca and Mg at 353 ppb. P was above the detection limit for 92 of 355 samples, with sample 30077 exhibiting the highest result at 862 ppb. Tables containing the maximum and average concentrations with standard deviation can be found in the CARB comments to the ANPRM in the docket.[75]

A review of the NREL, EPA, and ARB data sets indicate that biodiesel fuel is compliant with the ASTM D6751-18 limits for Na, K, Ca, and Mg. While the test results indicate that there is an occasional B100 blend stock that is off specification with respect to the ASTM D6751-18 limits, and occasional biodiesel blends that are off specification to the pseudo limits, these occurances are the exception. The NREL 2016, EPA, and CARB data sets all use measurement methods that afford low levels of detection (sub-100 ppb), and these data sets further indicate that the Na, K, Ca, and Mg content of biodiesel blends is extremely low in general, on the order of less than 100 ppb. While these metals are present in biodiesel blends and testing has shown that exposure to metals can adversely affect emission control system performance, data suggest that the low levels measured in today's fuels are not enough to adversely affect system performance out to full regulatory useful life values we are finalizing in this action, provided that the engine manufacturer properly sizes the catalysts to account for the low-level exposure.

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Chapter 2 Compliance Provisions

2.1 Compression-Ignition Engine Dynamometer Test Procedures

Test procedures are a crucial aspect of the heavy-duty criteria pollutant program. This rulemaking is establishing several new test procedures for spark and compression ignition engine compliance. This chapter will describe the existing test procedures as well as the development process for the final test procedures. This includes the determination of emissions from both engines and hybrid powertrains as well as the development of new duty cycles.

2.1.1 **Current CI Test procedures**

Heavy-duty compression-ignition engines currently are certified for non-greenhouse gas (GHG) pollutants using the Heavy-Duty Diesel Engine Federal Test Procedure (HDDE FTP) and Supplemental Emission Test Ramped Modal Cycle (SET). For 2007 and later Heavy-Duty engines, 40 CFR part 86 – "Control of Emissions from New and In-Use Highway Vehicles and Engines" and 40 CFR part 1065 – "Engine Testing Procedures" detail the certification process. 40 CFR 86.007-11 defines the standard settings of Oxides of Nitrogen, Non-Methane Hydrocarbons, Carbon Monoxide, and Particulate Matter. The duty cycles are defined in Part 86. The HDDE FTP is defined in 40 CFR part 86, appendix I. The SET is defined in 40 CFR 86.1362(a). All emission measurements and calculations are defined in Part 1065, with exceptions as noted in 40 CFR 86.007-11. The data requirements are defined in 40 CFR 86.001-23 and 40 CFR 1065.695.

The measurement method for CO is described in 40 CFR 1065.250. For measurement of NMHC, refer to 40 CFR 1065.260. For measurement of NO_X, refer to 40 CFR 1065.270. For measurement of PM, refer to 40 CFR 1065.140, 1065.170, and 1065.290. Table 1 of 40 CFR 1065.205 provides performance specifications that we recommend analyzers meet. Note that 40 CFR 1065.307 provides linearity verifications that the system must meet. For the calculation method for brake specific mass emissions for CO, NMHC, NO_X and PM, refer to 40 CFR 1065.650.

2.1.1.1 HDDE FTP

The Heavy-Duty Diesel Engine Federal Test Procedure (HDDE FTP) is a transient test consisting of second-by-second sequences of engine speed and torque pairs with values given in normalized percent of maximum form. The cycle was computer generated from a dataset of 88 heavy-duty trucks in urban operation in New York and Los Angeles. This procedure is well-defined, mirrors in-use operating parameters, and continues to be appropriate for assessment of criteria pollutant emissions from heavy duty engines.

A complete HDDE FTP involves three test sequences. First, a 20-minute test is run over the duty-cycle with the engine at the same ambient temperature as the test cell (between 68°F and 86°F). The engine undergoes a 10-minute hot-soak following the cold-start. A 20-minute hot start test is run over the duty-cycle following the hot-soak. The HDDE FTP emission level for the engine is determined by weighting the cold start emissions by 1/7 (about 14 percent) and the hot-start emission results by 6/7 (about 86 percent).

2.1.1.2 SET

The Supplemental Emission Test Ramped Modal Cycle (SET) is a 13-mode, steady-state engine dynamometer test that replaced the steady-state modal run SET and is based on the European Stationary Cycle (ESC). The engine is tested on an engine dynamometer over a sequence of steady-state modes. Emissions are collected over both the steady-state and transition portions (20-second transitions) of the test and the results are integrated to produce a single emission result to show compliance with the standard.

The current weighting of modes within the SET for engines complying with the 2010 NO_X and Phase 1 GHG standards is given in Table 2-1. A, B, and C speeds are determined according to 40 CFR 1065.610.

Table 2-1: SET Mode Weighting Factors for the 2010 NO_X and Phase 1 GHG Standards

Speed, % Load	Weighting factors of SET (%)
Idle	15
A, 100	8
B, 50	10
B, 75	10
A, 50	5
A, 75	5
A, 25	5
B, 100	9
B, 25	10
C, 100	8
C, 25	5
C, 75	5
C, 50	5
Total	100
Idle Speed	15
Total A Speed	23
Total B Speed	39
Total C Speed	23

2.1.1.3 Powertrain

Powertrain test procedures were created under EPA's Heavy-duty Greenhouse Gas: Phase 2 (Phase 2) rulemaking for vehicle certification.[1,2] At the time of their development, no certification procedure existed for powertrain certification of heavy-duty hybrid vehicles to any engine standards. The powertrain test was updated for powertrain certification of the engine to engine standards for GHG pollutants in the Heavy-Duty Engine and Vehicle Test Procedures, and Other Technical Amendments rulemaking (hereinafter "HD Technical Amendments").[3] The powertrain certification test was finalized for certification to both the SET and FTP and is carried out by following 40 CFR 1037.550 as described in 40 CFR 1036.510 and 1036.512 and is applicable for powertrain systems with the hybrid function located in the P0, P1, P2, and P3 positions.

The development of these test procedures required the addition of a speed and road grade profile to the existing HDDE FTP, HDOC FTP, and newly finalized LLC duty cycles in the recently finalized 40 CFR part 1036, appendix II, and to the SET in 40 CFR 1036.510.[3] It also required the development of vehicle parameters to be used in place of those in 40 CFR 1037.550; namely vehicle test mass, vehicle frontal area, vehicle drag area, vehicle coefficient of rolling resistance, drive axle ratio, tire radius, vehicle curb mass, and linear equivalent mass of rotational moment of inertias. Determination of system and continuous rated power along with the maximum vehicle speed (C speed) is also required using 40 CFR 1036.520. The combination of the generic vehicle parameters, the engine duty-cycle vehicle speed profile, and road grade profile fully defines the system load and this is designed to match up the powertrain load with the HDDE vFTP, HDOC vFTP, vSET, and vLLC load for an equally powered engine.

The development of these test procedures was previously described in detail in the HD Technical Amendments.[3]

2.1.2 **Final updates to CI Test procedures**

2.1.2.1 HDDE FTP

We are finalizing no changes to the HDDE FTP weighting factors or the duty-cycle torque values from the duty-cycle that currently applies to criteria pollutant regulations in 40 CFR part 86 Appendix I (f)(2). We are finalizing a change to the engine speed values that does not influence the ultimate denormalized speed, as noted below and finalized in the HD Technical Amendments, apply to criteria pollutant certification as well. We started the migration of some heavy-duty highway engine standard setting part test procedures to 40 CFR part 1036 in the HD Technical Amendments. This included the migration of the HDDE FTP drive schedule to Appendix II (c) of part 1036 in order to add vehicle speed and road grade to the duty-cycle to facilitate powertrain testing of hybrids for compliance with the Phase 2 GHG standards.

The change that was made for GHG and is being finalized to apply to criteria pollutant certification as well. We took the normalized vehicle speeds over the HDDE FTP duty-cycle and multiplied them by 100/112 to eliminate the need to divide by 112 in the diesel engine denormalization equation in 40 CFR 86.1333(a)(1)(i). This eliminated the need for inclusion of a denormalization equation in the standard setting part and allows commonization of (between compression and spark ignition engines) the use of the denormalization equation in 40 CFR 1065.610(c)(1) (equation 1065.610-3), with no effect on stringency.

2.1.2.2 SET

The SET weighting currently used for certification of heavy-duty highway compression ignition engines for criteria pollutants and Phase 1 GHG has a relatively large weighting in C speed. The C speed is typically in the range of 1800 rpm for current heavy heavy-duty engine designs. However, it is becoming much less common for engines to operate at such a high speed in real-world driving conditions, especially during cruise vehicle speeds in the 55 to 65 mph vehicle speed range. This trend has been corroborated by engine manufacturers' in-use data that has been submitted to the agencies in comments to previous rules and presented at technical conferences.[4] Thus, although the current criteria pollutant and HD Phase 1 GHG SET represents highway operation better than the FTP cycle, improvements have been made via the HD Phase 2 GHG program by adjusting its weighting factors to better reflect modern trends in in-use engine operation. The most recent trends for compliance with the Phase 2 GHG standards indicate that manufacturers are configuring drivetrains to operate engines at speeds down to a range of 1050-1200 rpm at a vehicle speed of 65mph.

To address this trend toward in-use engine down-speeding, the agency is finalizing the application of the refined SET weighting factors and resulting SET developed for the HD Phase 2 GHG standards to the criteria pollutant standards. The Phase 2 GHG SET mode weightings move most of the C weighting to the A speed, as shown in Table 2-2. To better align with in-use data, these changes also include a reduction of the idle speed weighting factor. This will apply the Phase 2 mode weightings to both criteria pollutants and the Phase 2 CO_2 emission and fuel consumption standards starting in model year 2027.

Table 2-2: New SET Mode Weighting Factors in Phase 2

Speed/% Load	**Weighting Factor in Phase 2 (%)**
Idle	12
A, 100	9
B, 50	10
B, 75	10
A, 50	12
A, 75	12
A, 25	12
B, 100	9
B, 25	9
C, 100	2
C, 25	1
C, 75	1
C, 50	1
Total	100
Idle Speed	12
Total A Speed	45
Total B Speed	38
Total C Speed	5

The Phase 2 SET mode weighting moves most of the C speed weighting to A speed and reduces the weighting factor on idle speed. These values are based on data from vehicle manufacturers that have been claimed as confidential business information. These revised SET weighting factors better reflect the lower engine speed operation of modern engines, which frequently occurs at tractor cruise speeds.

To evaluate how current engines perform on this cycle, we tested a 2018 Detroit DD15 and a 2018 Cummins B6.7. For both engines, there was no significant difference for any of the measured criteria pollutants between the two cycles. These results are summarized in Chapter 3.1

of the RIA. To assess the effect of stringency between the two SET cycles, the CARB Stage 3 demonstration engine was also run on both versions of the SET. The results from these tests can be found in Chapter 3.1 of the RIA.

2.1.2.3 LLC

Current certifications cycles (FTP and SET) and not-to-exceed (NTE) compliance requirements do not account for emissions over sustained low load operation. This is either because the idle time in the duty-cycle is too short, or, in-use, the operation is excluded from compliance requirements.

We are finalizing a new low load certification cycle to address deficiencies in our current certification duty-cycles and NTE field testing program with respect to emission control at low load.

The National Renewable Energy Lab (NREL) and Southwest Research Institute (SwRI) under contract with the California Air Resources Board (CARB) developed a suite of candidate low load cycles (LLC) from urban tractor and vocational vehicle real-world activity data. The goal of the cycle development was to develop a duty-cycle that was representative of real-world urban tractor and vocational vehicle operations that are characterized by low engine loads, have average power and duration adequate for demonstrating that hardware and controls needed to deal with low load challenges are present and functional, and set an emission standard that balances the need for NO_X emission reductions and any associated GHG emission impacts.[5,6,7,8]

NREL combined their Fleet DNA and CARB's heavy-duty diesel vehicle activity datasets, incorporating a total of 751 unique vehicles across the United States, to develop the LLC. The combined dataset included vehicles form 25 distinct locations, 26 combined vocational designations, and 55 unique fleets incorporating both urban tractor and vocational applications. A breakdown of the applications that were included can be found in Table 2-3.

Table 2-3: Breakdown of vehicles from combined NREL Fleet DNA and CARB datasets.

Vehicle Application	Number of Vehicles
Parcel delivery	100
Refuse pickup	90
Line Haul	84
Beverage delivery	65
Mass transit	61
Food delivery	60
Drayage	41
Utility	32
Linen delivery	30
Transfer truck	29
Tanker	25
Telecom	24
Freight	22
Public work	13
School bus	11
Agricultural	10
Snow plow	9
Warehouse delivery	9
Construction	8
Dump truck	7
Refrigerated truck	6
Local delivery	5
Towing	4
Concrete	3
Dry van	3
Delivery (other)	1

NREL initially developed a list of drive cycle metrics, including engine load specific calculations to describe engine/vehicle operation. Vehicle operation was then broken down into microtrips for operation over a given shift day, where a single microtrip was defined as the duration over which a vehicle speed increases from 0 mph to the time where the vehicle stops, including operation until the vehicle speed starts to increase above 0 mph again. 10 microtrips were averaged using moving average windows based on a sensitivity analysis of moving average windows of 5, 10, and 15 microtrips. The total number of microtrips from the combined data sets was approximately 1.25 million and resulted in a trimodal distribution with two large primary peaks and a tertiary lower load peak as shown in Figure 2-2.

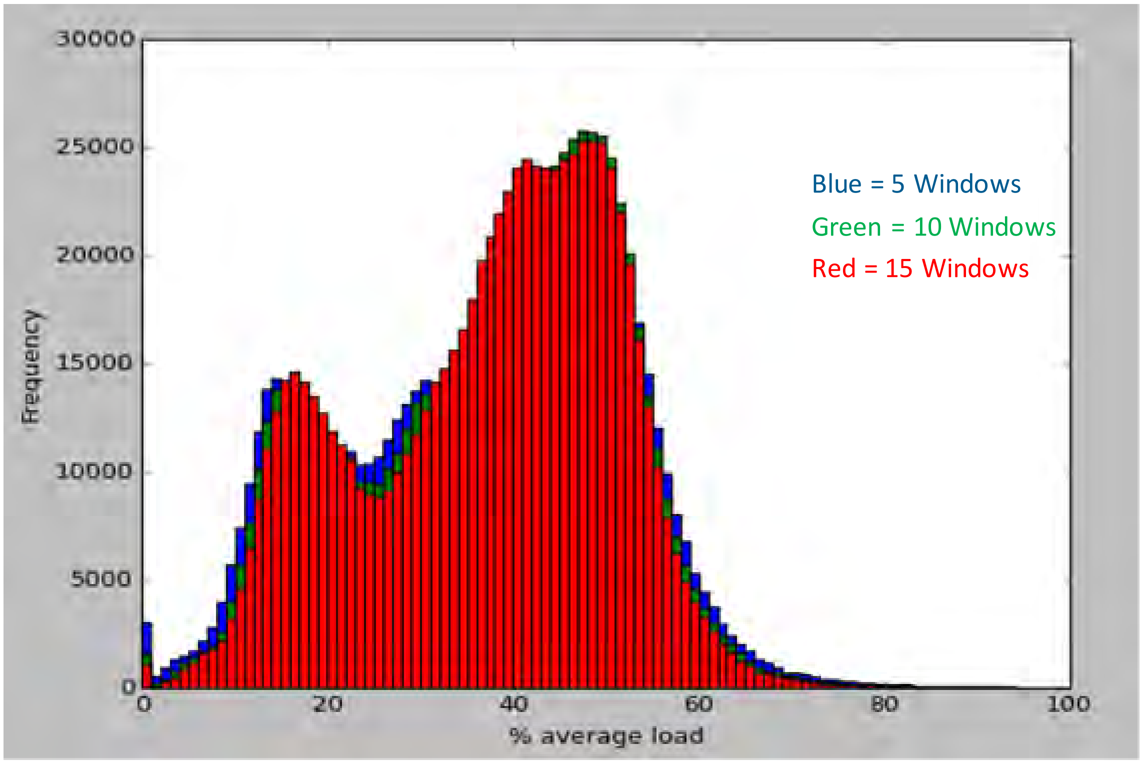


Figure 2-1

Figure 2-2: Window size comparison and load distribution profile.

The microtrips were then equated to operational profiles and profiles that contained average loads of 20% or less were considered further for construction of the LLC. These remaining profiles were then subjected to cluster analysis to identify unique groups of operation and to assess for outliers. K-means clustering was chosen to be applied to the dataset due to its computational efficiency and ability to identify and remove outliers during pre-processing. Elbow analysis was then applied to determine optimal cluster number. The resulting optimal cluster number was three.

Representative profiles from each cluster were then selected for Greenhouse Gas Emissions Model (GEM) simulation. Results for each cluster were ranked based on their distance to cluster center to identify the most representative profiles. Profiles were examined for behavior and final suitability for testing, starting with profiles closest to cluster center. Five primary modes of operation were generally observed in the low load profiles: sustained low load, long idle, motoring/short idle cooling (high to low load), post-cooling breakthrough (low to high load), and mid-speed cruise motoring. Profiles found to have outlying behavior were removed and not used for GEM modeling. These outliers were found to have one or more of the following: prolonged periods of idle, long key off periods, and missing data. The load data broadcast by engines is not accurate enough to allow translation of vehicle-based to engine-based profiles to create the engine duty-cycle, so the Phase 2 GEM simulation model was used to develop the normalized engine duty-cycle. A representative summary of ten GEM generated profiles is shown in Table 2-4.

Initial LLC candidate duty-cycles were constructed using DRIVE to include at least one example of each of the five primary modes of operation, incorporating five of the GEM generated duty-cycles. It should be noted that these candidate cycles did not always include the entire GEM generated profile if the candidate cycle could be completed in a shorter amount of time by removing portions of the profile that did not adversely affect the target modes of operation. Figure 2-3 gives an example of a candidate cycle incorporating representative profiles v9892 c0, v11660 c5, v073 c1, v9892 c1, and v11806 c5.

CARB narrowed the range of candidate LLC duty-cycles to the three that best represented the target modes of low load. The three were LLC Candidates 7, 8, and 10. Candidate cycle 7 is 90 minutes in duration, has 30 minutes of sustained low load operation, and retains the v073 c1 mid-speed cruise/motoring segment. Candidate cycle 8 is 81 minutes in duration, has 30 minutes of sustained low load operation, and has a shorter v073 c1 mid-speed cruise/motoring segment to assess breakthrough only. Candidate cycle 10 is 70 minutes in duration, has 20 minutes of sustained low load operation, and has a shorter v073 c1 mid-speed cruise/motoring segment to assess breakthrough only. Emission results for a representative engine compliant to EPA 2010 NO_X standard for the three-candidate duty-cycles can be found in Table 2-5.

CARB selected candidate cycle 7, with an option to add auxiliary load (1.5, 2.5, and 3.5 kW for LHD, MHD, and HHD engines respectfully), as the LLC that it moved forward with in its Omnibus heavy-duty low NO_X program. This cycle was chosen by CARB for inclusion of the highest percentage of non-idle operation when compared to the other two candidate cycles. CARB's decision to move forward with the option to add auxiliary load was based on making the duty-cycle more realistic with respect to real world operation as it affords more effective function of technologies such as cylinder deactivation (CDA) during idle.

We agree with CARB's assessment of the candidate low load cycles and their adoption of cycle 7. Thus we are finalizing to adopt LLC 7, however we are finalizing to require the use of auxiliary load, to bolster the current FTP and SET duty-cycles and to better align laboratory duty-cycles with the finalized changes we are making to off-cycle testing and compliance requirements, which afford better low load NO_X reduction. We are requiring the use of auxiliary load because this results in more representative testing with respect to real world operation. This duty-cycle also contains vehicle speed and road grade profiles to facilitate powertrain certification of hybrid powertrains to the engine standard. These profiles were developed in the same manner as the HDDE FTP and HDOC FTP as well as the SET as discussed in the HD Technical Amendments.[3] The LLC can be found in 40 CFR part 1036, appendix B(d).

Table 2-4: Representative Summary of GEM Generated Profiles for the Engine Duty-cycle

Profile	Vehicle	Cluster	Length	Avg % Speed	Avg % Torque	Repeats in SwRI Test Runs	Class	Chassis	Engine	Trans	Gears	Vocation
1	v9892	0	800	26.9	6.9	4	8	4x2	Volvo D13	AMT	12	Food Service
2	v11660	0	1295	21.4	6.6	3	8	6x4	Mack MP8-415C	MT	13	Drayage
3	v075	0	1130	26.3	7.4	3	8	6x4	Mack MP8-415C	AMT	10	Drayage
4	v11815	1	1949	11.5	8.8	3	8	6x4	Cummins ISX 15	MT	13	Transfer Truck
5	v11646	1	904	15.9	10.7	4	4	4x2	Cummins ISB 6.7	AT	6	Parcel Delivery
6	v073	1	1410	33.8	18.1	3	8	6x4	Mack MP8-415C	AMT	10	Drayage
7	v9892	1	1616	27	10.6	3	8	4x2	Volvo D13	AMT	12	Food Service
8	v11660	5	615	16.2	3.5	4	8	6x4	Mack MP8-415C	MT	13	Drayage
9	v11806	5	1810	7.5	6.8	3	8	6x4	Cummins ISX 12	AMT	10	Transfer Truck
10	v11817	5	739	15.3	7.7	4	8	6x4	Cummins ISM 11	AMT	10	Transfer Truck

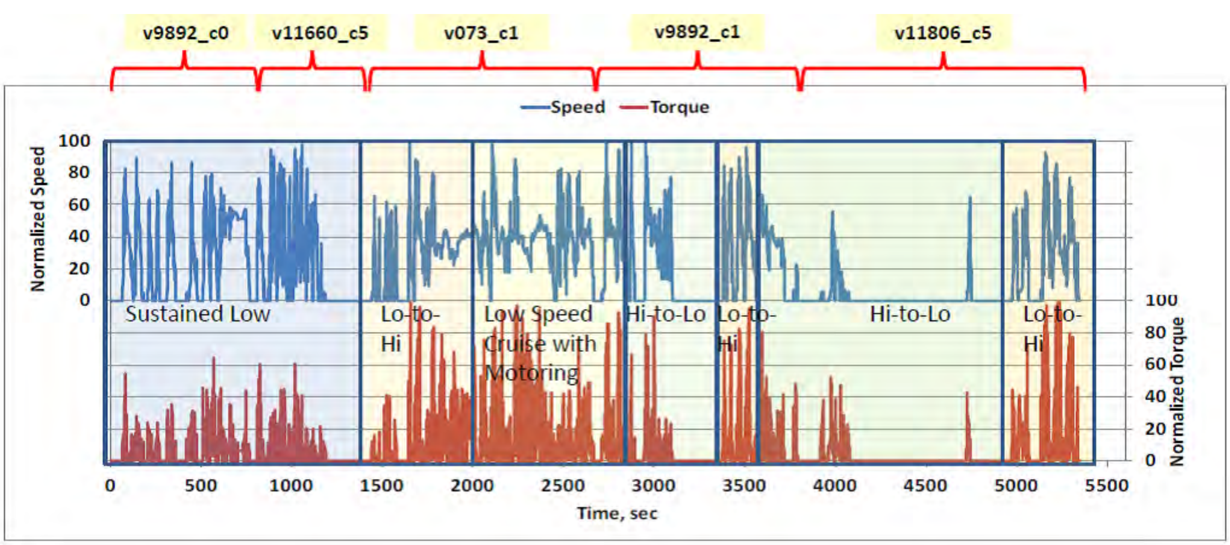


Figure 2-3: Example of a LLC 7 Candidate Cycle

Table 2-5: NO_X Emission Levels for a 2010 Compliant Engine on Three Candidate LLCs

Candidate #	Cycle Duration	NO_X Conversion Efficiency (%)	Engine Out NO_X (g/hp-hr)	Tailpipe NO_X (g/hp-hr)
7	90	74	3.2	0.8
8	81	77	2.9	0.7
10	70	69	3.2	1.0

2.1.2.4 Powertrain

We are finalizing the use of powertrain testing as an option for certification of hybrid powertrains to criteria pollutant standards. It is envisioned that this option will capture CO_2 and NO_X co-optimization benefits for hybrid integration; including engine start/stop, electric motor assist, electric vehicle mode, and brake energy recovery. The powertrain test procedures to facilitate this type of certification testing have already been developed during the HD Technical Amendments and are directly applicable to criteria pollutant testing. Duty-cycles that incorporate vehicle speed and road grade for the HDDE FTP, HDOC FTP, and the SET were finalized in the HD Technical Amendments where development of these cycles was discussed.[3] The finalized addition of the low load cycle in Chapter 2.1.2.3 also included a discussion of the vehicle speed and road grade profile development for powertrain testing.

2.2 Manufacturer-Run Off-Cycle Field Testing Program for Compression-Ignition Engines

The manufacturer run field testing program is crucial to ensuring compliance with the heavy-duty criteria pollutant program. This rulemaking establishes a new test procedure for evaluating off-cycle compression ignition engine compliance.[A] This chapter describes the existing test procedure as well as the development process for the final test procedure.

2.2.1 Current Field Testing Program and Off-Cycle Standards

EPA's current regulatory program for on-highway heavy-duty engines has evolved over the past four decades from relatively simple standards and test procedures appropriate for the mechanically controlled engines of the 1970s and 1980s, to a multi-faceted program designed to reduce emissions from modern computer-controlled engines. However, throughout the years, the compliance paradigm has focused heavily on the pre-production certification process, during which the manufacturers demonstrate that their engines will meet the applicable standards and related requirements.

Until fairly recently, compliance was determined only by testing engines in a laboratory on an engine dynamometer. Prior to model year 2004, engines were evaluated over a single transient

[A] Duty-cycle test procedures measure emissions while the engine is operating over precisely defined duty cycles in an emissions testing laboratory and provide very repeatable emission measurements. Off-cycle test procedures measure emissions while the engine is not operating on a specified duty cycle; this testing can be conducted while the engine is being driven on the road (e.g., on a package delivery route), or in an emission testing laboratory. When off-cycle testing is conducted on the road it is referred to as "field testing"; "In-use" is used to denote that testing is done on an engine that has entered into commerce. Both duty-cycle and off-cycle testing are conducted pre-production (e.g., for certification) or post-production (i.e., in-use) to verify that the engine meets applicable duty-cycle or off-cycle emission standards throughout useful life (see Section III for more discussion).

test cycle commonly referred to as the Heavy-Duty Diesel Engine FTP (HDDE FTP) cycle. Beginning with the 2004 standards, EPA added the engine dynamometer-based Supplemental Emission Test (SET)[B], and Not-to-Exceed (NTE) emission limits that are evaluated on heavy-duty highway engines while operating over the road.

Heavy-duty diesel engines are currently subject to Not-To-Exceed (NTE) standards that are not limited to specific test cycles, which means they can be evaluated in the field. Data from field testing are collected by manufacturers as described in section 2.2.1. The data is then analyzed pursuant to 40 CFR 86.1370 and 40 CFR 86.1912 to generate a set of engine-specific NTE events, which are intervals of at least 30-seconds when engine speeds and loads remain in the control area. The express purpose of the NTE test procedure is to apply the emission standard to engine operation conditions that could reasonably be expected to be seen by that engine in normal vehicle operation and use, including a wide range of real ambient conditions.

The NTE zone defines the range of engine operation where the engine must comply. The NTE zone is based on engine speed and load and includes some carve outs that include low load operation (excludes load points less than 30% of T_{max} and P_{max}, and less than 15% of the European Stationary Cycle speed), as described in 40 CFR 86.1370. In addition, there are carve outs for altitude (> 5500 ft), maximum ambient temperature (100 °F at sea level, 86 °F at 5500 ft), aftertreatment temperature for NO_X aftertreatment and oxidizing catalysts (carves out operation at temperatures < 250°C), and provides a cold temperature operating exclusion for EGR equipped engines (calculation based on intake manifold temperature, engine coolant temperature, and intake manifold pressure).[C]

Heavy-duty engines are required to comply with not-to-exceed (NTE) emission limits during real world operation. Engine manufacturers must acquire and submit data through the manufacturer run in-use testing program. These off-cycle NTE emission limits are 1.5 (1.25 for CO) times the laboratory certification standard or family emission limit (FEL) for NO_X, NMHC, PM and CO and can be found in 40 CFR 86.007-11. A measurement allowance value is added on to the standard to account for measurement inaccuracies that are associated with field measurement over short time periods and can be found in 40 CFR 86.1912. The engine standards and measurement allowances are in Table 2-6.

Table 2-6: Engine Standards and Field Testing Measurement Allowance

	NO_X (g/hp-hr)	**PM (g/hp-hr)**	**CO (g/hp-hr)**	**NMHC (g/hp-hr)**
Engine Standards	0.20	0.01	15.5	0.14
NTE Standards	0.30	0.015	19.4	0.21
Measurement Allowance	0.15	0.006	0.25	0.01

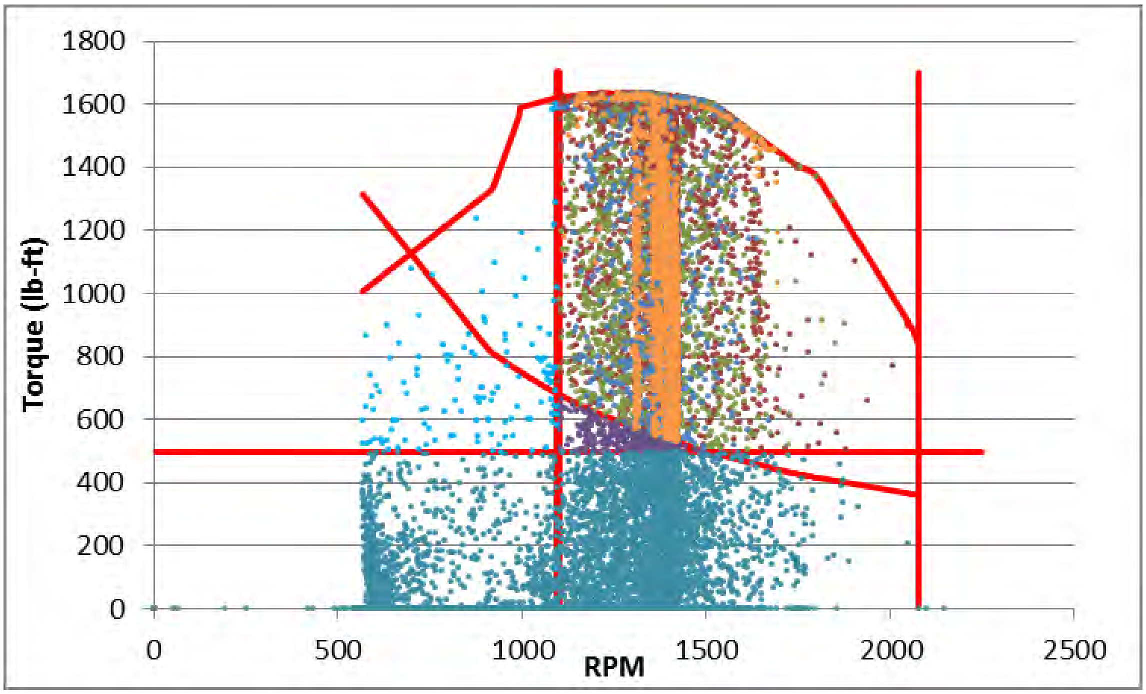
A valid NTE event is described as an event that is 30 seconds or more in duration under engine, aftertreatment, and ambient conditions that are within the NTE zone (i.e., do not occur

[B] The SET was later modified to run as a single continuous test (similar to how a transient cycle is run) and renamed the Supplemental Emission Test Ramped-Modal Cycle.

[C] For more on our NTE provisions, see 40 CFR 86.1370.

during the aforementioned exclusion conditions), see 40 CFR 86.1370 and 40 CFR 86.1912. The engine must meet a vehicle pass ratio of 90% of valid NTE events (i.e., 90% of the valid NTE events must comply with the off-cycle standard (0.45 g/hp-hr for NO_X) for the engine to be considered compliant) as described in 40 CFR 86.1912.

We have concerns with the current NTE regulations that compliance can be achieved over the entire operating regime of the engine due to low temperature aftertreatment exclusions, and the narrow engine operation that has to be met for at least 30 consecutive seconds, as shown in Figure 2-4. Removal of these exclusions will require the engine manufacturer to act to maintain aftertreatment temperature at low load modes of operation, and this in turn will lead to better real world performance with respect to emission compliance.[9]



Points excluded by reason:

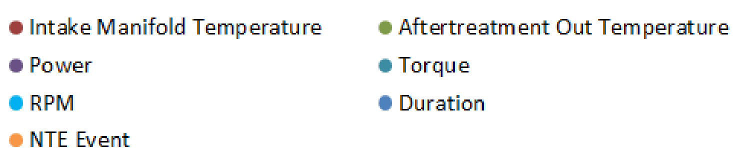


Figure 2-4: Sample of valid and invalid NTE events, separated by exclusion zones.

Data submitted under the current regulatory program has been acquired over more than 120,000 miles of vehicle testing. The data was generated by more than 540 unique vehicles and submitted by 14 different engine manufacturers. The percentage of test results meeting or exceeding the 0.9 pass ratio threshold since 2005 is presented in Table 2-7. A typical data set representing one test for an engine will contain twenty to forty valid NTE events, but under some routes no valid NTE events are measured.

Table 2-7: Percentage of Tests Meeting or Exceeding 0.90 Pass Ratio Threshold

Constituent	Percentage of Tests Meeting or Exceeding the 0.9 Pass Ratio
NO_X	96.0
CO	99.6
PM	99.4

The average pass ratio for each of the three constituents, for all data submitted since 2005, is presented in Table 2-8.

Table 2-8: Average Pass Ratios for Data Submitted Since 2005

Constituent	Overall Constituent Pass Ratio
NO_X	0.980
CO	0.996
PM	0.997

NTE standards have been successful in broadening the types of operation for which manufacturers design their emission controls to remain effective. Our analysis of field-testing data indicates that EPA's existing NTE limits did not apply to 95% of the elapsed operating time in these studies. Furthermore, we found that emissions are high during many of the excluded periods of operation, such as when the aftertreatment temperature drops below the catalyst light-off temperature. For example, 96 percent of tests from 2014, 2015, and 2016 field testing orders passed with NO_X emissions for valid NTE events well below the 0.3 g/hp-hr NTE standard. When we used the same data to calculate NO_X emissions over all operation measured, not just for valid NTE events, the NO_X emissions were more than double (0.5 g/hp-hr).[10] The results were higher when we analyzed the data to only consider NO_X emissions that occur during low load events. These results suggest there may be great potential to improve real world performance by considering more of the engine operation when we evaluate off-cycle compliance. The average value of idle NO_X for the 2014, 2015 and 2016 Test Orders is approximately 16 g/hr.

2.2.2 <u>Information evaluated for final updates</u>

2.2.2.1 CE-CERT Program description

The CE-CERT study involves instrumenting about 100 trucks and collecting real world second by second activity data over the course of one month from on-board GPS units and ECU scan tools. The instrumented vehicles are equipped with MY 2010 and later heavy-duty diesel engines utilizing SCR. Over 170 parameters are recorded including GPS based location, speed, elevation, ECU based vehicle speed, ECU based engine parameters (RPM, MAP, MAF, load), and aftertreatment variables (temperature, NO_X ppm). The vehicles cover 19 different groups based on vocational use, GVWR, and geographic operation.

Key findings of this study: Average speed varies from 41 mph for interstate line haul trucks to 9 mph for drayage trucks. Intrastate line haul trucks average 32 mph because they spend less time on freeways and double the time idling, compared to interstate line haul trucks. Most vocational vehicles spent about 33% of time operating at idle, irrespective of the time they spent on the freeway. SCR temperature for line haul operation has a bi-modal distribution with peaks at 100°C and 260-280°C. Drayage truck operation in Northern California exhibits a peak SCR operating temperature of around 110°C. Overall, the vehicles in this study spent 42-91% of their operating time with SCR temperatures below 250°C. Based on a generic SCR emission control efficacy versus SCR temperature curve, the average engine-out NO_X reduction could be 16% and 69% for agricultural trucks and refuse trucks, respectively.

In addition to giving a picture of heavy-duty vehicle activity, the study also provided data on NO_X sensors "on-time". It is widely known that NO_X sensors are not turned on when the humidity level of the exhaust is high enough to produced condensed water, but there is little data on how much time the sensors are off during heavy-duty vehicle operation. This information is useful if NO_X sensors are to be used as a compliance tool for an off-cycle standard. The sensors were not operational, or the tailpipe NO_X sensors were not reporting valid concentrations for 40% of the operating time based on the data collected. Figure 2-5 shows the time it takes until the sensors start reporting data from a cold start. The majority of the engine out sensors report data within the first 10 minutes after a cold start. The tailpipe sensors report data within the first 30 minutes, much later than the engine out sensors. This is understandable as the catalysts act as a heat sink, absorbing heat from the exhaust during warm-up and preventing this thermal energy from making it to the tailpipe sensor.

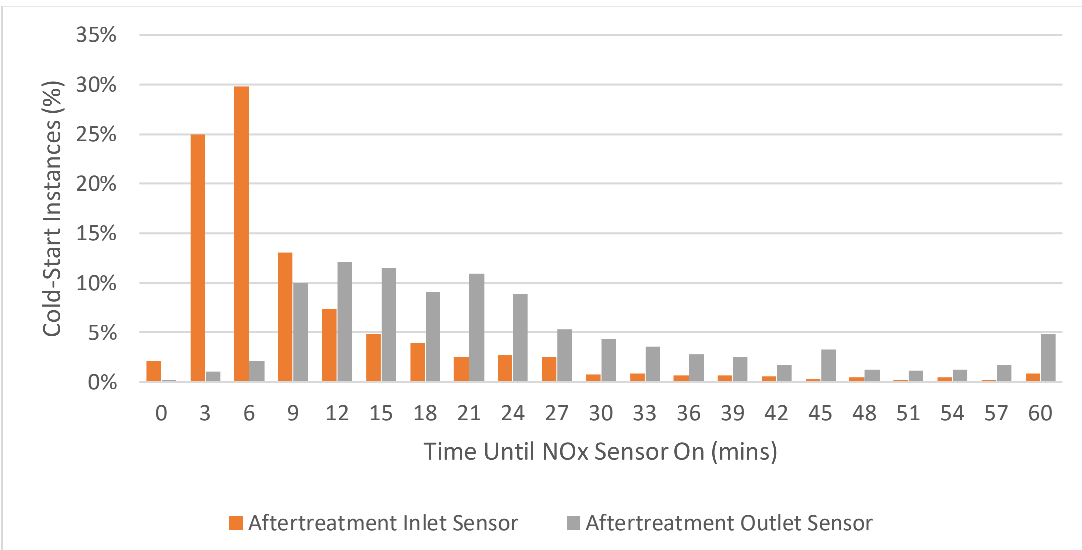


Figure 2-5: NO_X sensor time to on after engine cold-start

2.2.2.2 Summary of HDIUT Data

To evaluate the efficacy of current technology NO_X emissions controls, EPA analyzed the data from engines selected for testing in calendar years 2010 through 2016. This dataset covers 44 engine families, model years 2010 to 2015, from 11 manufacturers. The dataset includes about 8 million seconds of quality-assured second-by-second data collected during 68,000 miles of driving. The operational conditions include a wide range of driving speeds, transient and steady-state conditions, engine loads, and exhaust temperature conditions that have implications for emissions control efficacy, particularly for NO_X.[11] For the HHD class, out of a total 159 vehicles, 109 were line-haul, 46 were delivery, and the remaining were marked as “Other” in the metadata. Table 2-9 illustrates the distribution of family emission limit (FEL) values for the 291 vehicles tested. An FEL is a manufacturer-specified value that represents the maximum emission rate from the engines in that group during certification testing.

Table 2-9: Number of Diesel Vehicles with MY 2010+ Engines by NO_X FEL Group from the heavy-duty off-cycle field testing Program

Regulatory Class	NO_X FEL Group			Total
	0.20	0.35	0.50	
HHD	49	0	15	64
MHD	26	23	9	58
LHD	93	31	35	159
Urban Bus	0	10	0	10
Total	168	64	59	291

The following sections describe three analyses that use the HDIUT data to investigate the relationship between aftertreatment temperatures and NO_X emissions, and identify operations where emissions are elevated.

2.2.2.3 HDIUT Data by MOVES OpMode

For the first analysis, the data were grouped by operating mode (OpMode) used by EPA’s MOtor Vehicle Emission Simulator (MOVES) emissions inventory model. MOVES OpModes are defined in terms of power output using a scaled tractive power (STP_t) parameter, shown in Equation 2-1, and vehicle speed.

Equation 2-1

$$\boldsymbol{STP_t = \frac{Av_t + Bv_t^2 + Cv_t^3 + M \cdot v_t(a_t + g \cdot sin\theta_t)}{f_{scale}}}$$

Where:
STP_t = the scaled tractive power at time t [scaled kW or skW]
A = the rolling resistance coefficient [kW·sec/m],
B = the rotational resistance coefficient [kW·sec²/m²],
C = the aerodynamic drag coefficient [kW·sec³/m³],
m = mass of individual test vehicle [metric ton],
f_{scale} = fixed mass factor (LHD = 5, MHD = 7, HHD = 10),
v_t = instantaneous vehicle velocity at time t [m/s],
a_t = instantaneous vehicle acceleration [m/s²]
g = the acceleration due to gravity [9.8 m/s²]
$\sin \theta_t$ = the (fractional) road grade at time t

OpMode 0 is reserved for deceleration and braking events. OpMode 1 represents idle, defined as vehicle speeds less than 1.0 mph. The remaining OpModes are defined by STP_t ranges and vehicle speed (v_t) ranges of 1 to 25 mph, 25 to 50 mph, and greater than 50 mph. Figure 2-6 is a graphical representation of the MOVES OpModes, showing their relationship to STP_t and v_t.

See the MOVES Technical Report on heavy-duty emission rates for more information about OpModes.[12]

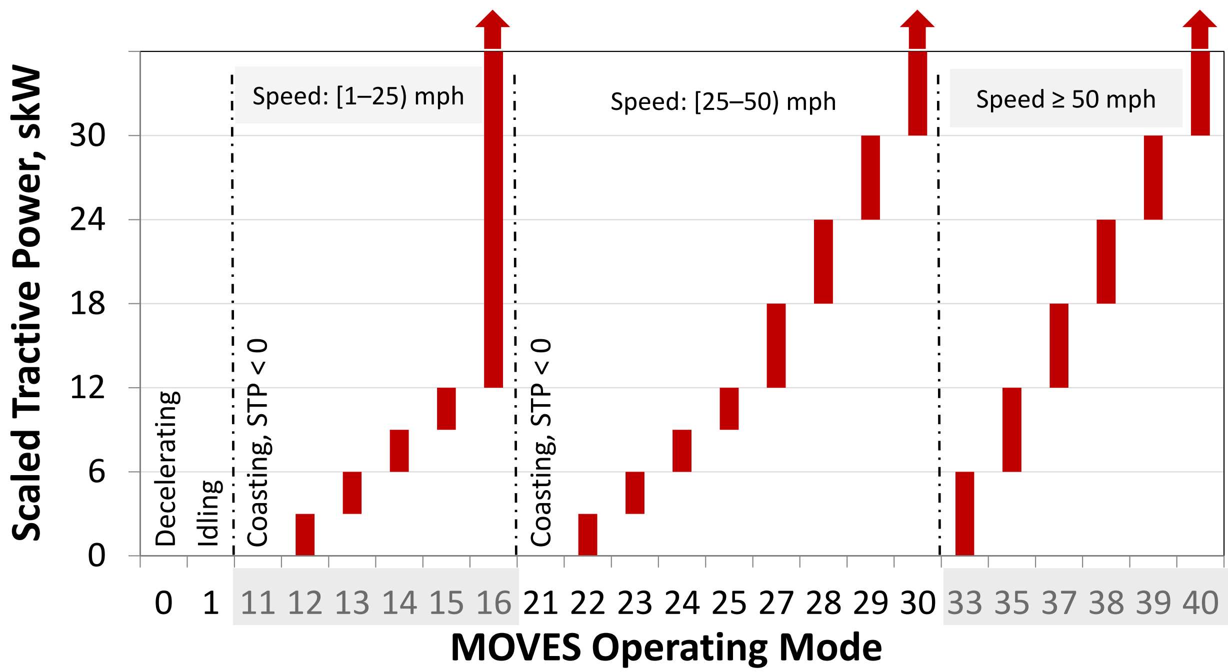


Figure 2-6: MOVES Operating Modes (OpModes) by scaled tractive power and vehicle speed

Figure 2-7 shows the NO_X emission rates in g/s for 65 vehicles from five manufacturers (by color) measured during off-cycle field testing. The engines are categorized into eight HHD engine families (E-1 through E-8) by shape with a NO_X FEL certification level of 0.20 g/bhp-hr. Real-world operation is known to produce large variability in emission rates, compared to certification testing performed in a lab. The graph shows that there is significant inter-engine family and intra-engine family variability in these field tests. For example, rates for E-1 are consistently lower than E-8. There is also variability within an engine family as represented by errors bars for each point. The spread is larger for engine families with higher emission rates, for example E-8. Similar trends are seen in MHD and LHD engine families.

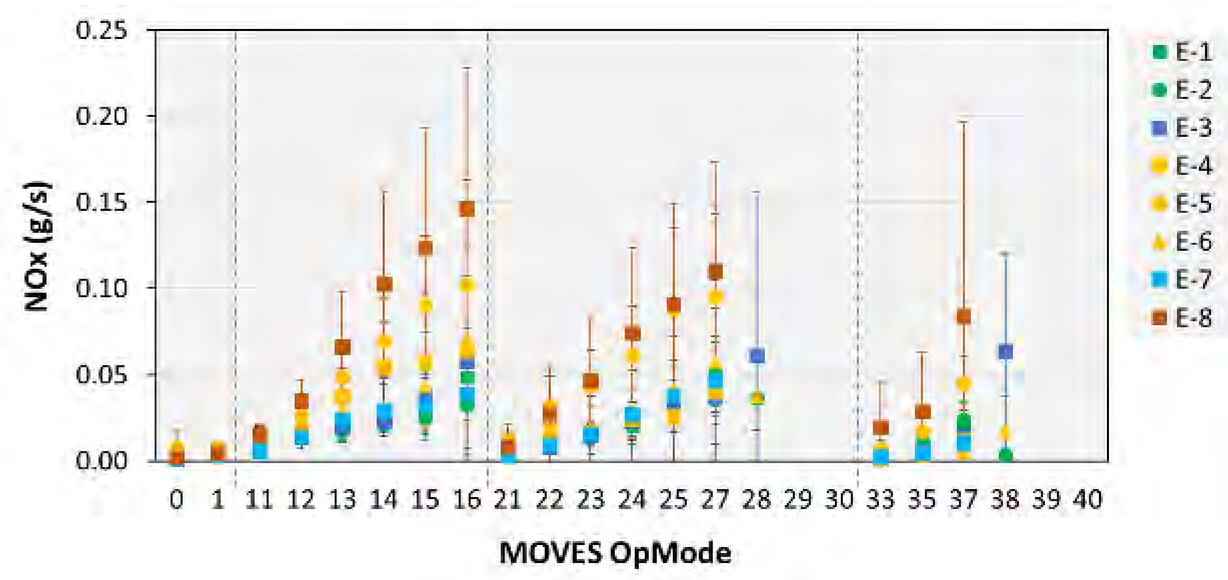


Figure 2-7: MOVES OpMode Emission Rates from HHD Engine Broken Down by Engine Family

Figure 2-8, Figure 2-9, Figure 2-10 show the OpMode-based NO_X emission rates for vehicles with NO_X FELs of 0.20 g/bhp-hr or better by aftertreatment temperature. The figures do not include all tested vehicles, as some tests did not report aftertreatment temperature. Figure 2-8 is based on data from 81 vehicles with HHD engines, Figure 2-9 is from 20 vehicles with MHD engines, and Figure 2-10 is from 42 vehicles with LHD engines. For all engines and most OpModes, the NO_X emission rates when a vehicle is operating with an aftertreatment temperature below 250°C are more than double compared to operation with an aftertreatment temperature above 250°C. The figures also show that high NO_X emissions occur across all OpModes and are not limited to low-speed or idle operation.

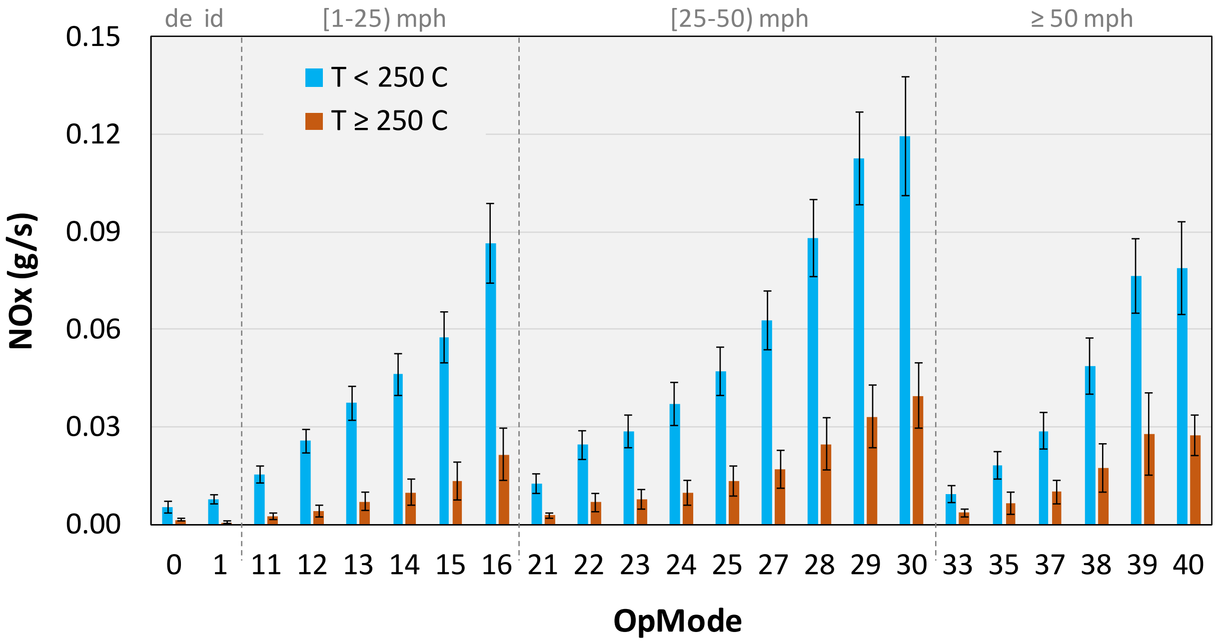


Figure 2-8: NOx Emission Rates from 81 Vehicles with 0.20 g/bhp-hr FEL HHD Engines by MOVES OpMode and Aftertreatment Temperature

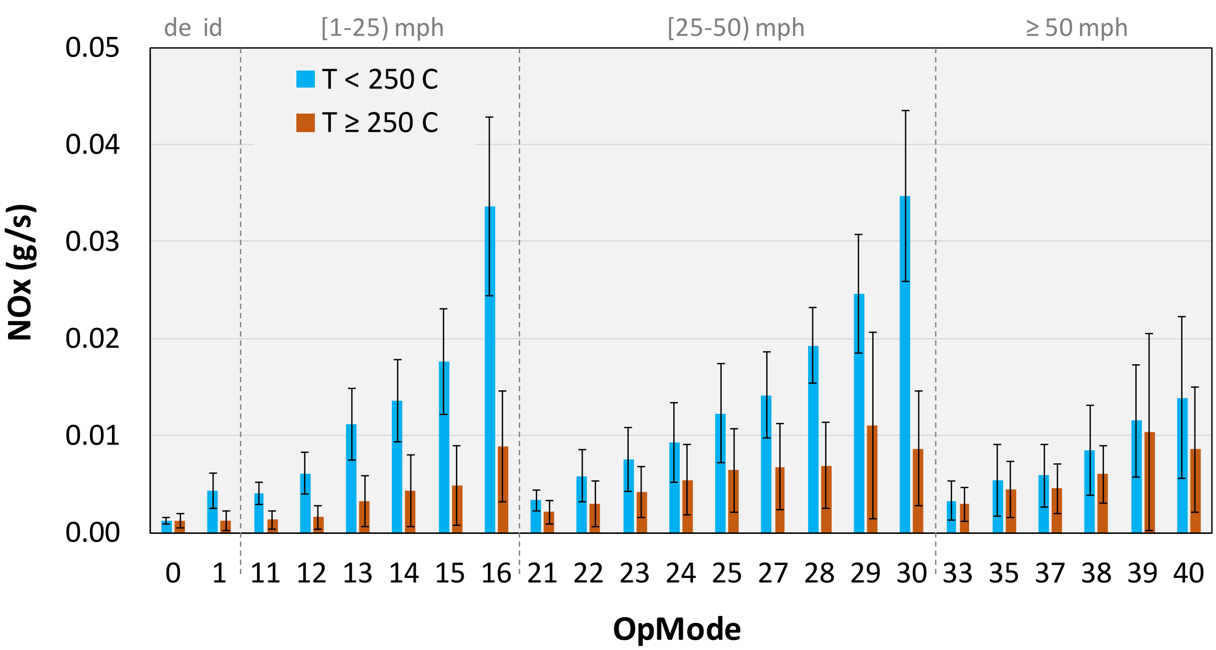


Figure 2-9: NOx Emission Rates from 20 Vehicles with 0.20 g/bhp-hr FEL MHD Engines by MOVES OpMode and Aftertreatment Temperature

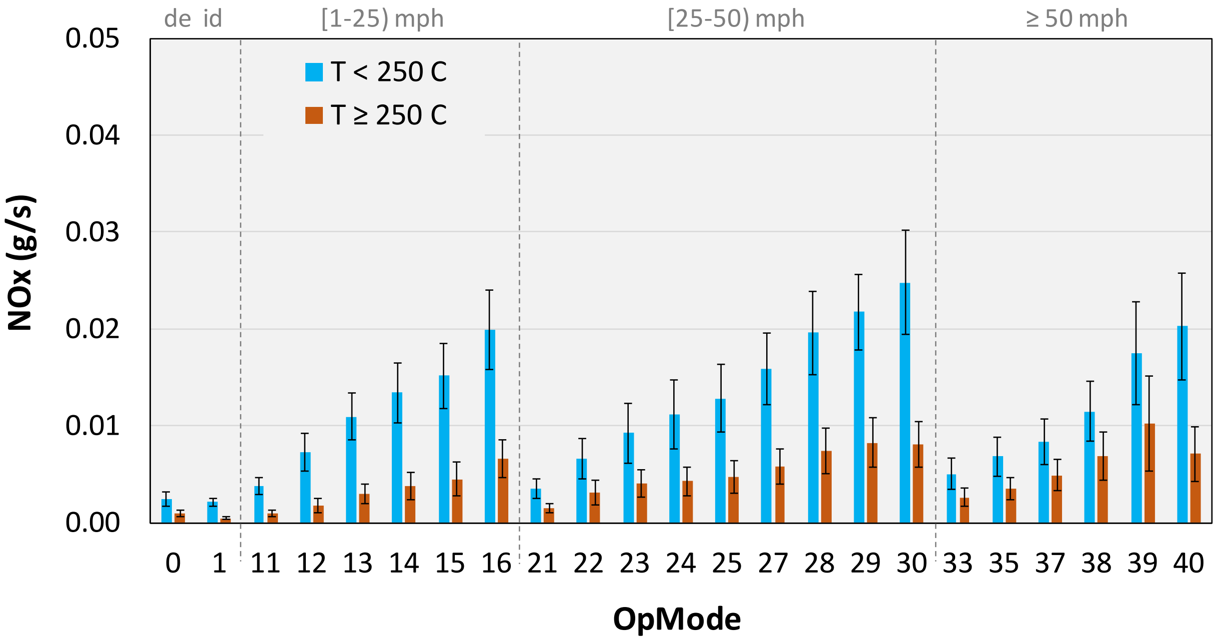


Figure 2-10: NO_X Emission Rates from 42 Vehicles with 0.20 g/bhp-hr FEL LHD Engines by MOVES OpMode and Aftertreatment Temperature

Figure 2-11, Figure 2-12, and Figure 2-13 compare the same data in terms of operational time by aftertreatment temperature. As expected, when the vehicles are operating at idle or low speeds, more time is spent at the lower temperature bin. However, even at high speeds, a nontrivial amount of time is spent at aftertreatment temperatures below 250°C.

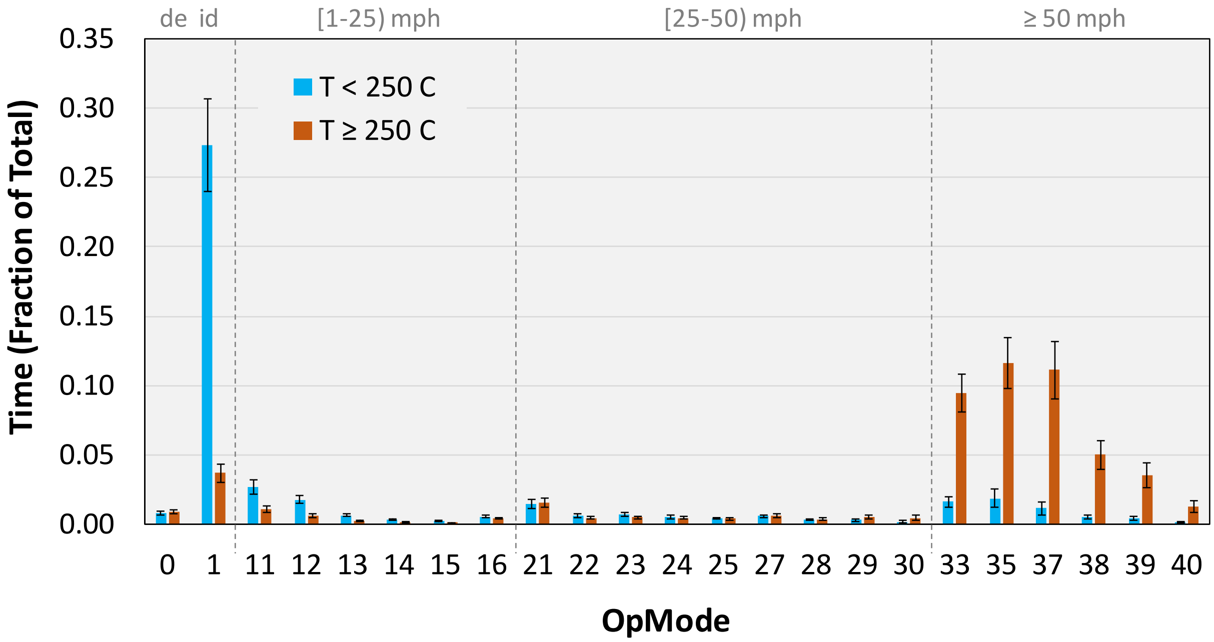


Figure 2-11: Time Fraction from 81 Vehicles with 0.20 g/bhp-hr FEL HHD Engines by MOVES OpMode and Aftertreatment Temperature

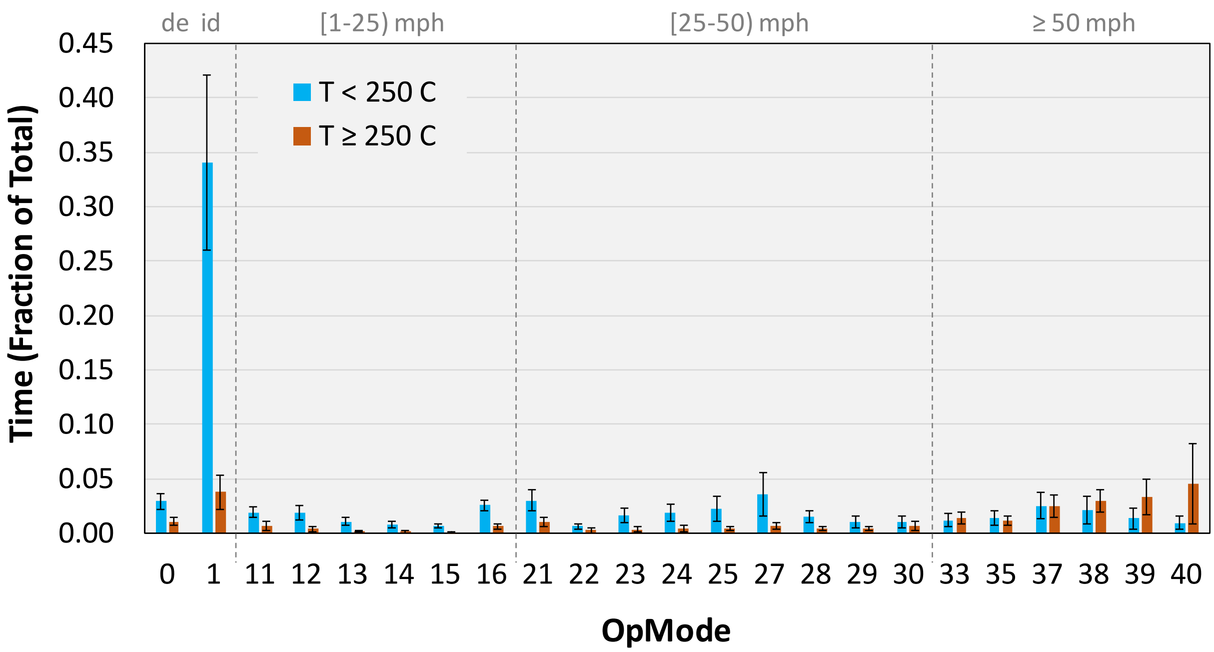


Figure 2-12: Time Fraction from 20 Vehicles with 0.20 g/bhp-hr FEL MHD Engines by MOVES OpMode and Aftertreatment Temperature

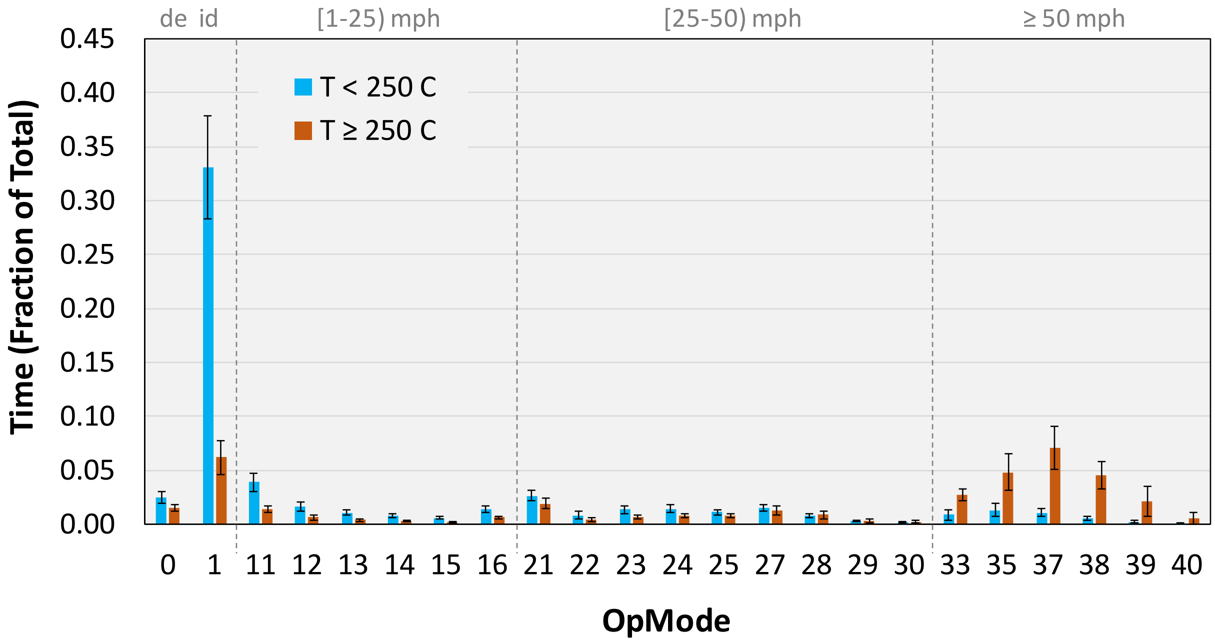


Figure 2-13: Time Fraction from 42 Vehicles with 0.20 g/bhp-hr FEL LHD Engines by MOVES OpMode and Aftertreatment Temperature

By combining the NO_X emission rate data with the time data, we can estimate the total NO_X contribution by operation, as shown in Figure 2-14, Figure 2-15, and Figure 2-16. The imbedded tables in each figure display the fraction of data in each temperature bin by operation time and NO_X mass. The emissions increase from low aftertreatment temperatures is not uniform across all operating modes. For HHD engines (Figure 2-14), the aftertreatment temperatures spent nearly as much time below 250°C as above it, but the contribution to total NO_X is much higher from the lower temperature operation due to the higher emission rates. The MHD and LHD engines spent much more time in low aftertreatment temperatures conditions, and it is reflected in a higher contribution to NO_X. For all engines, the low- and mid-speed operating ranges contribute the most NO_X emissions. These figures highlight the need to consider both activity and emission rate to effectively reduce NO_X.

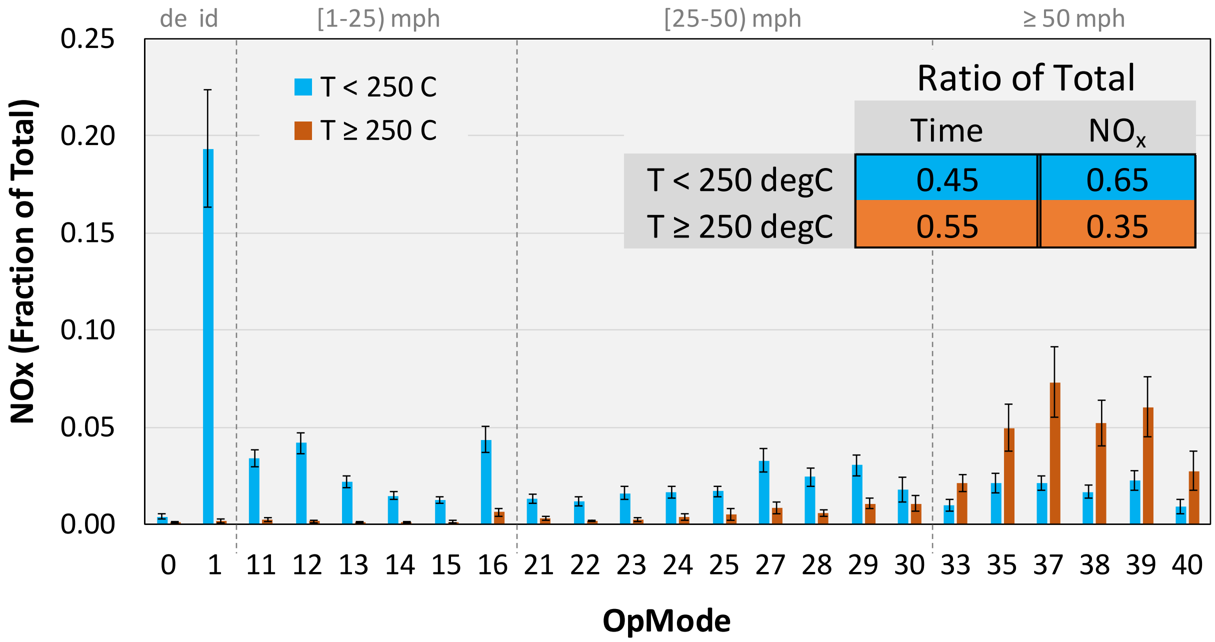


Figure 2-14: Total NO_X Contribution from 81 vehicles with 0.20 g/bhp-hr FEL HHD Engines by MOVES OpMode and Aftertreatment Temperature

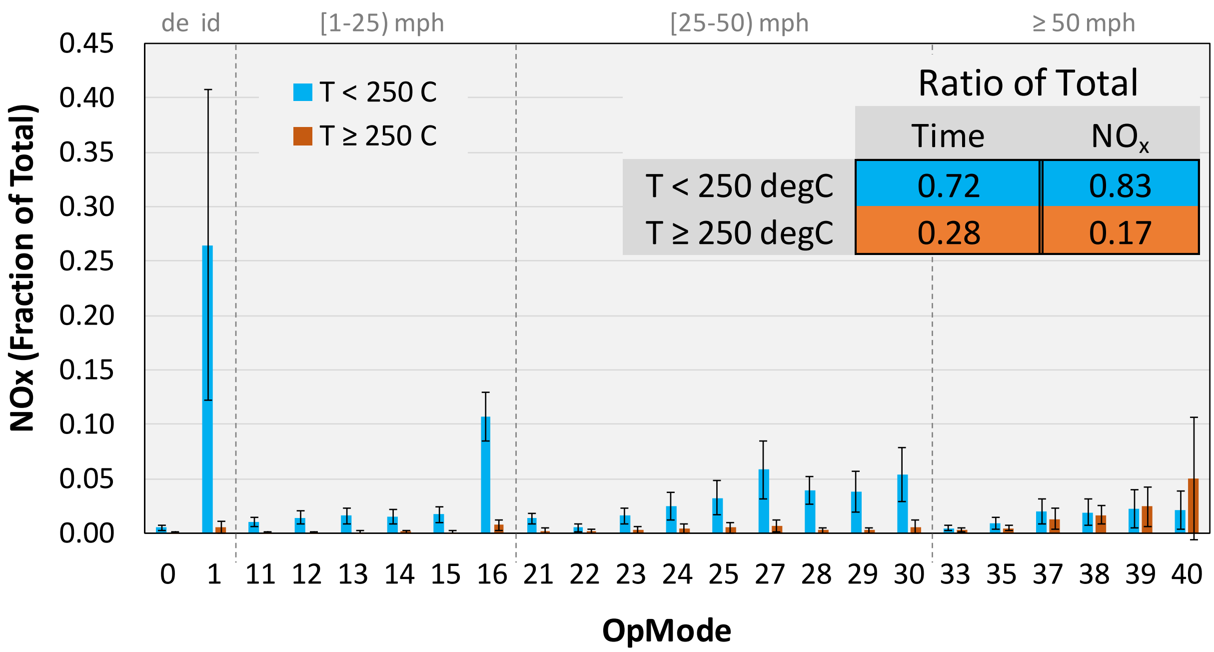


Figure 2-15: Total NO_X Contribution from 20 vehicles with 0.20 g/bhp-hr FEL MHD Engines by MOVES OpMode and Aftertreatment Temperature

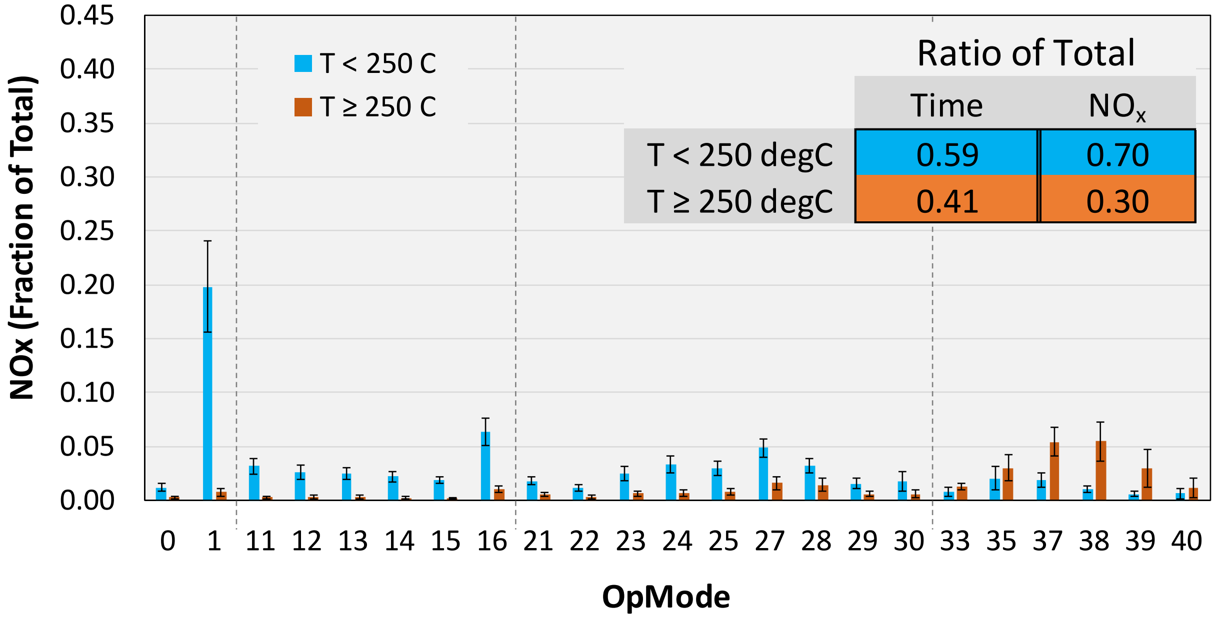


Figure 2-16: Total NO_X Contribution from 42 vehicles with 0.20 g/bhp-hr FEL LHD Engines by MOVES OpMode and Aftertreatment Temperature

2.2.2.4 HDIUT Data by Speed

Figure 2-17, Figure 2-18, and Figure 2-19 show the data binned by speed for HHD, MHD, and LHD vehicles, respectively, with NO_X FEL at or below 0.20 g/bhp-hr. Operation below 5 km/hr was excluded to fully remove any idle time or low creep operation, where emission rates are drastically higher.[D] The 0.20 g/bhp-hr FEL vehicles' average NO_X emission rate in the lowest speed bin is much higher compared to higher speeds. The overall average NO_X emission rate, shown in the right-most bin, is more than twice the certification value of 0.20 g/hp-hr. The figure suggests the difference is even greater for vehicles that spend significant time at lower speeds. The average time fraction (diamond marker) and the average NO_X fraction (square marker) reveal the outsized contribution to total NO_X from the low and medium speed bins where the time fraction is lower than the NO_X fraction.

[D] Average NOx emission rate for HHD, MHD, and LHD are 6.38 g/bhp-hr, 7.65 g/bhp-hr, and 4.61 g/bhp-hr, respectively.

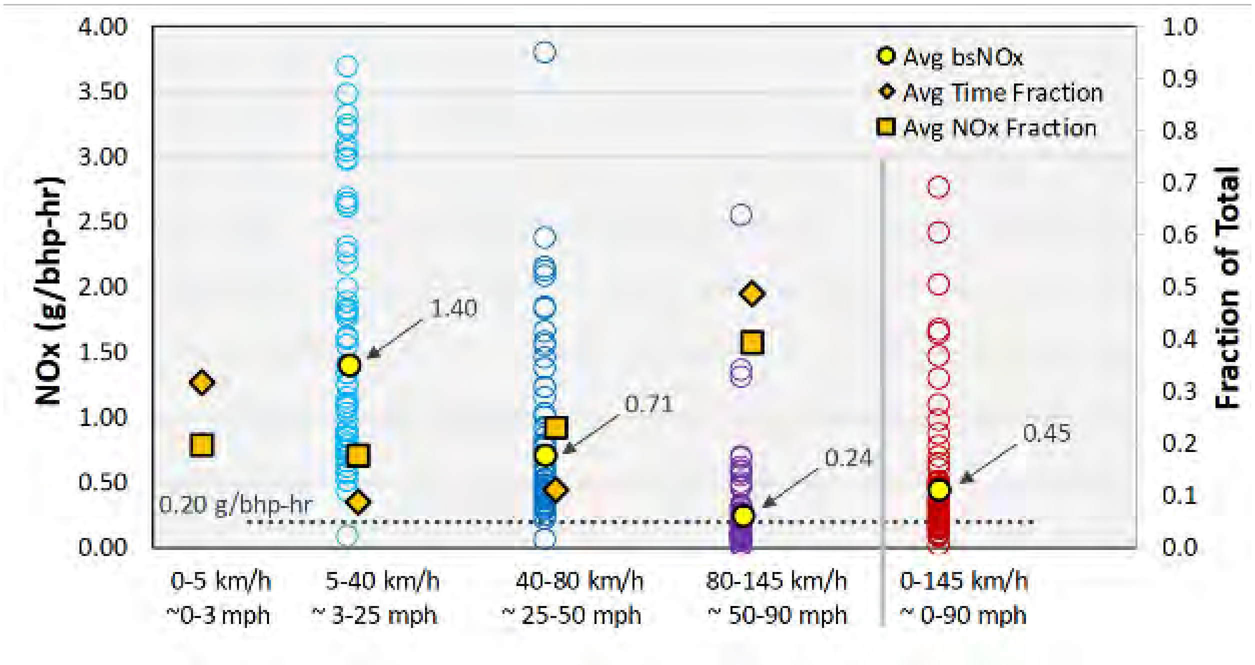


Figure 2-17: Brake-specific NO_X by Vehicle Speed Bins for 93 Vehicles with HHD Diesel Engines and an FEL of 0.20 g/bhp-hr

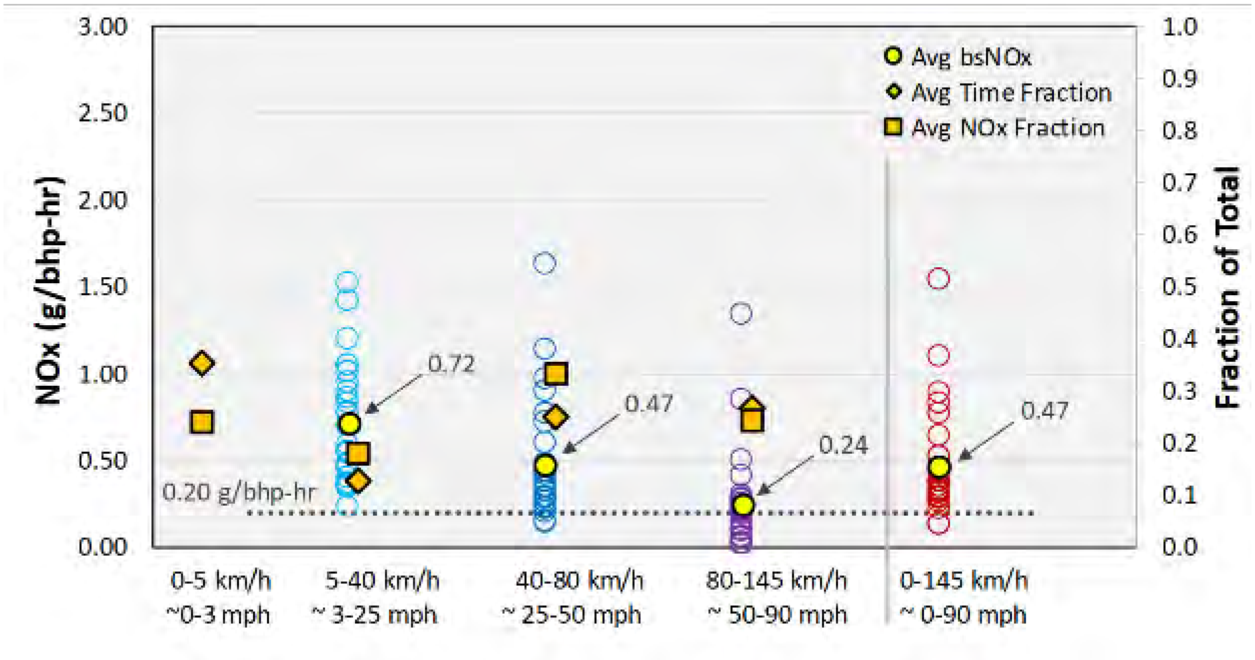


Figure 2-18: Brake-specific NOx by Vehicle Speed Bins for 26 Vehicles with MHD Diesel Engines and an FEL of 0.20 g/bhp-hr

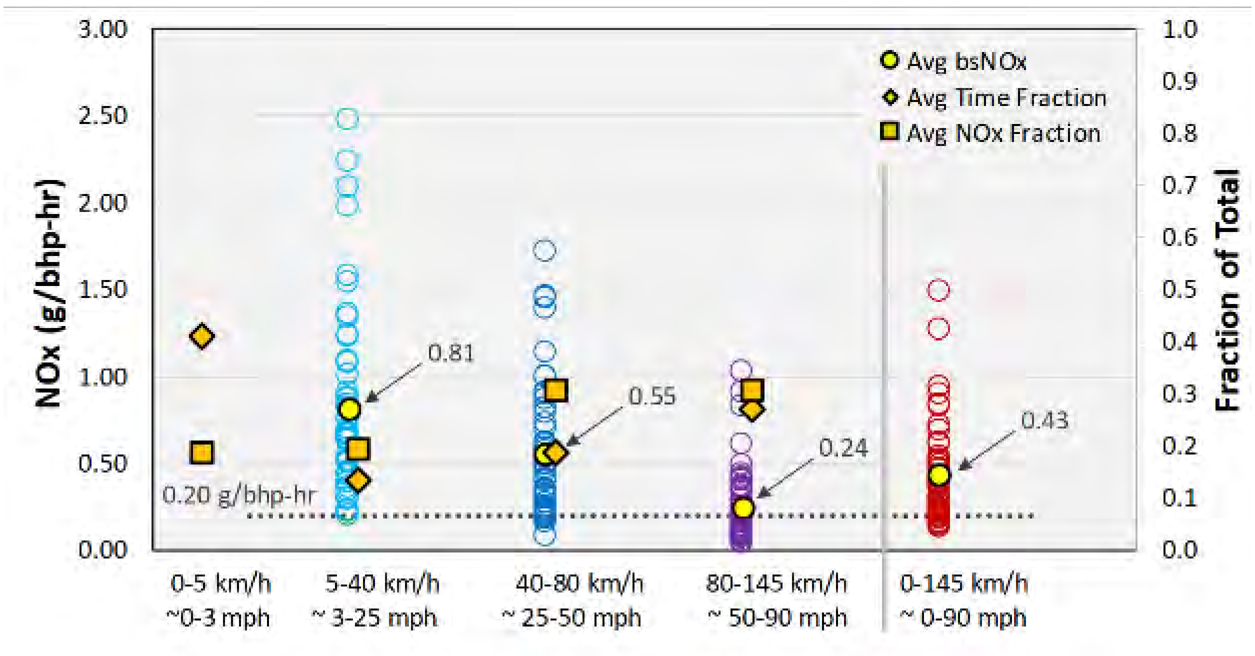


Figure 2-19: Brake-specific NOx by Vehicle Speed Bins for 49 Vehicles with LHD Diesel Engines and an FEL of 0.20 g/bhp-hr

2.2.2.5 HDIUT Data by Work-Based Window

Figure 2-20 shows a comparison of brake-specific NO_X (g/bhp-hr) calculated using the standard method ("bsNO_X method") and CO_2 based method ("NO_X/CO_2 method"). This graph is based on measurement data from 85 HHD diesel vehicles with NO_X FEL $\leq$ 0.20 g/bhp-hr. The binning is by average power of the window over engine rated power. The windows are continuous and non-exclusive – window n+1 starts at time t=n+1. The amount of time in each window is based on the engine power demand during the window. This analysis led to 2.90 million windows. The height of the columns represents the mean of all the windows in that power bin and the error bars represent standard deviation of the mean. The 95% confidence interval is not shown since windows are not independent. The CO_2-based method is more robust at very small loads, such as the 0-5% and 5-10% average power windows. In these cases, the small amount of work done (bhp-hr) leads to higher brake-specific NO_X values for the standard method while also causing very large standard deviation. The CO_2 based method addresses the low load in the denominator issue while also not penalizing vehicles that have lower CO_2 emissions, by normalizing against CO_2 over work (the second term in the equation). Another takeaway is that emissions are much higher at lower loads, lowest at loads near the FTP and SET cycles, and then creep up at higher loads. This suggests the engines are tuned to perform best at loads/conditions similar to the certification cycle while less optimized for other real-world operation.

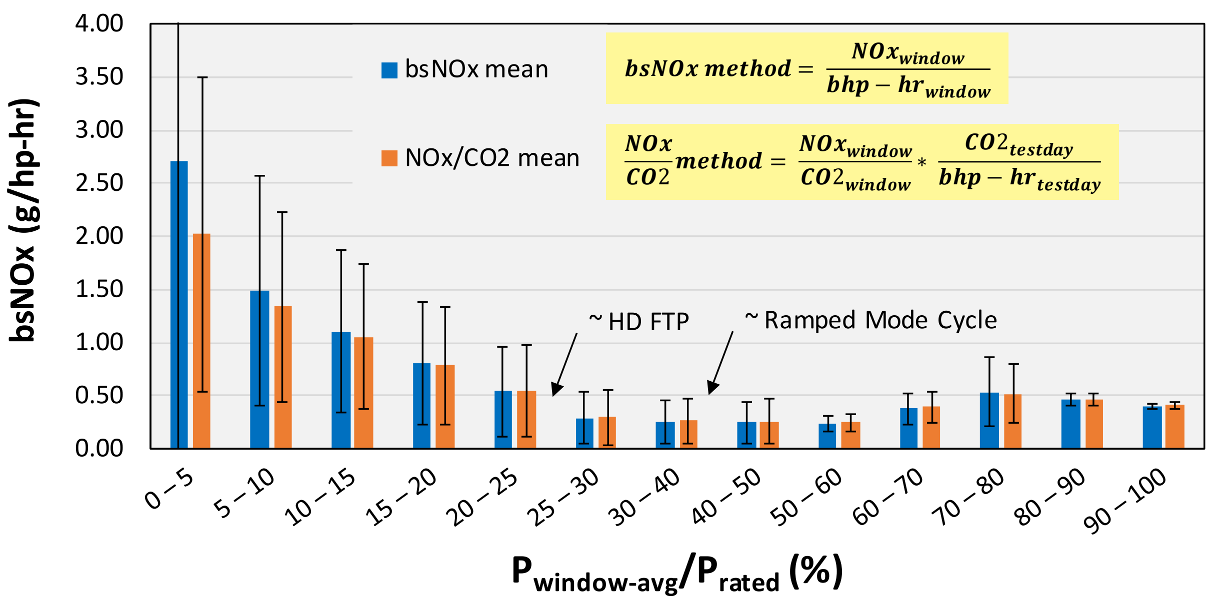


Figure 2-20: Brake-specific NO_X by Window Average Power Bins for 85 Vehicles with HHD Diesel Engines and an FEL of 0.20 g/bhp-hr

2.2.2.6 HDIUT Data by Simulated Cycle

Figure 2-21 and Figure 2-22 show analysis using simulated cycles. Figure 2-21 shows the drive cycles, which are converted to an OpMode time distribution, which is then combined with the OpMode based emission rates for each vehicle in the HDIUT data set. The MOVES national run OpMode distribution is also included for comparison in Figure 2-22. The Vehicle FTP and UDDS drive cycles have similar speed traces and thus their OpMode distributions are also similar. The transient drive cycle has a considerable amount of low-speed transient operation, which shows up as higher time spent in OpModes 11-14 and particularly OpMode 12. The MOVES national run for combination-long haul trucks has most of its operation at highway speeds, and thus most of the time is allocated to OpModes 33 and above.

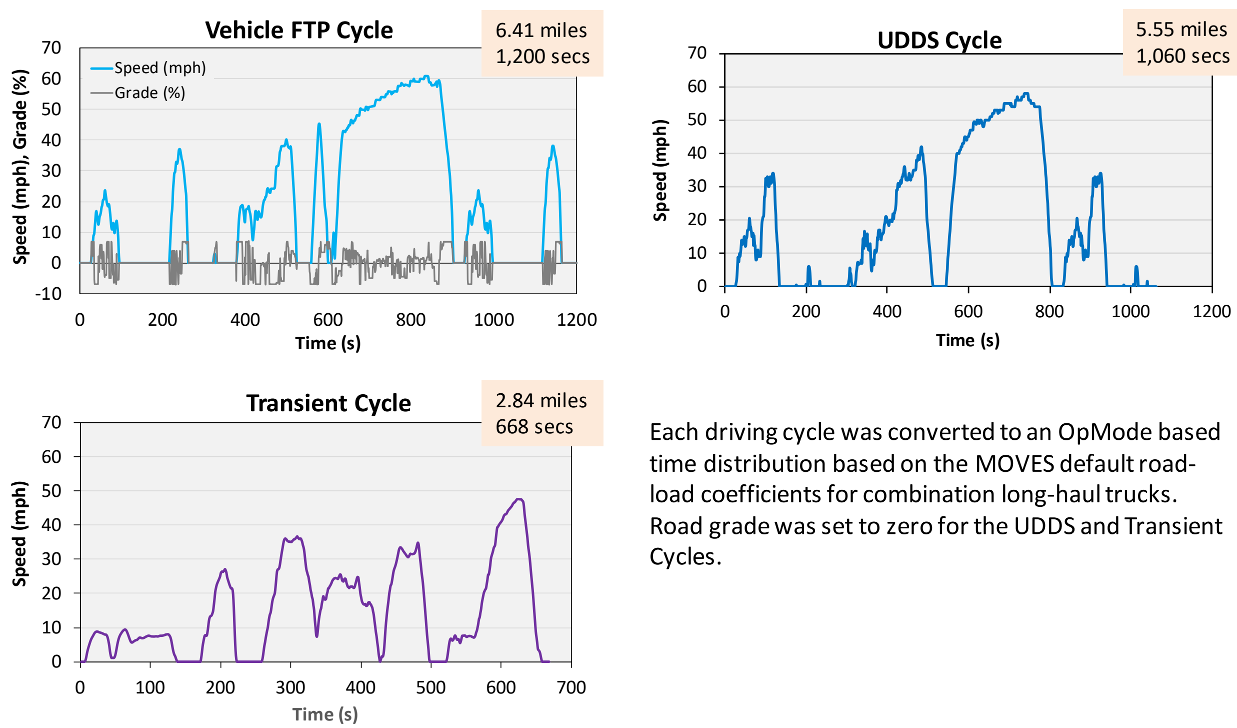


Figure 2-21: Vehicle Speed Profile of HD Duty Cycles

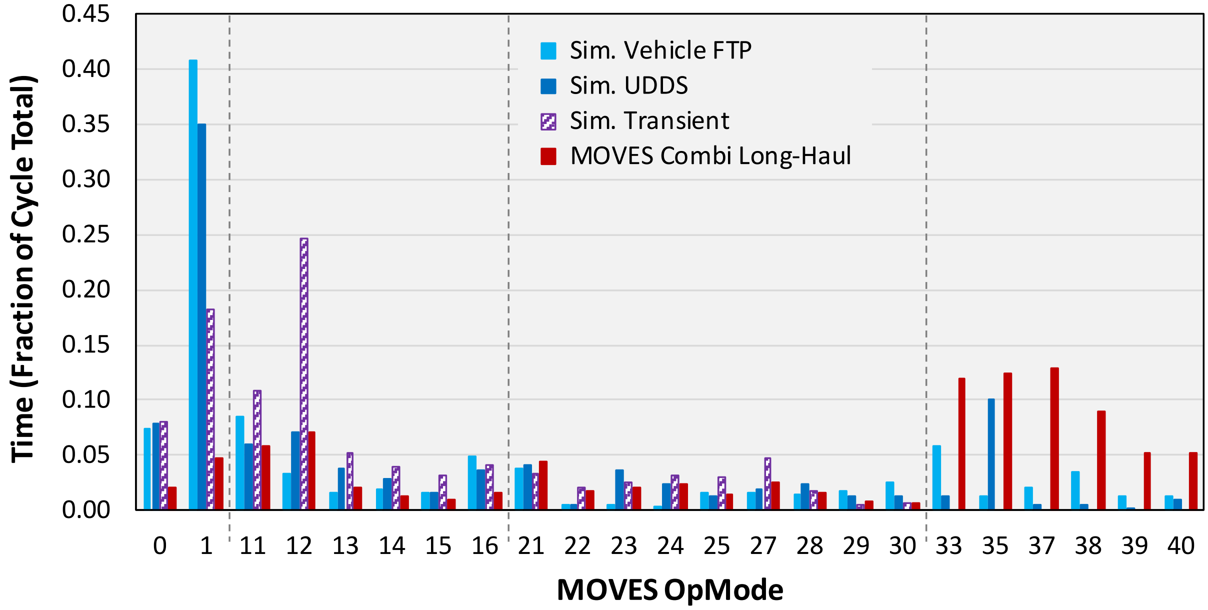


Figure 2-22: MOVES OpMode Time Fraction for each Simulated HD Combination Long-Haul Duty Cycle

Figure 2-23 and Figure 2-24 show the cycle average NO_X emission rates calculated using the per vehicle OpMode based emission rate and the drive cycle OpMode time distribution shown in Figure 2-22. The average rate and spread for Vehicle FTP and UDDS are similar since the cycles are similar. The transient cycle produces the highest average rate because of low-speed operation that has lower aftertreatment temperature. The transient cycle also has the largest spread, similar to the larger spread for the low-speed analysis in Figure 2-17. The MOVES cycle has the lowest average and spread because the operation is predominantly in the high-speed zone, where emission rates are better controlled.

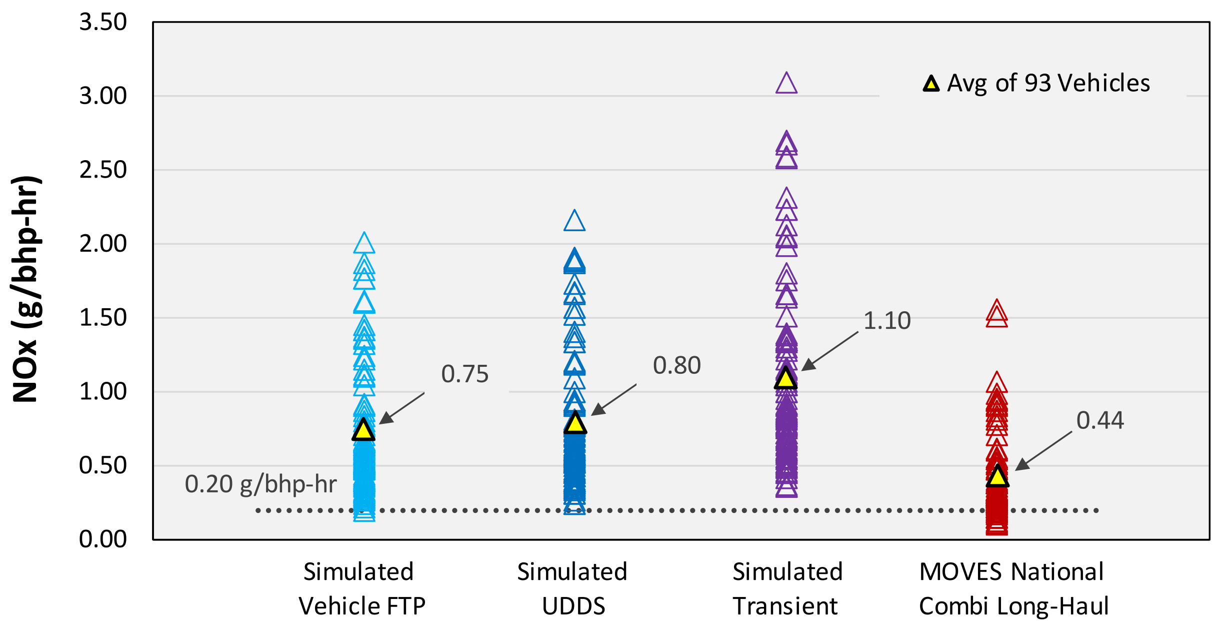


Figure 2-23: Brake-specific NOx emissions by simulated cycle for HHD diesel engines with NOx FEL of 0.20 g/bhp-hr

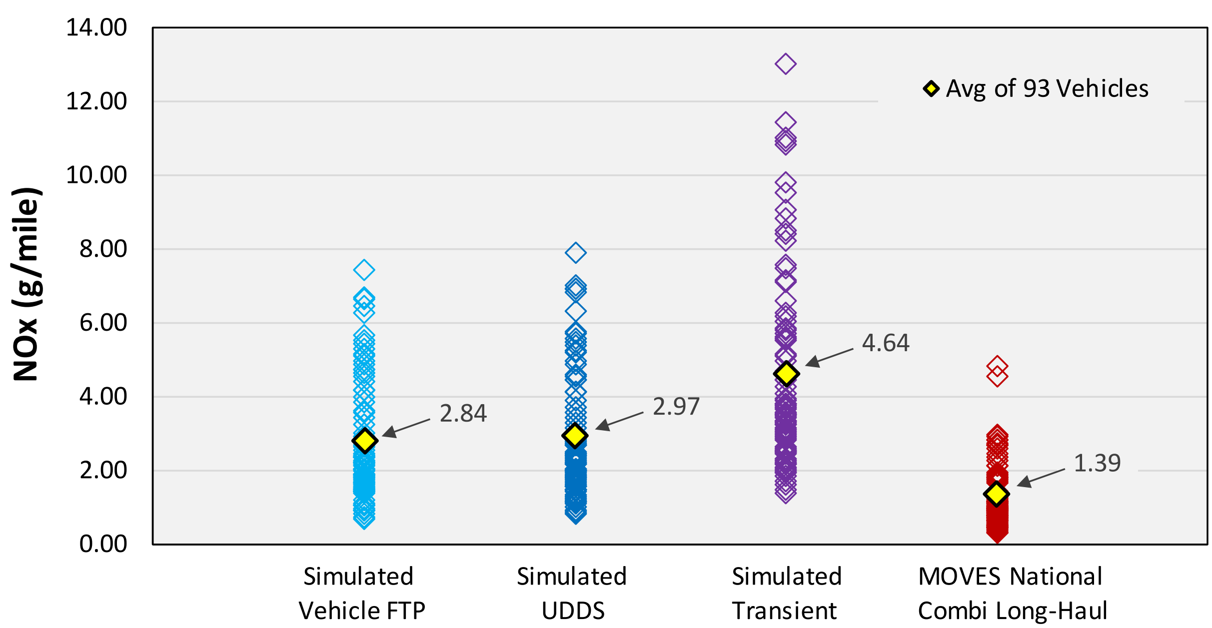


Figure 2-24: Distance-specific NOx emissions by simulated cycle for HHD diesel engines with NOx FEL of 0.20 g/bhp-hr

2.2.3 **Final Updates to CI Engine Off-Cycle Test Program and Off-Cycle Standards**

The focus of the current off-cycle NTE standards and off-cycle NTE compliance testing is operation at relatively high load; the data analysis procedure thus excludes a wide range of vehicle operations that occur in the real world, in particular operations at lower loads. Importantly, the excluded portion of the data makes up the bulk of vehicle operation, specifically areas where NO_X production is high.

To improve the coverage of the off-cycle standard, test procedures, and field testing program, we have finalized updates to the off-cycle standard and test procedures to include a broader range of vehicle operation that is now covered by the regulated off-cycle standard. To keep the results representative of actual engine/aftertreatment performance and minimize issues with temporally misaligned data, we have finalized an analysis methodology based on a series of moving average windows (MAW).

2.2.3.1 Background on Euro VI MAW

The European Union Euro VI emission standards for heavy-duty engines requires testing for in-service conformity starting with model year (MY) 2014 engines using portable emission measurement systems (PEMS).[13,14] The intent is to confirm whether heavy-duty engines continue to comply with the emissions standards while in use, under normal operation, over time, under real world conditions. Manufacturers must check for "in-service conformity" by operating their engines over a mix of urban, rural, and freeway driving on prescribed routes using portable emission measurement system (PEMS) equipment to measure emissions. Compliance is determined using a work-based windows approach where emissions data are evaluated over segments or "windows." A window consists of consecutive 1 Hz data points that are summed until the engine performs an amount of work equivalent to the European transient engine test cycle (World Harmonized Transient Cycle).

Engines are tested over a mix of urban, rural, and freeway driving. Testing starts at 18 months and a minimum of 25,000 km, and continues every two years thereafter out to seven years and 700,000 km. There are no carve-outs for engine load or aftertreatment low temperature operation. There are carve-outs for altitude (> 5577 ft), maximum ambient temperature (100 °F at sea level), and minimum ambient temperature ≥ 20 °F. There is no cold start emission measurement requirement. Emission and work integration start when engine coolant temp >70°C and is stable to within ± 2°C, or 20 minutes after engine start, whichever is first. Vehicle payload must be 50 to 60 % of maximum.

Compliance is determined using a work-based windows approach where emissions are evaluated over data segments or windows. For the window to be considered valid, the average power within a window must be ≥ 15% of engine maximum power for MY 2014 – 2018 and ≥ 10% for MY 2019 and later. The vehicle must accumulate at least 5 complete windows over its shift day and 50% of the windows must be valid. Compliance is demonstrated at 1.5 times the EURO VI emission limit and there is no separate measurement allowance to account for field testing measurement uncertainty (it is built into the conformity factor multiplier of 1.5).

EPA and others have compared the performance of US-certified engines and Euro VI-certified engines and concluded that the European engines' NO_X emissions are comparable to US 2010 standards-certified engines under city and highway operation, but lower in light-load conditions.[15] This suggests that manufacturers responded to the Euro VI test procedures by designing their emission controls to perform well over broader operation. EPA's final rulemaking expands our off-cycle standard and test procedures to capture nearly all real-world operation. Our final approach is similar to the European field testing program, with key distinctions that improve upon the Euro VI approach, as discussed below.

2.2.3.2 Final Updates

The updated off-cycle testing data analysis method uses a MAW methodology similar to that established in the Euro VI emission standards. However, most carve-outs are eliminated. Additionally, in order to adequately capture all vehicle operation, there is no minimum power requirement for valid windows. There are no prescribed routes for our field testing compliance program, as the previous NTE program required data to be collected in real-world operation. In what we believe to be an improvement to a work-based window, the final rule uses a moving average window (MAW) approach consisting of time-based windows. Instead of basing window size on an amount of work, the MAW includes a window size of 300 seconds.[E] The time-based windows are intended to equally weight each data point collected.

We also recognize that it is difficult to develop a single standard that is appropriate for the entire range of operation that heavy-duty engines experience. For example, a numerical standard for CO_2 specific NO_X that is technologically feasible under worst case conditions such as idle, would be higher than the levels that are feasible when the aftertreatment is functioning optimally.

Thus, the final rule has separate standards for distinct modes of operation. The 300-second windows constructed from the second-by-second field data are grouped into one of two bins using the nominal "normalized average CO_2 rate" from the certification test cycles to identify the boundaries. The normalized CO_2 rate is defined as the average CO_2 rate in the window divided by the engine's maximum CO_2 rate. The engine's maximum CO_2 rate is defined as the engine's rated maximum power multiplied by the engine's family certification level (FCL) for the FTP certification cycle.

Windows with a normalized average CO_2 rate of 6 percent or less (6 percent is equivalent to the average power of the low-load certification cycle) are classified as idle and binned together (bin 1). Windows with a normalized average CO_2 rate greater than 6 percent are classified as non-idle operation and binned together (bin 2).

The emissions performance of the binned data in bin 1 is determined using the sum of the total mass of emissions divided by the sum of CO_2 mass emissions. Emissions performance for the binned data in bin 1 is determined using the mass rate (total mass of emissions divided by total time) of the emissions. This "sum-over-sum" approach successfully accounts for all emissions; however, it requires the measurement system (PEMS) to be accurate across the complete range of emission concentrations.

As mentioned previously, there is a separate MAW-based standard for each bin. In the NTE-based program, the NTE standards are 1.5 times the certification duty-cycle standards. Similarly,

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[E] Our evaluation includes weighing our current understanding that shorter windows are more sensitive to measurement error and longer windows make it difficult to distinguish between duty cycles.

for the MAW-based standards, the off-cycle standards for each bin correspond to one or more laboratory-based cycles, with each bin having its own standard.

2.2.3.3 Data Collection and Exclusion

For the HDUIT, emissions data are to be collected from key-on (t = 0) until the end of the shift day when the engine is turned off. Data are to be collected once every second (i.e., 1-Hz data).

From the data collected at 1 Hz, some data points are to be excluded from the remaining process. Data to be excluded are (1) data collected when an analyzer or flow meter is performing in-service zero and span drift checks or zero and span calibrations and emission data cannot be collected, (2) data collected where the engine is off except in some circumstances where the vehicle under testing is equipped with stop-start and/or automatic engine shutdown systems as described in the preamble Section III.C, (3) data collected during infrequent regeneration events, (4) data collected when any approved AECD for emergency vehicle applications is active, and (5) data collected when ambient temperatures are below 5 °C, or when ambient temperatures are above the altitude-based value determined using Equation 40 CFR 1036.530-1.

2.2.3.4 Defining Windows

With the extended idle times frequently present in HDIUT samples, a work-based window approach would include longer periods of time for these windows, and the methodology would be very sensitive to small inaccuracies in power measurement. To ensure the final set of windows more accurately reflects the operation of the vehicle, we have adopted a time-based window approach, where each window contains an equal amount of time rather than an equal amount of work, as in the Euro VI work-based window approach.

For this methodology, a window will consist of the summation of 300 consecutive 1-Hz data points (i.e., a 300-second window). The windows are continuous and non-exclusive, with subsequent windows beginning one second after previous windows (i.e., at the next data point). The first window will begin at initial key-on (t = 0), and the final sequential window will begin 300 seconds before the last data point taken. To limit the impact of instances where data exclusions would reduce the weighting of an individual data point, exclusions of ≤ 600 seconds are removed and the remaining data concatenated. For exclusions > 600 seconds, the final pre-exclusion window begins 300 seconds before the exclusion, and the next subsequent window begins immediately after the exclusion.

Except for the data points at the beginning and end of the test and those around long data exclusions, this methodology equally weights emissions at each data point during the off-cycle testing. We believe this is appropriate, as the under-weighted data points consist of a small percentage of the HDIUT data, which contain a minimum of 10,800 1-Hz data points.

2.2.3.5 Emission bins

The agency recognizes that including operation currently excluded from the standard, including low-load operation and low aftertreatment temperature, will result in a higher range of variability in both the vehicle operation represented, and in the data captured during testing. Thus, we are differentiating the data collected by vehicle operation, and independently set standards for each operational characteristic.

To differentiate among various types of operation, the windows are divided among two bins that are characterized by the normalized average CO_2 rate: an idle bin (bin 1) and a non-idle load bin (bin 2). The normalized CO_2 rate of each window is defined as the total window CO_2 mass divided by the 300-second window length and then divided by the maximum CO_2 rate of the engine. The engine's maximum CO_2 rate is defined as the engine's rated maximum power multiplied by its family certification level (FCL) for the FTP certification cycle.

The two bins are defined as follows:

- Bin 1: window normalized average CO_2 rate ≤ 6 %
- Bin 2: window normalized average CO_2 rate > 6 %

2.2.3.6 Bin Size and Test Validity

For a test to be considered valid, bin 1 must contain a minimum of 2,400 windows and bin 2 must contain at least 10,000 windows. To ensure there are enough windows in bin 1, the engine may be idled at the end of the shift day. If the vehicle has tamper-resistant idle-reduction technology that prevents idling, populate bin 1 with additional windows by setting the 1-Hz emission rate for all regulated pollutants to zero as described in § 1036.415(g) to achieve exactly 2,400 bin 1 windows. If bin 2 contains fewer than 10,000 windows, or bin 1 contains less than 2,400 windows after inclusion of the optional end-of-day idle period, the vehicle must be tested over an additional shift day. The resulting windows from the second or subsequent shift day are added to the appropriate bin, so that all windows from all shift days are included.

Using data from 168 previous HDUIT tests of one shift day each, the 1-Hz data from these tests were collected into windows and binned according to the above process as seen in Figure 2-25. Of these single shift day tests, 98% contained over 10,000 windows in bin 2, and 80% contained over 2,400 windows in bin 1. From these data, we estimate that nearly all tests would be valid with a single shift days' worth of data, assuming manufacturers take advantage of the optional end-of-day idle period.

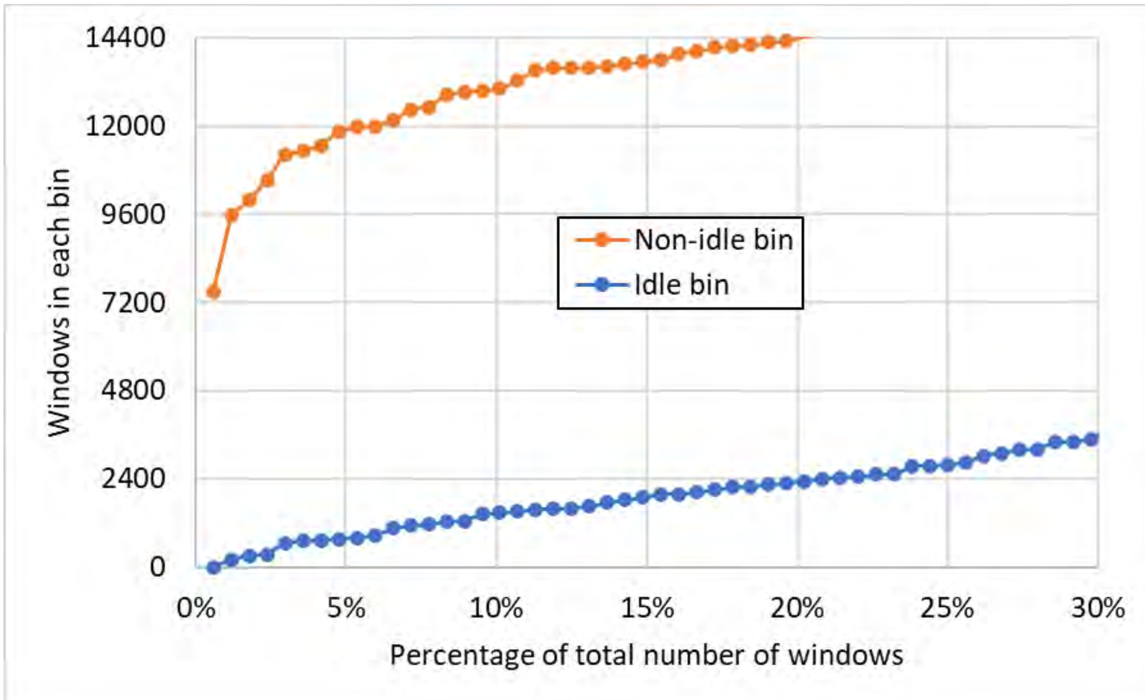

Figure 2-25: Number of windows in each bin from 168 HDUIT shift days, sorted in order (the first 30% are shown).

2.2.3.7 Defining Bin Emissions

Historically, engine standards have been work-specific. Using this approach, the standards can apply to a wide range of engine sizes. Where this approach is challenged is when the test interval that is being evaluated has very little, to no work, produced, such that the emissions are divided by zero or near zero. This methodology also does not rely on estimating or recording the second-by-second power output of the engine to determine work done. For this standard, the engine's FTP FCL CO_2 emission value ($e_{CO2FTLFCL}$) is used to convert the CO_2 specific emissions to equivalent work specific emissions.

For the HDUIT data processing, emission values are calculated for each bin, using a sum-over-sum value. For bin 2, CO_2 specific emissions are determined, and converted to work specific emissions using the engine's FTP FCL CO_2 emission value ($e_{CO2FTLFCL}$) as follows:

Equation 2-2

$$e\left(\frac{g}{hp-hr}\right) = \frac{Total\ bin\ emission}{Total\ bin\ CO_2} \times e_{CO2FTPFCL}$$

In bin 1, under nominal idle conditions the engine produces no work, giving an incentive to minimize CO_2 production in these operation modes (for example, with cylinder deactivation). So as not to artificially increase the stringency of the bin 1 standard for engines with low idle CO_2, we are setting standards as emission rates rather than work specific values. For bin 1, the emission rate is also calculated using a sum-over-sum value as follows:

Equation 2-3

$$e\left(\frac{g}{hr}\right) = \frac{Total\ bin\ emission}{Total\ bin\ time}$$

2.2.3.8 Bin Emissions and Standards

These standards apply to the sum-over-sum emissions value for the entire bin, as shown in Equation 2-2 and Equation 2-3. This methodology accounts for all emissions included in a particular bin equally and reduces the influence of potential errors in data collection.

2.3 Spark-Ignition Test Procedures and Standards

Spark-ignition test procedures are a crucial aspect of the heavy-duty criteria pollutant program. This rulemaking establishes new test procedures and requirements for spark-ignition engine compliance. This section describes the existing test procedures as well as the development process for the new requirements. This includes the determination of emission levels as well as the development of new duty cycles.

2.3.1 Current SI Test procedures

Heavy-duty spark-ignition engines currently are certified for criteria pollutants using the Heavy-Duty Otto-Cycle Engine Federal Test Procedure (HDOE FTP). For 2007 and later Heavy-Duty engines, 40 CFR part 86 – "Control of Emissions from New and In-Use Highway Vehicles and Engines" and 40 CFR part 1065 – "Engine Testing Procedures" detail the certification process. 40 CFR 86.007-11 defines the standard settings of Oxides of Nitrogen, Non-Methane Hydrocarbons, Carbon Monoxide, and Particulate Matter. The HDOE FTP duty cycle is defined in 40 CFR part 86, appendix I. All emission measurements and calculations are defined in Part 1065, with exceptions as noted in 40 CFR 86.007-11. The data requirements are defined in 40 CFR 86.001-23 and 40 CFR 1065.695.

The measurement method for CO is described in 40 CFR 1065.250. For measurement of NMHC refer to 40 CFR 1065.260. For measurement of NO_X refer to 40 CFR 1065.270. For measurement of PM, refer to 40 CFR 1065.140, 1065.170, and 1065.290. Table 1 of 40 CFR 1065.205 provides performance specifications that we recommend analyzers meet. Note that 40 CFR 1065.307 provides linearity verifications that the system must meet. For the calculation method brake specific mass emissions for CO, NMHC, NO_X and PM refer to 40 CFR 1065.650.

2.3.1.1 HDOE FTP

The current HDOE FTP is a transient test consisting of second-by-second sequences of engine speed and torque pairs with values given in normalized percent of maximum form. The cycle was computer generated from a dataset of 88 heavy-duty trucks in urban operation in New York and Los Angeles. This procedure is well-defined, mirrors in-use operating parameters, and continues to be appropriate also for the continued assessment of criteria pollutant emissions from heavy duty engines.

A complete HDOE FTP involves three test sequences. First, a 20-minute test is run over the duty-cycle with the engine at the same ambient temperature as the test cell (between 68°F and 86°F). The engine undergoes a 10-minute hot-soak following the cold-start. A 20-minute hot start test is run over the same duty-cycle following the hot-soak. The HDOE FTP emission level for the engine is determined by weighting the cold start emissions by 1/7 (14 percent) and the hot-start emission results by 6/7 (86 percent).

2.3.1.2 Test Procedure Engine Mapping Improvements

The heavy-duty FTP test cycle is composed of second-by-second speed and torque targets that are based on the engine's design operating speeds and the torque levels produced over the full range of allowed speeds. In order to determine the torque level at any engine speed, a mapping of the engine is performed prior to the actual FTP testing. The mapping is a sweep across the mechanically- or electronically-allowed operating speeds to determine the highest possible torque level the engine can produce at any specific speed. From this "maximum torque" sweep, the FTP targets are determined for the subsequent transient FTP test and for any additional test (e.g., SET).

As noted above, the measured torque values are intended to represent the maximum torque the engine can achieve under fully warmed-up operation with the appropriate design fuel grade (i.e., regular grade octane fuel) across the allowed range of engine speeds from idle to the typically electronically limited highest RPM. The intent is to reflect a torque value that is maintained if the engine is stabilized at a specific speed over a longer period of time such as what might be observed in a "real world" condition when an engine is held by a transmission in a specific RPM by the selected gear.

Electronic control of all aspects of engine hardware and operation has resulted in some challenges to performing the mapping of the engine. Variable torque levels have been observed in engine testing related to such things as electronic control response to fuel octane, anticipated exhaust thermal conditions, transmission torque limiting models, and other electronic features incorporated into the engine management strategies. This torque variability has been particularly evident after the change to the mapping test procedures in 40 CFR 1065.510(b)(5)(i), which requires that an engine be mapped by performing a transient sweep from idle to maximum rated speed. Prior to this change in the test procedure, the mapping procedure required the engine to stabilize at discrete engine speed breakpoints before recording the engine torque value for that engine speed that would be used for the FTP and other testing.

There are two potential improvements to reduce the torque measurement variability. One option could be to perform the sweep in both directions (i.e., idle to rated maximum speed and back to idle) and determine the torque value at any speed to be the highest measured torque level. Typically, we would expect any torque limiting AECDs (e.g., power enrichment delay) to be active during the sweep up, allowing the maximum torque to be determined during the sweep down after any torque limiting AECD has concluded. A second option could require manufacturers to turn off the torque limiting AECDs during the torque mapping sequence and allow them to perform the procedure as written today. For example, an AECD that results in delay of component thermal protection due to a temperature model would not be allowed for the power mapping sequence since any stabilized operation at a temperature limited speed point would typically require thermal protection. During the transient FTP test however, the AECD could be active to simulate real world operation and the implemented thermal management strategies.

2.3.2 **Summary of Updates Considered for SI Test Procedures and Standards**

We are updating the location of our highway heavy-duty engine regulations, moving from the current 40 CFR part 86 to part 1036. As part of this process, we clarified our nomenclature and no longer refer to "otto-cycle" engines; instead, these engines are more accurately labeled spark-ignition engines throughout part 1036. This section provides additional details related to the test procedures in Section III.D. of the preamble to this rulemaking. Refer to the preamble for direction on the final procedures.

2.3.2.1 HD SI FTP

As part of our migration to part 1036, the FTP duty cycle maintains the weighting factors for the duty cycle speed values from the current HDOE FTP duty cycle that applies to criteria pollutant regulation in 40 CFR part 86, Appendix I(f)(1). We changed to the negative torque values, as noted below. The HD Technical Amendments that were published in June 2021 finalized the migration of some heavy-duty highway engine standard setting part test procedures from 40 CFR part 86 to part 1036. This included the migration of the HDOE FTP drive schedule to 40 CFR part 1036, appendix B(b) in order to add vehicle speed and road grade to the duty-cycle to facilitate powertrain testing of hybrid powertrains for compliance with the HD Phase 2 GHG standards.

We are finalizing in this rule changes that better align certification to GHG standards with criteria pollutant testing; specifically, the removal of and footnoting of the negative normalized vehicle torque values over the HDOE FTP duty-cycle. The footnote denotes that these torque points are controlled using closed throttle motoring, which matches how negative torque values have been controlled in the HDDE FTP. This change reflects the way that engine manufacturers were already controlling to negative torque from spark-ignition engines and harmonizes the methodology with the HDDE FTP, with no effect on stringency.

The spark-ignition engine denormalization equation in 40 CFR 86.1333(a)(1)(ii) contains a divide by 100 which equates it to the denormalization equation in 40 CFR 1065.610(c)(1) (equation 1065.610-3); thus the elimination of the 40 CFR part 86 equation from the standard setting part will have no consequence.

2.3.2.2 Engine mapping

We are finalizing a change to the procedure for SI engine torque mapping in 40 CFR part 1065.510. In order to determine the torque level at any engine speed, a mapping of the engine is performed prior to the actual FTP testing. The mapping is a sweep across the mechanically- or electronically-allowed operating speeds to determine the highest possible torque level the engine can produce at any specific speed. From this "maximum torque" sweep, the FTP targets are determined for the subsequent transient FTP test and for any additional test (e.g., SET).

The measured torque values are intended to represent the maximum torque the engine can achieve under fully warmed-up operation with the appropriate design fuel grade (i.e., regular grade octane fuel) across the allowed range of engine speeds from idle to the typically electronically limited highest RPM. The intent is to reflect a torque value that is maintained if the engine is stabilized at a specific speed over a longer period of time such as what might be observed in a "real world" condition when an engine is held by a transmission to a specific RPM by the selected gear.

Variable torque levels have been observed in engine testing related to such things as electronic control response to fuel octane, anticipated exhaust thermal conditions, transmission torque limiting models, and other electronic features incorporated into the engine management strategies. This torque variability has been particularly evident with the change to the mapping test procedures in 40 CFR part 1065.510(b)(5)(i), which allowed an engine to be mapped by performing a transient sweep from idle to maximum rated speed. Prior to this change in the test procedure, the mapping procedure required the engine to stabilize at discrete engine speed breakpoints before recording the engine torque value for that engine speed that is used for the FTP and other testing.

We are finalizing the requirement that the engine achieve a stabilized torque reading at different speeds prior to recording the final torque values. This is accomplished by disabling any controls that limit or reduce torque during the engine mapping test.

2.3.2.3 Supplemental Emissions Test for HD SI

As noted in Section 2.1.1.2, the compression-ignition engines currently comply with SET-based standards that represent high-speed and high-load operation. The SET duty cycle is a ramped modal cycle in which the engine is tested on an engine dynamometer over a sequence of steady-state modes. As we show in Section 4.2.3.2, there are opportunities to reduce emissions in high-load operating conditions where engines often experience enrichment for either catalyst protection or a power boost. We are finalizing SET-based standards for HD SI engines to ensure that emission controls are properly functioning in the high load conditions covered by that duty cycle. We are finalizing the same CI-based SET procedure, summarized in Section 2.1.2.2, for HD SI engines, including the existing weighting factors shown in Table 2-2.

2.3.2.4 Onboard Refueling Vapor Recovery

The current ORVR test procedure, which can be found in 40 CFR part 1066, subpart J, for measuring emissions from chassis-certified vehicles during a refueling event, requires that the testing occur in a sealed housing evaporative determination (SHED) enclosure containing the complete vehicle. This procedure applies to all light-duty and heavy-duty complete vehicles subject to the ORVR standards, and manufacturers designed and built the SHEDs at their test facilities for these vehicles.

During a recent test program, EPA discovered that very few SHEDs are available that could fit vehicles in the over-14,000 lb GVWR class because of a combination of the length and height of these work vehicles. Additionally, the limited large volume SHEDs that were available at third-party laboratories proved to have challenges measuring the refueling emissions because of the very large volume inside the enclosures.[16]

Large background volumes of ambient air create a challenge for evaporative emissions testing because a measurement is only considered representative if emissions are able to reach a homogeneous distribution throughout the cell prior to initiating a measurement by the emission analyzers. In EPA's test program, we found that the two heavy-duty test vehicles required a substantially longer mixing time than the current test procedure developed for light-duty vehicles.[16,17] Another challenge to adapting the existing procedures for larger vehicles is that the calculations must account for the volume displaced in the SHED by the test vehicle, which can be highly variable for the range of commercial vehicle designs. If a manufacturer opts for performing a full SHED test for ORVR certification, we considered adjusting the duration of the test to achieve a representative emissions distribution in the larger SHED, as well as adjustments to calculating the displaced volume of these diverse vehicle designs in order to get an accurate measurement of the refueling emissions.[16,17]

Figure 2-26 and Figure 2-27 show examples of the estimated extrapolated mixing time for the two trucks that were tested in the summer of 2018. The data does not show how much additional mixing time is necessary to achieve stabilization. Extrapolated test results from this test program suggest that at least three additional minutes of mixing time would be needed.

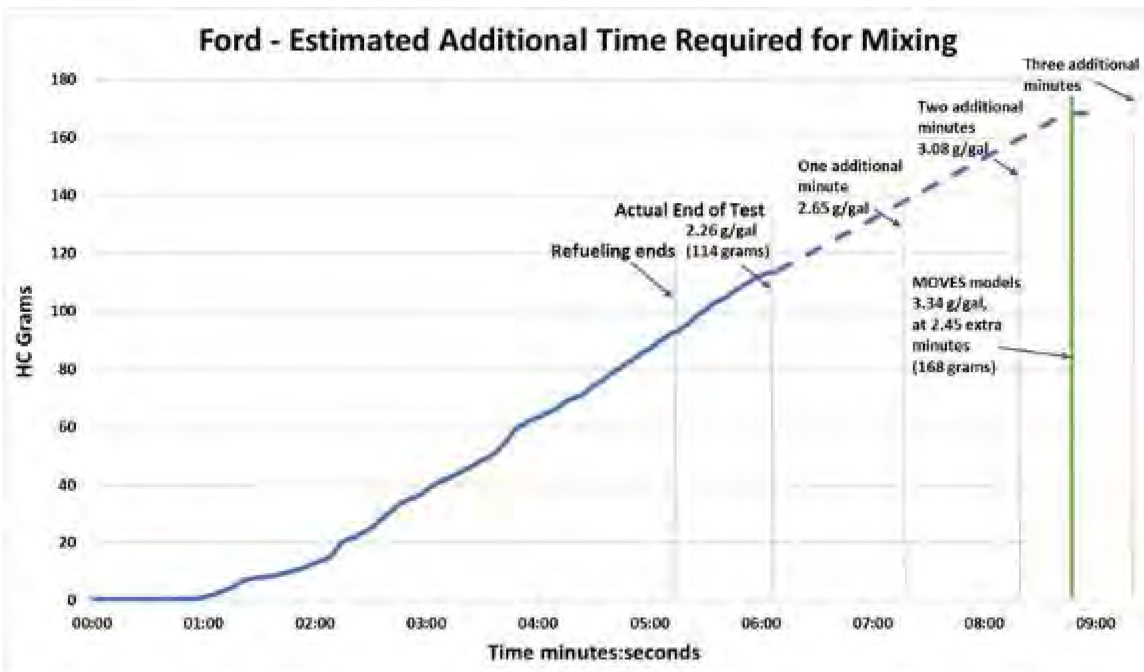

Figure 2-26: Estimated Projected Ford E-450 ORVR Results based on Extrapolation

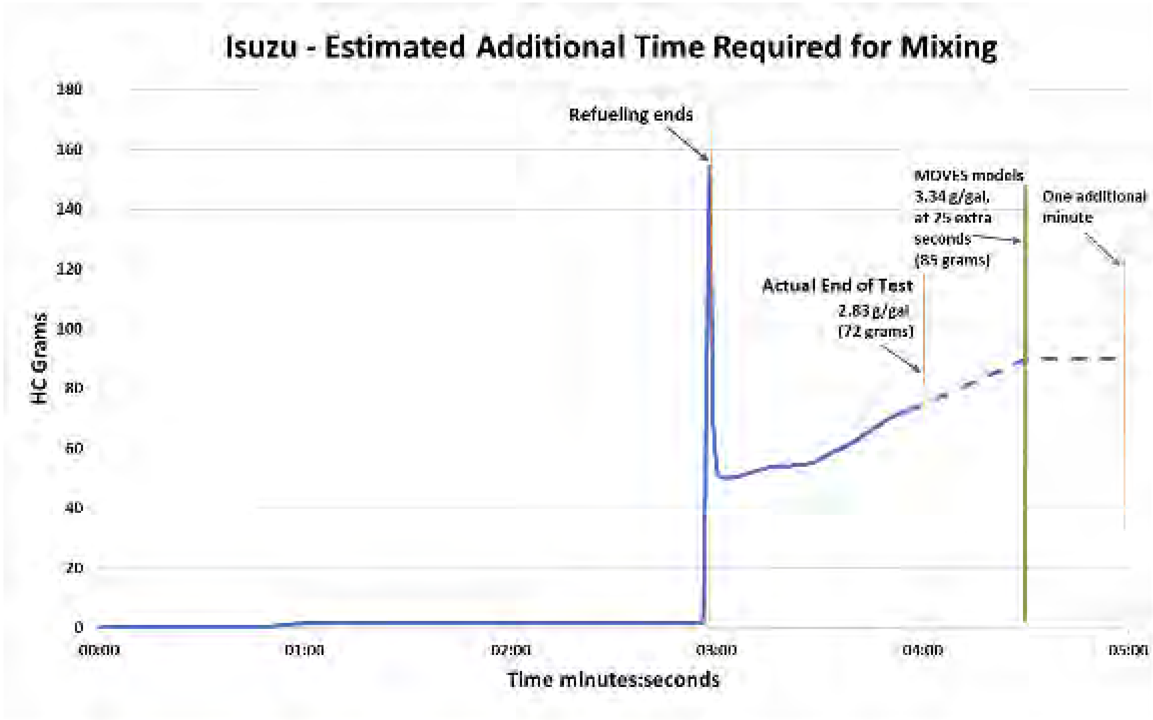

Figure 2-27 Estimated Projected Isuzu ORVR Results based on Extrapolation

Since there is limited availability of large volume SHED equipment, we considered the benefits and challenges of other options to demonstrate compliance with refueling standards for these large commercial vehicles. One option we considered was testing the complete ORVR system with all of the components (fuel tank, filler pipe, canister, control valves, etc.) independent of the actual vehicle using an existing SHED designed for smaller vehicles, such as light-duty applications. These existing SHED enclosures are widely available to test and certify other vehicles that are currently subject to refueling standards. This approach of only testing the components associated with refueling emissions would remove the challenge of finding a SHED with sufficient dimensions to contain the vastly larger (i.e., longer and taller) commercial vehicles that are part of this final rule. Testing the refueling related components independent of the vehicle would also eliminate the challenge of minimizing other hydrocarbon sources not associated with fuel or the fuel system (i.e., tires, plastics, paints, etc).

Another option we considered was allowing independent ORVR hardware described above to be tested in a small enclosure designed for only component specific testing (i.e., mini-SHED) similar to the methodology for the "rig" test, allowed for other evaporative testing by California and accepted by EPA. Similar to the ORVR system-based concept described previously, the mini-SHED approach would provide a simpler test methodology that would capture just the refueling-related emissions. Furthermore, this smaller scale, component-based test would eliminate much of the variability encountered when attempting to test a full vehicle or hardware in a large SHED.

For both system- and component-based ORVR testing, the canister would require a specific conditioning cycle to get it into a representative state of a real-world loading entering a refueling event. This could be performed on or correlated to an actual vehicle driven over an existing EPA test cycle or a "real world" drive cycle. The current preparatory cycle used by today's ORVR-required vehicles is designed for light-duty vehicle driving patterns and vehicles with typically much smaller fuel tanks and canisters. The current conditioning procedure is designed to challenge the purge system in scenarios such as heavy traffic, slow speeds and start-stop events. Heavy-duty vehicles, with larger fuel tanks and canisters, may drive more miles and have greater power demands that may help purge the larger canisters more easily than allowed for in the current light duty vehicle test. Commercial vehicles may still drive under heavy traffic scenarios but the expectation is that they will drive more miles daily and operate under higher loads on a regular basis which help to purge larger amount of vapors from the system.

Another option we considered was allowing engineering analysis in lieu of testing, similar to what is done for evaporative emission standards today. The current 2-day and 3-day evaporative standards program for vehicles greater than 14,000 lb GVWR acknowledges that the lighter classes (i.e., under 14,000 lb GVWR) often use the same hardware-and purge-related calibrations. Because of this similarity between the classes and a high degree of consistent performance, the agency currently allows the data and testing from the lighter vehicles to be accepted for the certification compliance demonstration of the larger class of vehicles. We expect manufacturers will apply ORVR systems similar to their lighter vehicle classes on the larger vehicles as a result of this rule. In these cases, an engineering analysis could provide a similar level of emission control assurance if applied as a means to demonstrate compliance with new refueling standards for the larger vehicles.

2.3.2.5 Idle Test Procedures Considered

It is important to ensure that the main component of the emission control system, the catalyst, remains effective during prolonged idle situations. The current heavy-duty FTP test does not include extended idle conditions indicative of real-world behaviors when work vehicles are started and allowed to idle while warming-up the engine or when the vehicle operator requires the interior temperature to stabilize with either heat or air conditioning. This prolonged idle can also occur when the vehicle is brought to a stop for unloading or a similar situation.

We considered the addition of a new test procedure to ensure emission controls are maintained during idle. We considered the FTP or SET run as a pre-conditioning cycle to stabilize the engine and emission control system, followed by 10-30 minutes of idle. Previous idle provisions, established for in-use inspection and maintenance programs, required a 0.50 percent by volume CO limit over a 30-minute idle period, but did not set a limit for NMHC or NO_X.[18]

We also considered options that take advantage the existing SET duty cycle to avoid introducing a new test procedure. One option was to reevaluate the weighting factors of the SET to place a greater emphasis on the idle modes, but this option has two drawbacks. First, providing more weight to the idle mode deemphasizes the high-load modes that we believe are critical to encouraging reduced fuel enrichment. Second, the existing SET procedure collects the emissions from all modes in a single bag for analysis, which means idle performance would be masked by the composite result and the additional weighting at idle would require us to reevaluate the SET standard feasibility. Another option was to require two bags for SET such that the idle modes would be collected in the second bag. This option would isolate the idle results but would require two separate standards. While this option reduced the need for a new test procedure, manufacturers would still likely need to make adjustments to their test cells to accommodate the second bag.

Finally, we considered ensuring catalyst operation at idle by driving manufacturers to maintain an effective catalyst bed operating temperature that promotes appropriate idle emission controls, particularly during real world conditions that include prolonged idling typically followed by a driveaway. Current technology shows that 350 °C is a typical low limit for effective catalyst operation. If manufacturers control the engine operation such that the catalyst bed temperature is maintained above this light-off temperature, proper emissions control during idle conditions and drive-away events is expected to be achieved.

2.4 Useful Life

In addition to emission standards and test procedures, appropriate regulatory useful life periods are critical to assure emission performance of heavy-duty highway engines. Our regulations require manufacturers to perform durability testing to demonstrate that engines will meet emission standards not only at certification but also over the full useful life periods specified by EPA. Useful life represents the period over which emission standards apply for certified engines, and, practically, any difference between the regulatory useful life and the generally longer operational life of in-use engines represents miles and years of operation without an assurance that emission standards will continue to be met. In this section, we present a summary of the history of our regulatory useful life provisions and describe our estimates of the length of operational lives of heavy-duty highway engines, which are almost double the current useful life mileages in EPA's regulations for all primary intended service classes.

2.4.1 History of Regulatory Useful Life

The Clean Air Act specifies that emission standards under section 202(a) "shall be applicable to such vehicles and engines for their useful life … whether such vehicles and engines are designed as complete systems or incorporate devices to prevent or control such pollution." Practically, this means that to receive an EPA certificate of conformity under the CAA, a manufacturer must demonstrate that an engine or vehicle, including the aftertreatment system, will meet each applicable emission standard, including accounting for deterioration, over the useful life period specified in EPA's regulations. In addition, CAA section 207(c) requires manufacturers to recall and repair vehicles or engines if the Administrator determines that "a substantial number of any class or category of vehicles or engines, although properly maintained and used, do not conform to the regulations prescribed under [section 202(a)], when in actual use throughout their useful life (as determined under [section 202(d)])." Taken together, these sections define two critical aspects of regulatory useful life: (1) the period over which the manufacturer must demonstrate compliance with emissions standards to obtain EPA certification, and (2) the period for which the manufacturer is subject to in-use emissions compliance liability, e.g., for purposes of recall. Manufacturers perform durability testing to demonstrate that engines will meet emission standards over the full useful life. Manufacturers may perform scheduled maintenance on their test engines only as specified in the owner's manual and consistent with our maintenance regulations. As part of the certification process, EPA approves such scheduled maintenance, which is also subject to minimum maintenance intervals as described in the regulation.

EPA prescribes regulations under CAA section 202(d) for determining the useful life of vehicles and engines. CAA section 202(d) provides that the minimum useful life for heavy-duty vehicles and engines is a period of 10 years or 100,000 miles, whichever occurs first. This section authorizes EPA to adopt longer useful life periods that we determine to be appropriate. Under this authority, we established useful life periods for heavy-duty engines by primary intended service class. Heavy-duty highway engine manufacturers identify the primary intended service class for each engine family by considering the vehicles for which they design and market their engines.[F] Heavy-duty compression-ignition engines are distinguished by their potential for rebuild and the weight class of the final vehicles in which the engines are expected to be installed.[G] Heavy-duty spark-ignition engines are generally classified as a single "spark-ignition" service class unless they are designed or intended for use in the largest heavy-duty vehicles and are thereby considered heavy heavy-duty engines.[H] The following useful life periods currently apply to the criteria pollutant emission standards for heavy-duty highway engines:[I,J]

- 110,000 miles or 10 years for heavy-duty spark-ignition engines and light heavy-duty compression-ignition engines
- 185,000 miles or 10 years for medium heavy-duty compression-ignition engines
- 435,000 miles, 10 years, or 22,000 hours for heavy heavy-duty compression-ignition engines

In our 1983 rulemaking, which first established class-specific useful life values for heavy-duty engines and vehicles, EPA adopted the principle that useful life mileage values should reflect the typical mileage to the first rebuild of the engine (or scrappage of the engine if that occurs without rebuilding).[19] Significantly, this approach was adopted at a time when diesel engine emission control strategies relied mainly on in-cylinder engine combustion controls.

Over time, mileage values became the primary metric for useful life duration. This is because, due to advancements in general engine durability, nearly all heavy-duty engines reach the mileage value in-use long before 10 years have elapsed. The age (years) value has meaning for only a small number of low-annual-mileage applications, such as refuse trucks. Also, manufacturer durability demonstrations generally target the mileage values, since deterioration is a function of engine work and hours rather than years in service and a time-based demonstration would be significantly longer in duration than one based on applicable mileage value.

[F] See 40 CFR 1036.140 as referenced in the definition of "primary intended service class" in 40 CFR 86.090-2.

[G] As specified in the current 40 CFR 1036.140(a), light heavy-duty engines are not designed for rebuild and are normally installed in vehicles at or below 19,5000 pounds GVWR; medium heavy-duty engines may be designed for rebuild and are normally installed in vehicles from 19,501 to 33,000 lbs GVWR; heavy heavy-duty engines are designed for multiple rebuilds and are normally installed in vehicles above 33,000 pounds GVWR.

[H] See 40 CFR 1036.140(b).

[I] See 40 CFR 86.004-2. EPA adopted useful life values of 110,000, 185,000, and 290,000 miles for light, medium, and heavy heavy-duty engines, respectively, in 1983 (48 FR 52170, November 16, 1983). The useful life for heavy heavy-duty engines was subsequently increased to 435,000 miles for 2004 and later model years (62 FR 54694, October 21, 1997).

[J] The same useful life periods apply for heavy-duty engines certifying to the greenhouse gas emission standards, except that the spark-ignition standards and the standards for model year 2021 and later light heavy-duty engines apply over a useful life of 15 years or 150,000 miles, whichever comes first. See 40 CFR 1036.108(d).

In the 1997 rulemaking that most recently increased heavy-duty engine useful life, EPA included an hours-based useful life of 22,000 hours for the heavy heavy-duty engine intended service class. This unique criterion was added to address the concern that urban vehicles, particularly urban buses, equipped with heavy heavy-duty engines had distinctly different driving patterns compared to the line-haul trucks from which the agency based its useful life value of 435,000 miles.[20] Commenters in that rulemaking indicated that urban bus average speed was near 13 miles per hour. Considering that speed, many of these bus engines would reach the end of their operational life or be candidates for rebuild before the applicable mileage value or the 10-year criterion is reached. The 22,000 hours value was adopted in lieu of a proposed minimum useful life value of 290,000 miles for heavy heavy-duty engines. Considering the current 435,000 useful life mileage for heavy heavy-duty engines, the 22,000-hour useful life value only has significance for the small subset of vehicles equipped with heavy heavy-duty engines with an average speed of less than 20 miles per hour.

In the Phase 1 GHG rulemaking, we promulgated useful life periods for the GHG emission standards for heavy-duty highway engines and their corresponding heavy-duty vehicles that aligned with the current useful life periods for criteria pollutant emission standards.[21] In the HD Phase 2 GHG rulemaking, we extended the useful life for Light HDV, light heavy-duty engines, and spark-ignition engines for the GHG emission standards to 15 years or 150,000 miles to align with the useful life of chassis-certified heavy-duty vehicles subject to the Tier 3 standards.[22] See 40 CFR 1036.108 and 40 CFR part 1037, subpart B, for the GHG useful life periods that apply for heavy-duty highway engines and vehicles, respectively.

2.4.2 Identifying Appropriate Useful Life Periods

Emission standards apply for the engine's useful life and manufacturers must demonstrate the durability of engines to maintain certified emission performance over their useful life. Thus, the emission standards must be considered together with their associated useful life periods. Larger useful life mileage values require manufacturers to demonstrate emission performance over a longer period and should result in effective emission control over a greater proportion of an engine's operational (sometimes referred to as "service") life. Consistent with our approach to adopting useful life mileages in the 1983 rulemaking, we continue to consider a comprehensive out-of-frame rebuild to represent the end of a heavy-duty CI engine's "first life" of operation. For SI engines that are less commonly rebuilt, engine replacement is a more appropriate measure of an engine's operational life.

2.4.2.1 Compression-Ignition Engine Rebuild Data

In 2013, EPA commissioned an industry characterization report on heavy-duty diesel engine rebuilds.[23] The report relied on existing data from MacKay & Company surveys of heavy-duty vehicle operators. In this report, an engine rebuild was categorized as either an in-frame overhaul (where the rebuild occurred while the engine remained in the vehicle) or an out-of-frame overhaul (where the engine was removed from the vehicle for more extensive service).[K] The study showed that the mileage varied depending on the type of rebuild. Rebuilding an engine

[K] Note that these mileage values reflect replacement of engine components, but do not include aftertreatment components. At the time of the report, the population of engines equipped with DPF and SCR technologies was limited to relatively new engines that were not candidates for rebuild.

while the block remained in the frame was typically done at lower mileage than rebuilding an engine removed from the vehicle. The results of the study by vehicle weight class are presented in Table 2-10.

Table 2-10 Average Mileage and Age at First Rebuild for Heavy-Duty CI Engines From 2013 EPA Rebuild Industry Characterization Report

Vehicle Weight Class	In-Frame Rebuild		Out-of-Frame Rebuild	
	Mileage	Years	Mileage	Years
Class 3	216,900	9.5	256,000	9.5
Class 4	236,800	11.6	346,300	10.3
Class 5	298,300	10.9	344,200	11.9
Class 6	332,200	13.0	407,700	10.6
Class 7	427,500	15.8	509,100	13.2
Class 8	680,200	11.9	909,900	8.9

McKay & Company does not collect information on aftertreatment systems (e.g., diesel oxidation catalysts, SCR systems, or three-way catalysts), so neither EPA's 2013 report nor CARB's more recent report for their HD Omnibus rulemaking include aftertreatment system age information.[L] We consider the mileage at rebuild or replacement of an engine to represent the operational life of that engine, including any aftertreatment components that were part of its original certified configuration. We have no data to indicate aftertreatment systems lose functionality before engines are rebuilt or replaced, and our technology demonstrations Chapter 3 show aftertreatment catalysts are able to maintain performance when bench-aged to beyond the equivalent of current useful life mileages.

We averaged the mileages for these vehicle classes according to EPA’s primary intended service classes for heavy-duty CI engines as defined in 40 CFR 1036.140. Specifically, we averaged Classes 3, 4, and 5 to represent Light HDE, Classes 6 and 7 to represent Medium HDE, and Class 8 to represent Heavy HDE. These values are shown in Table 2-11 with the current useful life mileages that apply to each intended service class. As seen in the tables, the study reported typical engine rebuild mileages that are more than double the current useful life mileages for those classes.

[L] See Chapter 2.5.2.3 for a summary of the CARB report that reflects engine rebuilds and replacements occurring between calendar years 2012 and 2018.

Table 2-11 Average Mileage at First Rebuild for Heavy-Duty CI Engines Based on EPA Intended Service Classes

Primary Intended Service Class	Mileage at First In-Frame Rebuild	Mileage at First Out-of-Frame Rebuild	Current Useful Life Mileage
Light HDE (Vehicle Classes 3-5)	250,667	315,500	110,000[a]
Medium HDE (Vehicle Classes 6-7)	379,850	458,400	185,000
Heavy HDE (Vehicle Class 8)	680,200	909,900	435,000

[a] The useful life mileage that applies for Light HDE for GHG emission standards is 150,000 miles. See existing 40 CFR 1036.108(d).

We note that Light HDE intended for smaller vehicle classes are not designed with cylinder liners to facilitate rebuilding, suggesting they are more likely to be scrapped at the end of their operational life. The rebuilding report notes that seven percent of the diesel-fueled engines in the 2012 Class 3 vehicle population were removed from the vehicle to be rebuilt, but does not include data on the corresponding number of engines or vehicles scrapped in that year. We assume the mileage at which an engine has deteriorated enough to consider rebuilding would be similar to the mileage of an engine eligible for scrappage and both similarly represent the operational life of an engine for the purpose of this analysis.

2.4.2.2 Spark-Ignition Engine Service Life Data

The current useful life mileage that applies for GHG emission standards for Spark-ignition HDE is 150,000 miles, which is longer than the current useful life of 110,000 miles for criteria pollutant emission standards for those same engines.[M] For our updates to the useful life for Spark-ignition HDE criteria pollutant emission standards, we considered available data to represent the operational life of recent heavy-duty SI engines. A review of market literature for heavy-duty gasoline engines showed that at least one line of engine-certified products is advertised with a service life of 200,000 miles.[24] Compliance data for MY 2019 indicate that the advertised engine model represents 20 percent of the Spark-ignition HDE certified for MY 2019. Additionally, CARB's HD Omnibus rulemaking cited heavy-duty Otto-cycle engines (i.e., Spark-ignition HDE) for vehicles above 14,000 lb GVWR that were replaced during calendar years 2012 through 2018 as reaching more than 217,000 miles on average.[25] The mileages in these two examples are almost double the current useful life for Spark-ignition HDE, indicating many miles of operation beyond the current useful life.

2.4.2.3 CARB HD Omnibus Useful Life Values

The CARB HD Omnibus rulemaking, finalized in August 2020, lengthens useful life for heavy-duty CI and SI engines in two steps.[N,26] As part of their rule, CARB analyzed recent MacKay & Company survey data from calendar years 2012 through 2018 and reported rebuild mileages for CI engine categories that were similar to those described in the Chapter 2.4.2.1.

[M] See 40 CFR 1036.108(d) for the GHG useful life, and the definition of "useful life" in 40 CFR 86.004-2 for the criteria pollutant useful life.

[N] EPA is reviewing a waiver request under CAA section 209(b) from California for the Omnibus rule. See 87 FR 35765 (June 13, 2022).

CARB also included average replacement mileage information for heavy-duty Otto-cycle (HD SI) engines.[27] The CARB/MacKay & Company data is summarized in Table 2-12.

Table 2-12 Summary of CARB/MacKay & Company engine rebuild and replacement mileages for the HD Omnibus regulation [a]

Engine Class	Average Mileage at Rebuild or Replacement
HD Otto (Spark-ignition HDE) (All Vehicle Classes above 14,000 lb GVWR)	217,283
LHDD (Light HDE) (Vehicle Classes 4-5)	326,444
MHDD (Medium HDE) (Vehicle Classes 6-7)	432,652
HHDD (Heavy HDE) (Vehicle Class 8)	854,616

[a] CARB's naming conventions for HD engines differ from EPA; corresponding EPA names are noted in parentheses

Although the CARB HD Omnibus program set standards for MY 2024, the program maintained the current useful life values through MY 2026. Table 2-13 summarizes the useful life values finalized in the HD Omnibus rule for heavy-duty Otto-cycle engines (HDO), and light heavy-duty diesel (LHDD), medium heavy-duty diesel (MHDD), and heavy heavy-duty diesel (HHDD) engines.

Table 2-13 CARB useful life mileages for heavy-duty engines in the HD Omnibus rulemaking [a]

Model Year	HDO (Spark-ignition HDE)	LHDD (Light HDE)	MHDD (Medium HDE)	HHDD (Heavy HDE)[b]
2024-2026	110,000 miles 10 years	110,000 miles 10 years	185,000 miles 10 years	435,000 miles 10 years 22,000 hours
2027-2030	155,000 miles 12 years	190,000 miles 12 years	270,000 miles 11 years	600,000 miles 11 years 30,000 hours
2031 and later	200,000 miles 15 years	270,000 miles 15 years	350,000 miles 12 years	800,000 miles 12 years 40,000 hours

[a] CARB's naming conventions for HD engines differ from EPA; corresponding EPA names are noted in parentheses.
[b] CARB adopted an intermediate useful life mileage of 435,000 miles for MY 2027 and later HHDD engines.

As seen in the table, CARB's Omnibus increased useful life first in MY 2027 with a second step in MY 2031. The final useful life mileages in the CARB regulation are the result of stakeholder engagement throughout the development of CARB's HD Omnibus rulemaking. In two 2019 public workshops, CARB staff presented useful life mileage values under consideration that were longer than these final mileages and, in their September 2019 presentation, very close to the engine rebuild mileages.[28] In response to feedback from stakeholders indicating concerns with availability of data for engines and emission controls at those mileages, CARB shortened their final useful life mileages for MY 2031 and later engines from the values presented in 2019, and the MY 2027 values were chosen to be approximately the mid-point between the current and final useful life mileages.[29] Additionally, CARB finalized an intermediate useful life mileage for MY 2027 and later HHDD engines that correspond to the current useful life of 435,000 miles. Consistent with current useful life periods, CARB finalized hours values for the HHDD engine class based on the useful life mileage and an average vehicle speed of 20 miles per hour.

Similar to the useful life mileage values, CARB's useful life values in years were also adjusted from the values presented in their public workshops based on stakeholder feedback. In particular, emission controls manufacturers recommended CARB consider replacing the 18-year useful life presented in their September 2019 workshop with a useful life of 12 years for heavy-duty engines.[30] CARB agreed that 12 years was reasonable for MHDD and HHDD, but adopted a 15 year useful life for HDO and LHDD based on the useful life in years that applies to chassis-certified engines.

Chapter 2 References

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Chapter 3 Feasibility Analysis for the Final Standards

3.1 Compression-Ignition Technology Feasibility

3.1.1 Diesel Technology Demonstration Programs

3.1.1.1 CARB Heavy-duty Low NO_X Stage 3 Research Program

In 2016, the California Air Resources Board (CARB) funded the CARB Heavy-duty Low NO_X Research Program at Southwest Research Institute (SwRI) in San Antonio, TX to explore the feasibility of diesel HDEs achieving 0.02 g/bhp-hr NO_X composite emissions over the FTP transient test cycle. Stage 3 of this research program investigated the use of dual-SCR systems, cylinder deactivation (CDA), heated urea dosing system, intercooler bypass, turbine bypass, and EGR cooler bypass via engine dynamometer testing of a developmental engine based on the MY 2017 Cummins X15 diesel Heavy HDE. Major specifications for the engine are shown in Table 3-1. Like many other MY 2010 and later diesel HDEs, the X15 is equipped with a variable geometry turbocharger (VGT), high pressure common rail fuel injection and cooled EGR. The X15's original equipment exhaust aftertreatment system (EAS) consists of a DOC, DPF, SCR, and ASC in series. The X15 engine was modified by SwRI and Eaton to incorporate individual cylinder deactivation. SwRI also developed an engine calibration to aid catalyst warmup using a combination of later combustion phasing and increased idle speed. Further details of the specific, fixed CDA modes evaluated and other details regarding CDA development and engine instrumentation can be found in two papers by SwRI and Eaton.[1,2] Details regarding the EAS, control systems, and calibration are summarized in three additional papers by SwRI.[3,4,5] The complete summary of the work completed as part of Stage 3 is included in a final report from SwRI to CARB.[6]

Table 3-1: Major engine specifications for the MY 2017 Cummins X15 engine used for the CARB Low NO_X Stage 3 Research Program

Engine Displacement	14.95 L
Bore X Stroke	137 X 169 mm
Rated Power @ Speed	373 kW @ 1800 rpm
Rated Torque @ Speed	2500 N·m @ 1000 rpm

Figure 3-1: Developmental Cummins X15 Engine equipped with individual cylinder deactivation undergoing engine dynamometer testing as part of the CARB Stage 3 research at SwRI.

A schematic layout of the developmental EAS with light-off SCR is shown in Figure 3-2. Photos of the EAS showing details of its installation within the engine dynamometer test cell are shown in Figure 3-3. The heated urea dosing system, mixer, and light-off SCR with ASC zone-coated onto the rear of the SCR substrate were mounted downstream of the turbocharger. The remaining components were mounted in an insulated single-box housing to improve heat retention. Catalyst substrate specifications are summarized in Table 3-2. The total EAS volume was approximately 4.6 times the engine displacement. The total volume of the SCR, not including ASC, was approximately 2.8 times engine displacement. The volume of the light-off SCR was approximately 0.58 times engine displacement and the volume of the downstream SCR system was approximately 2.2 times engine displacement (both excluding ASC). The volume of the downstream SCR is comparable to the sales-weighted displacement-specific SCR volume for MY 2019 Heavy HDE, so the increase in total SCR volume relative to MY 2019 Heavy HDE applications was due to the addition of light-off SCR.

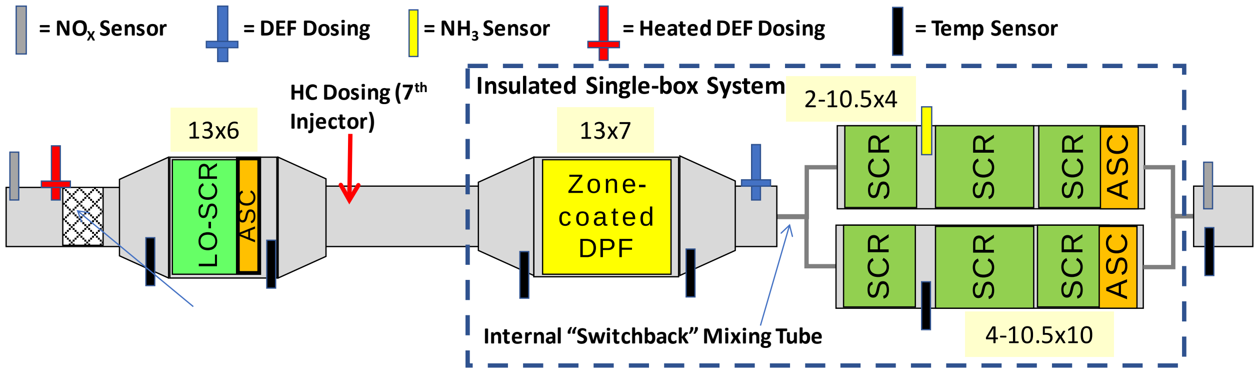

Figure 3-2: Schematic layout (not to scale) of the dual-SCR EAS tested as part of the CARB Stage 3 research at SwRI.

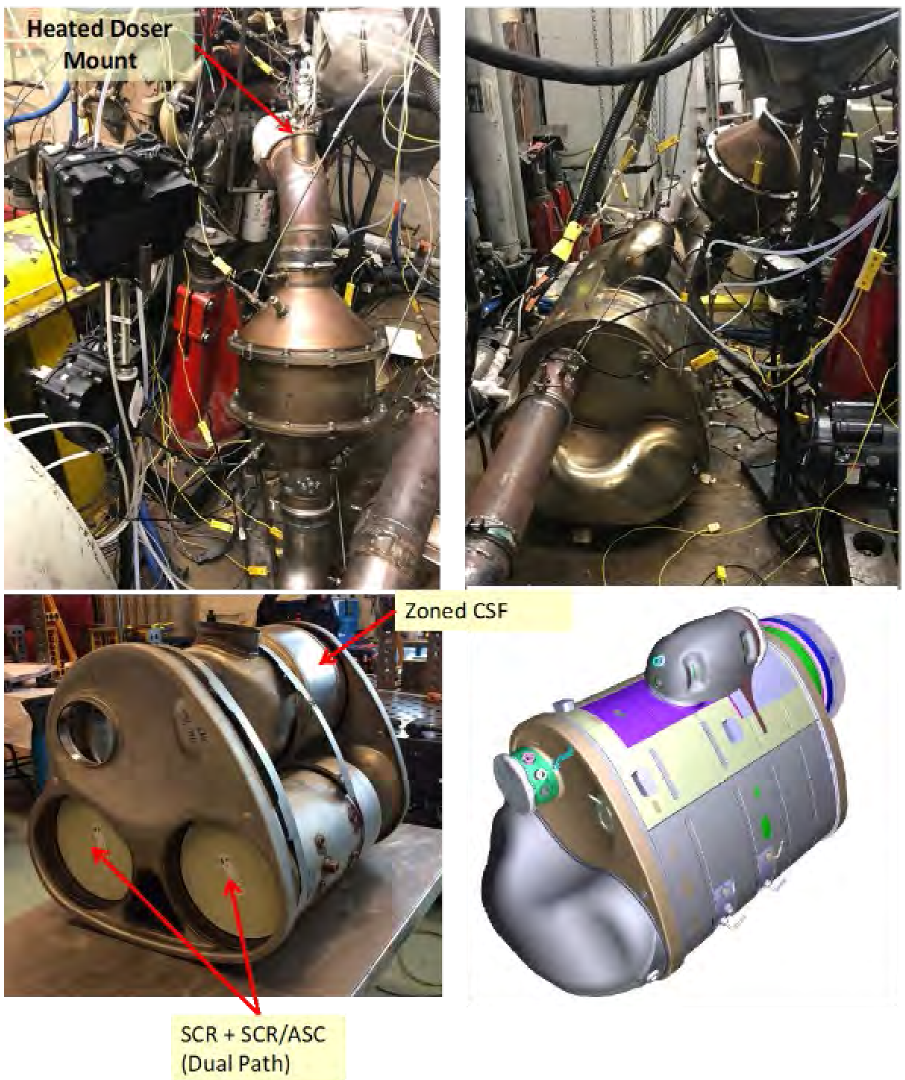

Figure 3-3: Developmental EAS with light-off SCR installed in engine dynamometer test cell at SwRI (upper left, upper right) and details of the downstream, single "box" unit (lower left, lower right).

Table 3-2: Summary of catalyst specifications for developmental EAS with light-off SCR.

Component	Dimensions, Dia. X Length (inches)	Substrate Volume (liters)	Cell Density (cpsi) / wall thickness (mil)	Notes
Light-off SCR/ASC	13 X 6	13.1	400/4	ASC is zone-coated to the rearmost 2" of the light-off SCR substrate
Zone-coated CDPF	13 X 7	15.2	300/7	Zone-coated wall-flow substrate providing both DOC and CDPF functionality
SCR	10.5 X 4	11.4	600/4.5	Two substrates in parallel
SCR	10.5 X 5	14.2	600/4.5	Two substrates in parallel
SCR/ASC	10.5 X 5	14.2	600/4.5	Two substrates in parallel, both with ASC zone-coated to the rearmost 2" of the SCR substrates

Emissions were evaluated using engine dynamometer testing over the cold-start and hot-start FTP transient cycle, the SET, the LLC, and over specific cycles representing real world operation that were provided by the Engine Manufacturers Association (EMA). A baseline, original equipment MY 2017 Cummins X15 EAS was tested at a low-hour condition with the EAS in a degreened (broken-in) state. The developmental EAS with light-off SCR was tested in a degreened state and then was subjected to accelerated aging using the Diesel Aftertreatment Accelerated Aging Cycle (DAAAC).[7] The DAAAC incorporated chemical deSOx at 30-hour intervals and DPF ash maintenance at 500-hour intervals. Emissions results over the FTP, SET, and LLC for the baseline and developmental EAS are summarized in Table 3-3 through Table 3-7. FTP Composite and SET NO_X emissions results were just over 20 mg/bhp-hr after accelerated aging equivalent to approximately 435,000 miles. The NO_X emissions results over the LLC were just under 50 mg/bhp-hr. Emissions of N_2O were approximately half that of the current 0.10 g/bhp-hr standards. The infrequent regeneration factor (IRAF) for this engine and EAS configuration was 2 mg/hp-hr for the FTP and SET and 5 mg/hp-hr for the LLC.

Table 3-3: Baseline (degreened) emissions results for the OE Cummins EAS. Results do not include adjustments for IRAF.

Cycle Results:	FTP cold	95% CI	FTP hot	95% CI	FTP composite	95% CI	SET	95% CI	LLC	95% CI
NO_X (mg/bhp-hr)	271	31	132	99	152	87	140	6	1005	335
PM (mg/bhp-hr)	2.0	0.2	2.0	1.0	2.0	0.9	1.2	0.3	NA	NA
NMHC (mg/bhp-hr)	3	7	2	2	3	3	1.7	0.2	12	25
CO (mg/bhp-hr)	48	37	17	31	22	29	7.9	0.9	30	24
CO_2 (g/bhp-hr)	530	4	508	8	511	8	452	4	609	7
N_2O (mg/bhp-hr)	42	2	63	9	61	11	68	8	64	NA*

For results where the 95% CI is greater than the average, the results are not statistically different from zero based on a 2-sided Student's t-test at α=0.05

Table 3-4: 0-hour (degreened) emissions results for the developmental EAS system with light-off SCR. Results do not include adjustments for IRAF or crankcase emissions.

Cycle Results:	FTP cold	95% CI	FTP hot	95% CI	FTP composite	95% CI	SET	95% CI	LLC	95% CI
NO_X (mg/bhp-hr)	54.7	0.8	11	2	17	1	8.9	0.8	20	8
PM (mg/bhp-hr)	2	1	1.4	0.1	1.5	0.2	0.9	0.2	4.0	0.7
NMHC (mg/bhp-hr)	23	56	12	50	14	51	1	2	47	171
CO (mg/bhp-hr)	110	41	12	3	26	6	7	1	62	51
CO_2 (g/bhp-hr)	539	4	499	1	505	2	454	3	600	4
N_2O (mg/bhp-hr)	39	3	47	1	46	2	53	9	43	9

For results where the 95% CI is greater than the average, the results are not statistically different from zero based on a 2-sided Student's t-test at α=0.05

Table 3-5: Emissions results for the developmental EAS system with light-off SCR after 334 hours of accelerated thermal and chemical aging using the DAAAC (equivalent to approximately 145,000 miles of

operation). The SET (2021) results represent updated 40 CFR 1036.510 SET procedures. Results do not include adjustments for IRAF or crankcase emissions.

Cycle Results:	FTP cold	95% CI	FTP hot	95% CI	FTP composite	95% CI	SET	95% CI	SET (2021)	95% CI	LLC	95% CI
NO_X (mg/bhp-hr)	56	5	12	2	18	2	15	2	15	2	22	9
PM (mg/bhp-hr)	1.3	0.2	1.6	0.5	1.6	0.4	0.7	0.0	0.7	0.4	3	6
NMHC (mg/bhp-hr)	33	58	18	42	20	44	10	15	3.0	0.6	19	18
CO (mg/bhp-hr)	186	94	25	10	48	22	7	1	8	2	104	45
CO_2 (g/bhp-hr)	541	4	506	5	511	5	454	1	450	2	602	2
N_2O (mg/bhp-hr)	31	4	37	4	36	4	34	2	34	1	32	17

For results where the 95% CI is greater than the average, the results are not statistically different from zero based on a 2-sided Student's t-test at α=0.05

Table 3-6: Emissions results for the developmental EAS system with light-off SCR after 667 hours of accelerated thermal and chemical aging using the DAAAC (equivalent to approximately 290,000 miles of operation). The SET (2021) results represent updated 40 CFR 1036.510 SET procedures. Results do not include adjustments for IRAF or crankcase emissions.

Cycle Results:	FTP cold	95% CI	FTP hot	95% CI	FTP composite	95% CI	SET	95% CI	SET (2021)	95% CI	LLC	95% CI
NO_X (mg/bhp-hr)	63	6	15	2	22	3	19	2	14.8	0.4	50	6
PM (mg/bhp-hr)	1.8	0.4	2.0	1.3	2.0	1.0	1	2	0.8	0.3	4.5	0.5
NMHC (mg/bhp-hr)	29	12	12	5	14	3	3	2	2.1	0.7	43	5
CO (mg/bhp-hr)	227	40	139	25	151	17	10	2	11	3	305	42
CO_2 (g/bhp-hr)	538	1	512	2	515	2	461	2	454.1	0.7	616	1
N_2O (mg/bhp-hr)	28	9	32	7	31	7	23	3	25	1	21	6

Table 3-7: Emissions results for the developmental EAS system with light-off SCR after 1000 hours of accelerated thermal and chemical aging using the DAAAC (equivalent to approximately 435,000 miles of

operation). The SET (2021) results represent updated 40 CFR 1036.510 SET procedures. Results do not include adjustments for IRAF or crankcase emissions.

Cycle Results:	FTP cold	95% CI	FTP hot	95% CI	FTP composite	95% CI	SET	95% CI	SET (2021)	95% CI	LLC	95% CI
NO_X (mg/bhp-hr)	61	8	17	3	23	3	22	4	20	4	47	4
PM (mg/bhp-hr)	2	0	2	1	2	1	2	0	1	0	6	1
NMHC (mg/bhp-hr)	41	10	24	13	26	12	5	5	4	0	28	77
CO (mg/bhp-hr)	257	44	184	38	194	34	12	1	11	1	371	79
CO_2 (g/bhp-hr)	535	9	512	1	515	2	461	6	456	2	617	1
N_2O (mg/bhp-hr)	37	7	42	21	41	18	23	2	23	1	18	12

For results where the 95% CI is greater than the average, the results are not statistically different from zero based on a 2-sided Student's t-test at α=0.05

In addition to evaluating the feasibility of the new criteria pollutant standards, we also evaluated how CO_2 was impacted using the CARB Stage 3 engine for both the test procedures used to show compliance with the engine standards in 40 CFR 1036.108 and the vehicle standards in 40 CFR part 1037, subpart B. To do this we evaluated how CO_2 emissions changed from the base engine on the FTP, SET, and LLC, as well as the fuel mapping test procedures defined in 40 CFR 1036.535 and 1036.540. For all three cycles the Stage 3 engine emitted CO_2 with no statistically significant difference at a 95% level of confidence when compared to the base MY 2017 Cummins X15 engine. Comparing the CARB Stage 3 engine using the 0-hour (degreened) EAS provided the most direct comparison with the MY 2017 Cummins X15 engine since the MY 2017 Cummins X15 original equipment (OE) EAS was degreened but not hydrothermally or chemically aged. The percent reduction in CO_2 for the FTP, SET and LLC, was 1, 0 and 1% respectively for the Stage 3 configuration relative to the OE configuration. Because SwRI made changes to the thermal management strategies of the CARB Stage 3 engine (which increased CO_2 emissions from the engine) to improve NO_X reduction at low SCR temperatures after these initial data were taken, there is no direct comparison between the baseline engine and the CARB Stage 3 engine. For the data at an equivalent of 435,000 miles that include these changes, the percent increase in CO_2 for the FTP, SET and LLC, was 0.6, 0.7 and 1.3% respectively, but since the aftertreatment had been aged to an equivalent of 435,000 miles prior to these tests, which included ash exposure from this aging and which thus increased the back pressure on the engine (Figure 3-4), this was not a direct comparison with the baseline engine. To evaluate impacts to CO_2 emissions of the CARB Stage 3 engine on the HD GHG Phase 2 test procedure, the test procedures were executed with both the baseline engine and the CARB Stage 3 engine with development aged aftertreatment. The fuel maps from these tests were used as inputs for GEM simulations. The results from this analysis (summarized in the SwRI Stage 3 report)[6] also showed that the CARB Stage 3 engine emitted CO_2 at approximately the same rate as the MY 2017 Cummins X15.

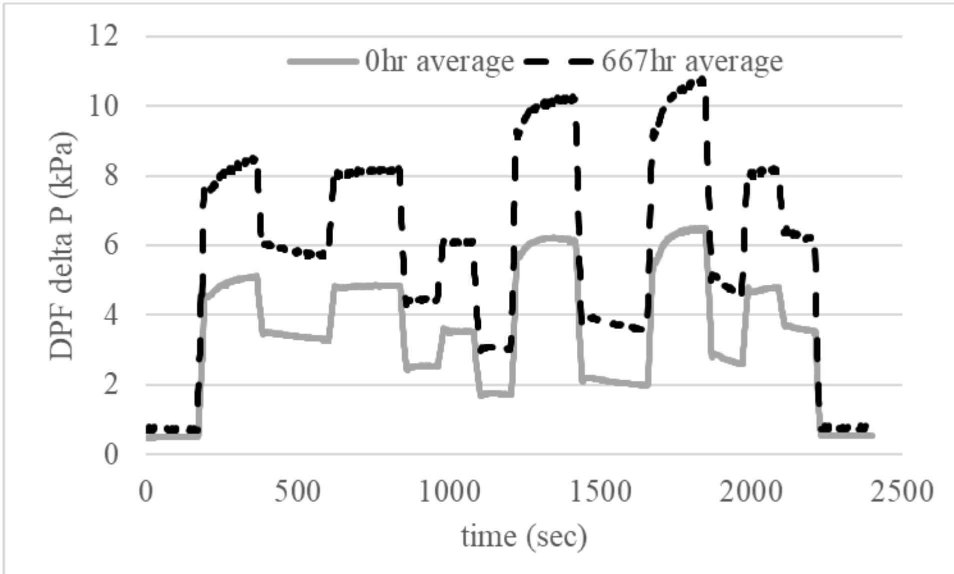

Figure 3-4: The average pressure drop across the DPF on the SET for the degreened aftertreatment and the equivalent of 290,000 miles of operation aftertreatment

3.1.1.2 EPA Stage 3 Demonstration

Once the CARB Stage 3 Demonstration was completed, the work was continued into a second phase of the demonstration by EPA and is referred to as the EPA Stage 3 Demonstration. During the EPA Stage 3 Demonstration, improvements were made to the aftertreatment by replacing the zone-coated catalyzed soot filter with a separate 13-inch diameter by 4-inch length DOC and 13-inch diameter by 7-inch length DPF. These components were separately aged via the DAAAC to the equivalent of 435,000 miles prior to their integration into the EPA Stage 3 EAS. Changes were also made to the downstream "One-box" system to further improve urea mixing and distribution. The entire system (LO-SCR, DOC, DPF, SCR, and SCR/ASC) was then aged over the DAAAC to the equivalent of 800,000 miles. A schematic of the aftertreatment is shown in Figure 3-5. The results of testing the EPA Stage 3 engine at the equivalent of 435,000, 600,000 and 800,000 miles are shown in Table 3-8, Table 3-9 and Table 3-10. The testing of the EPA Stage 3 engine also included testing with the crankcase vent not connected to the CVS tunnel to determine the crankcase emissions. The results from these tests showed that the NO_x emissions from the EPA Stage 3 engine are at a minimum 6 mg/hp-hr for the FTP, SET, and LLC cycles. By closing the crankcase, the NO_x emissions from each of these duty cycles would be reduced to 0 mg/hp-hr. For the feasibility assessment of the standards, we included the emissions reductions from closing the crankcase. The complete summary of the work completed is included in a final report from SwRI.[8]

For our feasibility analysis the 435,000 miles test point was used for assessing the Light HDE standards since the final useful life for Light HDE is below 435,000 miles, and because we believe the 435,000 miles test point adequately represents the deterioration of Light HDE to its final useful life. The interpolated emissions performance at 650,000 miles was used for Medium and Heavy HDEs because we believe that test point adequately reflects the deterioration of these engines to each final useful life mileage (350,000 miles and 650,000 miles, respectively). While the final useful life for Medium HDE is fewer miles than this test point, EPA considered comments with supporting data that showed, due to the real-world operation of these engines, that Medium HDE experience more hydrothermal aging than a Heavy HDE at the final useful life values of 350,000 miles for Medium HDE and 650,000 miles for Heavy HDE (up to 4x). EPA also considered comments with supporting data that showed, regarding Heavy HDE real world operation, that the aftertreatment of Medium HDE experience approximately 1/3 less chemical poisoning than Heavy HDE do at the final useful life values of 350,000 miles and 650,000 miles respectively. Thus, the magnitude of these two aging mechanisms (hydrothermal and chemical poisoning) is different for Medium HDE compared to Heavy HDE. When considering the aging carried out on the EPA Stage 3 engine, in our assessment, the greater real-world aftertreatment hydrothermal aging that Medium HDEs are exposed to when compared to Heavy HDE is addressed by the additional chemical poisoning the aftertreatment of the EPA Stage 3 engine is exposed to during aging out to 650,000 miles. The real-world data from Medium and Heavy HDEs supports the assessment that the EPA Stage 3 data at the equivalent of 650,000 miles is the appropriate data to be used when determining the feasibility of the Medium HDE standards at the final useful life value of 350,000 miles. The hydrothermal and chemical poisoning of Medium HDEs versus Heavy HDEs was provided as a late comment on the proposal, with information claimed as CBI comment. The late comment is included in the docket, though the information claimed as CBI is not publicly available.

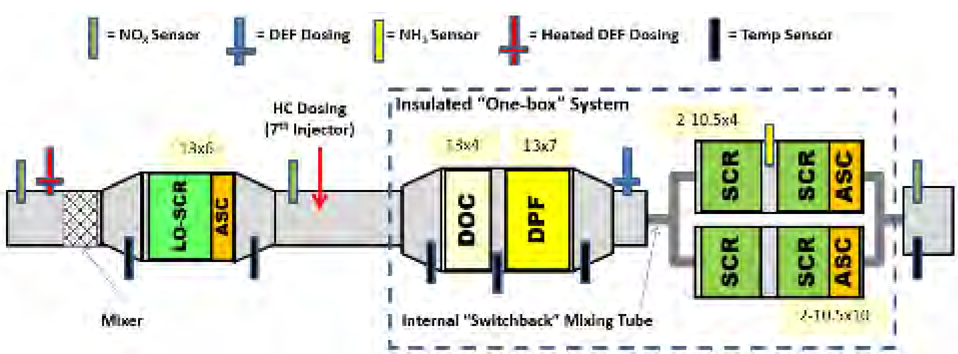

Figure 3-5: Schematic layout (not to scale) of the dual-SCR EAS tested as part of the EPA Stage 3 research at SwRI.

Table 3-8: Emissions results for the developmental EPA Stage 3 EAS system with light-off SCR and separate DOC and DPF after 1000 hours of accelerated thermal and chemical aging using the DAAAC (equivalent to

approximately 435,000 miles of operation). The SET (2021) results represent updated 40 CFR 1036.510 SET procedures. Results do not include adjustments for IRAF or crankcase emissions.

Cycle Results:	FTP cold	95% CI	FTP hot	95% CI	FTP composite	95% CI	SET (2021)	95% CI	LLC	95% CI
NO_X (mg/bhp-hr)	55	1	14	1	20	1	17	1	29	11
PM (mg/bhp-hr)	2	1	2	1	2	1	1	1	3	1
NMHC (mg/bhp-hr)	25	7	9	2	12	2	1	1	35	51
CO (mg/bhp-hr)	221	61	128	77	141	75	30	22	245	438
CO_2 (g/bhp-hr)	534	1	511	2	514	2	455	4	617	11
N_2O (mg/bhp-hr)	84	7	74	9	76	9	24	69	132	45

For results where the 95% CI is greater than the average, the results are not statistically different from zero based on a 2-sided Student's t-test at $\alpha=0.05$

Table 3-9: Emissions results for the developmental EPA Stage 3 EAS system with light-off SCR and separate DOC and DPF after 1379 hours of accelerated thermal and chemical aging using the DAAAC (equivalent to approximately 600,000 miles of operation). The SET (2021) results represent updated 40 CFR 1036.510 SET procedures. Results do not include adjustments for IRAF or crankcase emissions.

Cycle Results:	FTP cold	95% CI	FTP hot	95% CI	FTP composite	95% CI	SET (2021)	95% CI	LLC	95% CI
NO_X (mg/bhp-hr)	61	5	21	2	27	2	24	1	33	2
PM (mg/bhp-hr)	2	0	1	2	1	2	1	0	4	1
NMHC (mg/bhp-hr)	23	11	7	4	9	5	1	0	16	6
CO (mg/bhp-hr)	245	31	127	134	144	119	15	0	153	20
CO_2 (g/bhp-hr)	546	3	515	2	519	2	460	1	623	6
N_2O (mg/bhp-hr)	69	9	57	4	58	4	30	6	64	22

For results where the 95% CI is greater than the average, the results are not statistically different from zero based on a 2-sided Student's t-test at $\alpha=0.05$

Table 3-10: Emissions results for the developmental EPA Stage 3 EAS system with light-off SCR and separate DOC and DPF after 1839 hours of accelerated thermal and chemical aging using the DAAAC (equivalent to approximately 800,000 miles of operation) before ash cleaning of the DPF. The SET (2021) results represent

updated 40 CFR 1036.510 SET procedures. Results do not include adjustments for IRAF or crankcase emissions.

Cycle Results:	FTP cold	95% CI	FTP hot	95% CI	FTP composite	95% CI	SET (2021)	95% CI	LLC	95% CI
NO_X (mg/bhp-hr)	74	3	32	2	38	2	28	1	28	4
PM (mg/bhp-hr)	1	0	1	1	1	1	1	0	3	1
NMHC (mg/bhp-hr)	34	12	14	2	17	3	11	5	49	13
CO (mg/bhp-hr)	259	54	143	25	160	14	18	1	215	71
CO_2 (g/bhp-hr)	540	10	514	7	518	7	457	6	620	19
N_2O (mg/bhp-hr)	118	13	88	5	93	6	34	3	126	8

For results where the 95% CI is greater than the average, the results are not statistically different from zero based on a 2-sided Student's t-test at $\alpha=0.05$

Table 3-11: Emissions results for the developmental EPA Stage 3 EAS system with light-off SCR and separate DOC and DPF after 1839 hours of accelerated thermal and chemical aging using the DAAAC (equivalent to approximately 800,000 miles of operation) after ash cleaning of the DPF. The SET (2021) results represent updated 40 CFR 1036.510 SET procedures. Results do not include adjustments for IRAF or crankcase emissions.

Cycle Results:	FTP cold	95% CI	FTP hot	95% CI	FTP composite	95% CI	SET (2021)	95% CI	LLC	95% CI
NO_X (mg/bhp-hr)	73	12	31	3	37	1	30	0	34	8
PM (mg/bhp-hr)	1	1	1	1	1	1	2	1	1	4
NMHC (mg/bhp-hr)	32	16	11	6	14	7	1	0	40	20
CO (mg/bhp-hr)	260	68	130	73	149	68	23	7	205	40
CO_2 (g/bhp-hr)	544	2	516	4	520	4	458	0	629	2
N_2O (mg/bhp-hr)	99	59	91	45	92	47	28	4	125	17

For results where the 95% CI is greater than the average, the results are not statistically different from zero based on a 2-sided Student's t-test at $\alpha=0.05$

3.1.1.2.1 *EPA Stage 3 Off-cycle Emissions Performance*

In addition to the FTP, SET and LLC, the EPA Stage 3 engine (with the DAAAC aged aftertreatment to an equivalent of 435,000 miles) was run on 5 cycles that that cover a range of off-cycle operation. These cycles are the CARB Southern Route Cycle, Grocery Delivery Truck Cycle, Drayage Truck Cycle, Euro-VI ISC Cycle and the ACES 4-hour Cycle. The CARB Southern Route Cycle is predominantly highway operation with elevation changes resulting in extended motoring sections followed by high power operation. The Grocery Delivery Truck Cycle represents goods delivery from regional warehouses to downtown and suburban supermarkets and extended engine-off events characteristic of unloading events at supermarkets. The Drayage Truck Cycle includes near dock and local operation of drayage trucks, with extended idle and creep operation. The Euro-VI ISC Cycle is modeled after Euro VI ISC route requirements with a mix of 30% urban, 25% rural and 45% highway operation. The ACES 4-hour Cycle includes a 5 mode cycle developed as part of the ACES program. Figure 3-6 through Figure 3-10 show the engine speed, engine torque and vehicle speed of the cycles. The engine speed and torque shown in the plots are specific to the EPA Stage 3 engine.

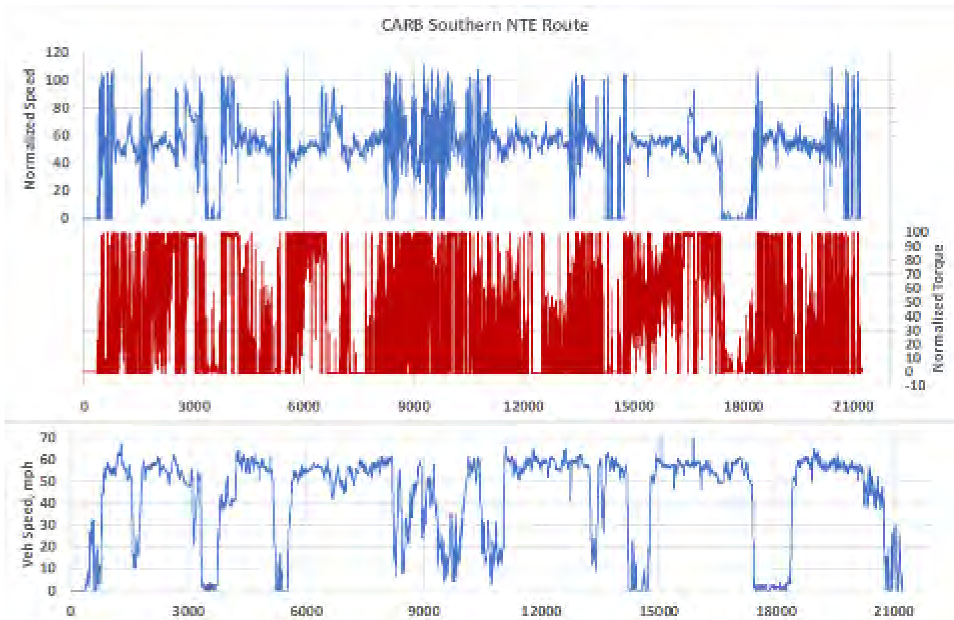

Figure 3-6: CARB Southern Route Cycle

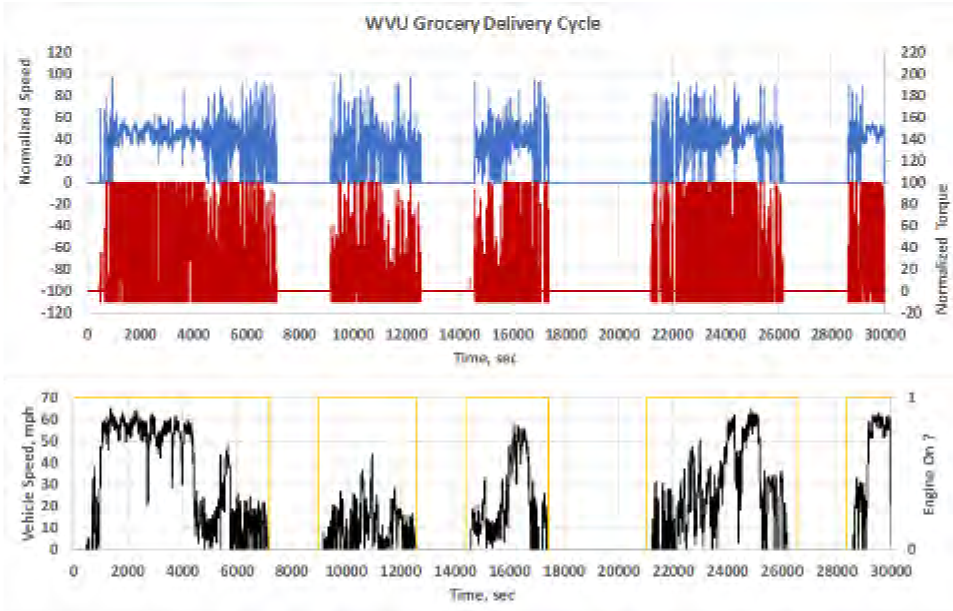

Figure 3-7: Grocery Delivery Truck Cycle

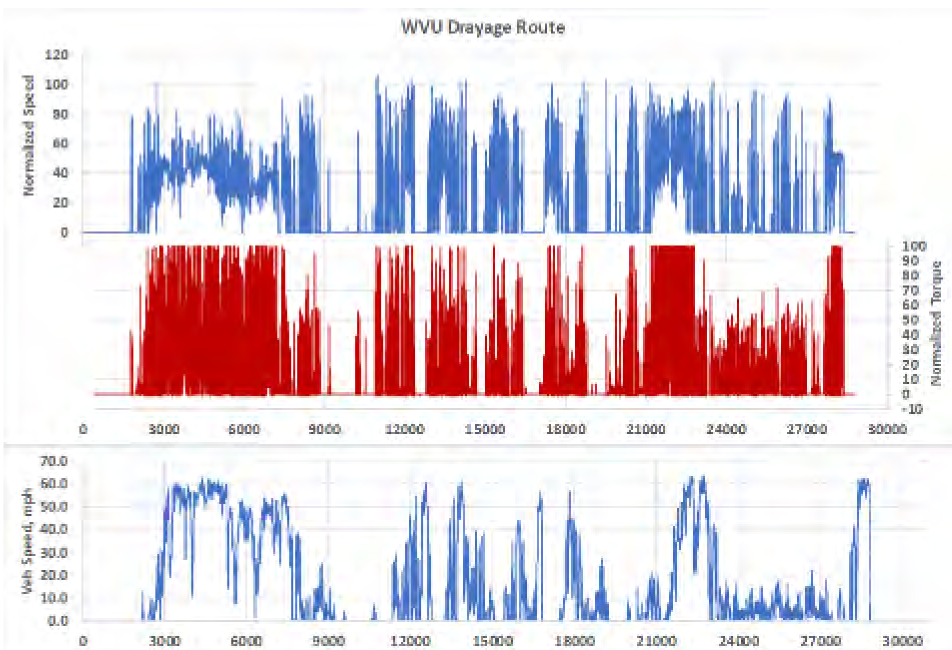

Figure 3-8: Drayage Truck Cycle

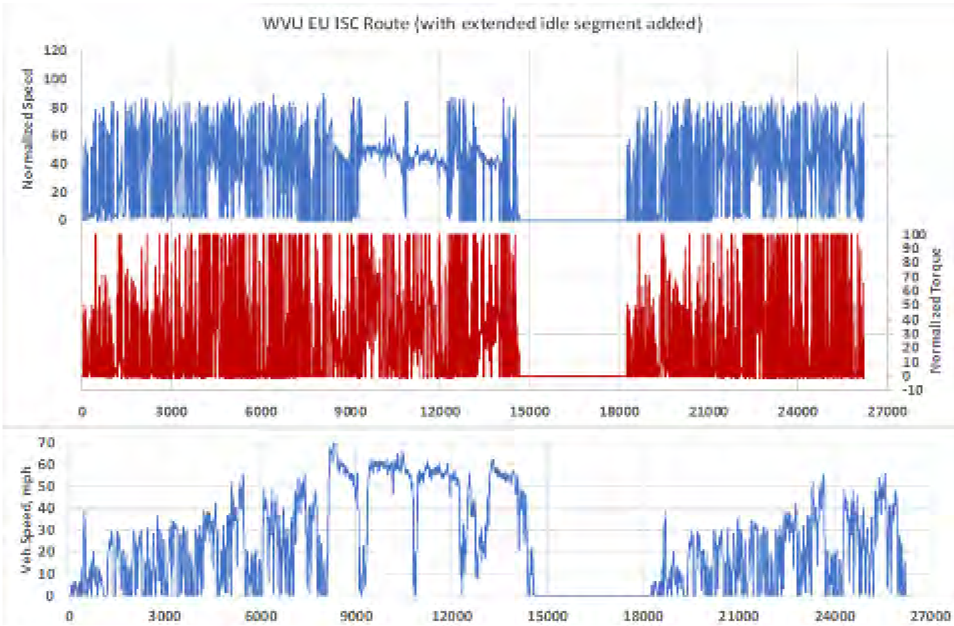

Figure 3-9: Euro-VI ISC Cycle

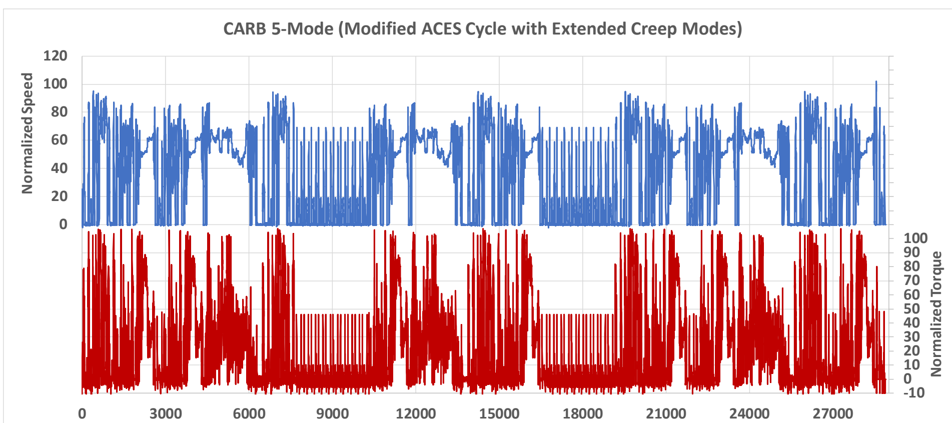

Figure 3-10: ACES 4-hour Cycle

The NO_X emissions from the EPA Stage 3 engine with aftertreatment aged to the equivalent of 435,000 miles in Table 3-12 are below the finalized off-cycle standards for Light HDE full useful life standards, with margin. The margins to the NOx standards after accounting for the emissions reductions from closed crankcase (estimated at 6 mg/hp-hr) are greater than 90% and 55%, for Bin 1 and Bin 2, respectively. Table 3-13 and Table 3-14 show that the NHMC and CO emissions from this engine are well below the finalized NMHC and CO standards for Light, Medium, and Heavy HDEs. With the aftertreatment aged to the equivalent of 800,000 miles, the engine was tested over the same five real world cycles. The NO_X emissions (shown in Table 3-15) from the EPA Stage 3 engine for each of the cycles are below the finalized Medium and Heavy HDE full useful life standards (plus in-use allowance), with margin. The margins to the

NOx standards after accounting for the emissions reductions from closed crankcase (estimated at 6 mg/hp-hr) are greater than 90% and 43%, for Bin 1 and Bin 2, respectively.

Table 3-12: Off-cycle NO_X emissions results for the developmental EAS system with light-off SCR and separate DOC and DPF after 1000 hours of accelerated thermal and chemical aging using the DAAAC (equivalent to approximately 435,000 miles of operation) without adjustments for IRAF or crankcase emissions

Bin	SNTE	Grocery Cycle	ACES	EU ISC	Drayage
1 (g/hr)	0.7	1.0	0.9	0.4	0.3
2 (mg/hp-hr)	32	21	20	31	19

Table 3-13: Off-cycle NMHC emissions results for the developmental EAS system with light-off SCR and separate DOC and DPF after 1000 hours of accelerated thermal and chemical aging using the DAAAC (equivalent to approximately 435,000 miles of operation) without adjustments for IRAF or crankcase emissions

Bin	SNTE	Grocery Cycle	ACES	EU ISC	Drayage
2 (mg/hp-hr)	1	19	0	2	19

Table 3-14: Off-cycle CO emissions results for the developmental EAS system with light-off SCR and separate DOC and DPF after 1000 hours of accelerated thermal and chemical aging using the DAAAC (equivalent to approximately 435,000 miles of operation) without adjustments for IRAF or crankcase emissions

Bin	SNTE	Grocery Cycle	ACES	EU ISC	Drayage
2 (mg/hp-hr)	16	122	31	47	261

Table 3-15 Off-cycle NO_X emissions results for the developmental EAS system with light-off SCR and separate DOC and DPF after 1839 hours of accelerated thermal and chemical aging using the DAAAC (equivalent to approximately 800,000 miles of operation) without adjustments for IRAF or crankcase emissions

Bin	SNTE	Grocery Cycle	ACES	EU ISC	Drayage
1 (g/hr)	0.7	3.3	1.5	0.4	1.1
2 (mg/hp-hr)	47	32	34	32	28

Table 3-16 Off-cycle NMHC emissions results for the developmental EAS system with light-off SCR and separate DOC and DPF after 1839 hours of accelerated thermal and chemical aging using the DAAAC

(equivalent to approximately 800,000 miles of operation) without adjustments for IRAF or crankcase emissions

Bin	SNTE	Grocery Cycle	ACES	EU ISC	Drayage
2 (mg/hp-hr)	4	30	12	20	30

Table 3-17 Off-cycle CO emissions results for the developmental EAS system with light-off SCR and separate DOC and DPF after 1839 hours of accelerated thermal and chemical aging using the DAAAC (equivalent to approximately 800,000 miles of operation) without adjustments for IRAF or crankcase emissions

Bin	SNTE	Grocery Cycle	ACES	EU ISC	Drayage
2 (mg/hp-hr)	42	300	123	108	334

3.1.1.2.2 *EPA Stage 3 Idle Emissions Performance*

To evaluate the idle NO_x emissions performance of the EPA Stage 3 engine, we directed our contractor to test the engine in three different configurations over the Clean Idle test procedure at an ambient temperature range of 20 to 30 °C. The first configuration tested was the EPA Stage 3 calibration. This calibration requires an EGR cooler bypass to be able to stay at this operational mode for an extended period of time. The second configuration tested simulated the original idle calibration that the engine was certified with, which wouldn't require the use of an EGR cooler bypass at ambient temperatures greater than 0 °C. The original idle calibration was simulated by matching the HC and soot levels during idle operation of the Stage 3 engine which includes CDA to the original Cummins X15 engine. Under this calibration, the engine could operate at idle without an EGR cooler bypass (even for extended periods of time) until the EGR had to be shut off, for example, due to low ambient temperature. The third configuration tested represented the second configuration but was operated with the EGR shut off at idle to approximate the emissions at idle of the second configuration under low ambient conditions. The results from these tests are in Table 3-18. The data from this testing shows that NO_x emission can be controlled below 10 g/hr under all conditions above 0 °C, even without the use of an EGR cooler bypass.

Table 3-18 Idle NO_x Emission Results from EPA Stage 3 Engine with EAS aged to the Equivalent of 800,000 miles

	Mode 1 NO_x (g/hr)		Mode 2 NO_x (g/hr)	
	Engine Out	Tailpipe	Engine Out	Tailpipe
1st configuration: Normal EPA Stage 3 calibration	2.5	0.13	96	0.4
2nd configuration: EGR rates that wouldn't require EGR cooler bypass when ambient temperature is greater than 0 °C	9.5	0.33	104	0.48
3rd configuration: No EGR	86	10.3	229	0.35

3.1.1.2.3 *Description of the In-use Testing Allowance*

The following section describes how we determined the in-use testing allowance for Medium and Heavy HDEs.

The basic methodology used leverages the methods outlined in the Guide to Expression of Uncertainty in Measurement (GUM). The basic approach is to describe the distribution of each term, where each term might have a different shape depending on the input data. The details for each input are described below. In the case of errors that have a bias, the total variance is generally developed as the variance plus the bias squared. For the tool[A] used, the bias was incorporated as a separate term from the variance (so two lines are used) where the full range of the bias was input, and a uniform distribution was chosen (because the bias on a given measurement could uniformly fall anywhere within that range). Rect X 2 was chosen in the tool to account for the bias-squared relationship discussed above.

Description of Inputs

1. Sulfation. This term represents influence of sulfation between $deSO_X$ events. This term incorporates a bias term because this impact will only serve to increase emissions. The variance was determined based on test results conducted on the Stage 3 engine after 150 hours of equivalent sulfation, prior to the LO-SCR $deSO_X$, which is the $deSO_X$ frequency set for the strategy. It should be noted that the downstream $deSO_X$ occurs more frequently as part of DPF regeneration every 100 hours (or less for lower load duty cycles). The results indicated an increase of 0.003 g/hp-hr in tailpipe emissions at the end of the sulfation interval. Testing could be expected to happen with equal probability at any point during the interval, therefore a rectangular distribution was chosen. A second bias term was also entered (one Line 8) as Rect X 2 to account for the bias, since the range is actually from 0 to + 0.003 g/hp-hr. **Rectangular at 0.003 g/hp-hr + Bias (Rect x2) at 0.003 g/hp-hr.**

2. Fuels. This term represents the <u>short-term</u> influence of fuel variation on test results. Short-term fuel influence means an impact which is observed as soon as a new fuel has replaced the previous one in the engine fuel system. It does not include the potential long-term impact of a given fuel (if any) on the durability of the emission control system (that is accounted for later). It was anticipated that this term would be defined based on the results of testing funded by CRC, but this result was not available in time. Therefore, the result here was calculated from examining the impact of fuels on engine-out NO_X emissions that has been documented in previous studies, and assuming the same NO_X conversion efficiency observed for the certification fuel. A value of +/- 20% was used for fuel impact on engine out emissions (examples include observations of B20 at up to +15% and highly paraffinic renewable diesels fuels at -15%), but this represents the upper end of most studies and therefore this value was chosen as a 2SD value on a normal distribution, as most fuels are anticipated to have a

[A] A copy of the spreadsheet tool is in the docket for this rule.

smaller impact. The impact of this change on the final tailpipe level was 0.002 g/hp-hr. **2SD = 0.002 g/hp-hr.**

3. Sensors. This term represents an element of production variation associated with potential sensor drift and errors. It was determined based on the results of the Step 1 sensor testing that was conducted on behalf of the Truck and Engine Manufacturers Associate (EMA) using the EPA Stage 3 engine. A number of offsets to the sensors involved in the SCR control were examined, although the potential mitigating effect of long-term trim was not yet evaluated (short-term feedback trim was active however). It was observed that both positive and negative offsets were seen. The variance was determined based on the largest positive offset observed which was +0.005 g/hp-hr for an error involving a +10% offset on both the engine-out and LO-SCR out NOX sensors on the RMC-SET cycle. Given that this offset required a combination of errors to line up, this value was chosen as a 2SD level on a normal distribution as most of the sensor errors was observed to have a smaller impact. **2SD = 0.005 g/hp-hr.**

4. Production Variability (not including Sensors). This term is meant to represent the impact of other production variations, such as batch-to-batch variation in catalysts, dosing system variations (after short-term and long-term trim corrections), and production variations that might impact engine-out NO_X. We note that the variance of this term is more difficult to pin down because many of the factors influencing it are not yet well documented. These variations could lead to higher emissions or lower emissions; therefore, a normal distribution was assumed. We understand that for this particular element we could attempt to leverage current production data, but this would be a problematic approach due to the fact that the current NO_x standard would permit a wider range of production variation to be tolerated, and thus there is not currently incentive to control variation any more than is necessary. It is reasonable to assume however, that this value would likely be at least as large as the value observed for the sensor term. Based on this assumption and consideration of statements from various manufacturers, we selected a projected value of 0.006 g/hp-hr, and this was set as a 2SD value using a normal distribution. This value represents a variation of +/- 0.1 % NO_X conversion efficiency. We acknowledged that this term represents more guesswork than others but think that our approach is an appropriate projection based on our analysis. **2SD = 0.006 g/hp-hr.** We note that if, for example, 0.006 g/hp-hr represents 1SD, the final tolerance stack-up would be 3 mg/hp-hr larger.

5. Field Aging. This term represents the degree to which in-field aging might be more or less severe than the aging data that were used to develop the DAAAC aging protocol used for the EPA Stage 3 program. This could involve things such as the long-term impact of a more severe duty cycle (that might require more DPF regenerations, for example), impact of fuel impurities (as noted earlier), or the impact of other engine related issues (such as an EGR cooler leak that was too small to detect, or the impact of a short-term high temperature excursion results from a turbocharger failure prior to

fault detection, etc.). Collectively these could result in more degradation than was predicted, or less in the case of a less severe duty cycle, though the latter is considered less likely. To estimate the full range of this impact, we decided to project this by looking at a range of aftertreatment degradation from half as much as what was observed in the EPA Stage 3 program to twice as much as what was observed in the CARB Stage 3 program. This calculation results in an impact ranging from -0.003 g/hp-hr to + 0.012 g/hp-hr. This is a full range of 0.015 g/hp-hr centered on a positive bias of 0.007 g/hp-hr. It is assumed that a normal distribution would apply to this range. Therefore, a 1SD spread was assigned at +/- 0.005 g/hp-hr, and a bias term was defined at +0.012 g/hp-hr with a Rect x 2 distribution as discussed earlier.

When the tolerance stack-up calculations were completed, a final variance at 1SD of 0.00755 g/hp-hr was determined. For this kind of tolerance, it is standard practice to use a "coverage factor" of 2 to define the reasonable limits of variation, therefore a **2SD value of 0.0151** g/hp-hr was determined, which when rounded to the nearest 2 significant figures becomes **15 mg/hp-hr.**

3.1.1.3 EPA Heavy-duty Diesel Low NO_X Demonstration Program

EPA evaluated two different EAS designs provided to the Agency by the Manufacturers of Emissions Control Association (MECA). Both EAS designs incorporated LO-SCR and dual urea injection. One of the systems, System A, used close-coupling of the light-off SCR (Figure 3-11, Table 3-19). The other EAS design mounted the light-off SCR closer to the other EAS components in an under-cab position (Figure 3-12, Table 3-20).

Both EAS designs utilized conventional urea dosing systems for the downstream SCR position and were evaluated using a heated urea dosing system in the upstream SCR position. In addition, both EAS designs were tested as part of an EPA contract at SwRI using the same developmental version of a MY 2017 Cummins X15 15-liter Heavy HDE engine equipped with CDA as was used for the CARB and EPA Stage 3 Demonstrations.[8]

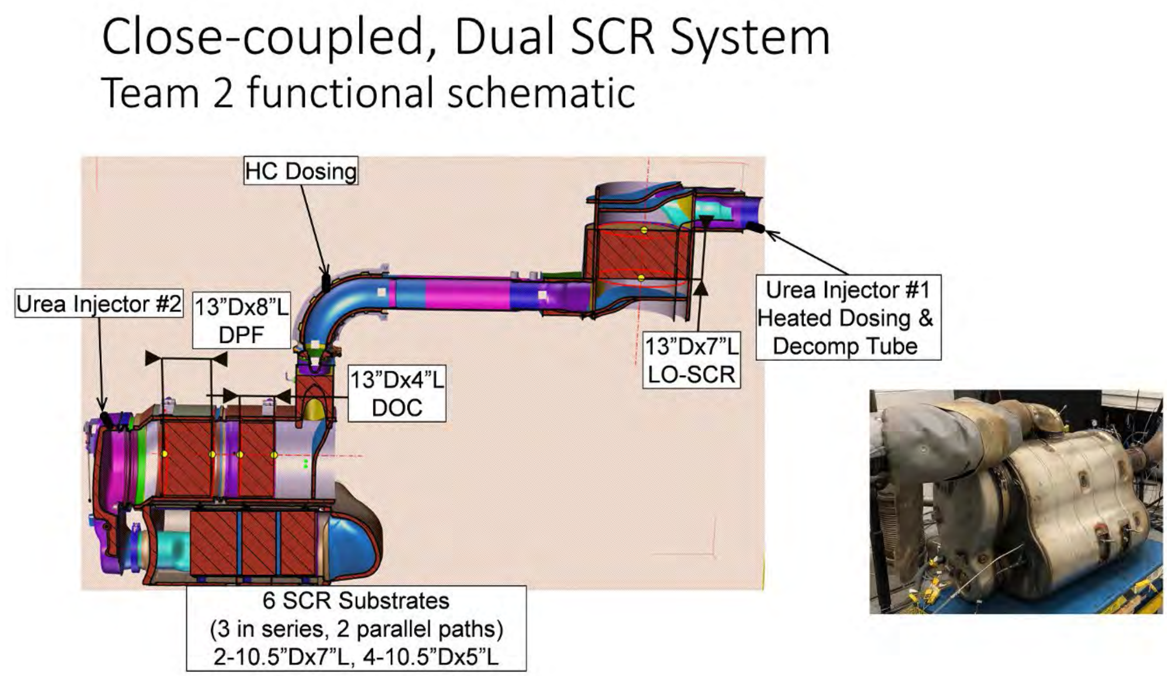

Figure 3-11: MECA "System A" EAS with close-coupled light-off SCR.

Table 3-19: Summary of catalyst specifications for developmental EAS with close-coupled light-off SCR.

Component	Substrate Dimensions, Dia. X Length (inches)	Substrate Volume (liters)	Cell Density (cpsi) / wall thickness (mil)	Notes
Light-off SCR/ASC	13 X 7	15.2	400/4	Thin wall/low-mass substrate with ASC zone-coated to the rearmost 2"
DOC	13 X 4	8.7	400/4	Thin wall/low-mass substrate
CDPF	13 X 8	17.4	300/7	
SCR	10.5 X 7	19.9	600/2	Two substrates in parallel
SCR	10.5 X 5	14.2	600/2	Two substrates in parallel
SCR/ASC	10.5 X 5	14.2	600/2	Two substrates in parallel, both with ASC zone-coated to the rearmost 2" of the SCR substrates

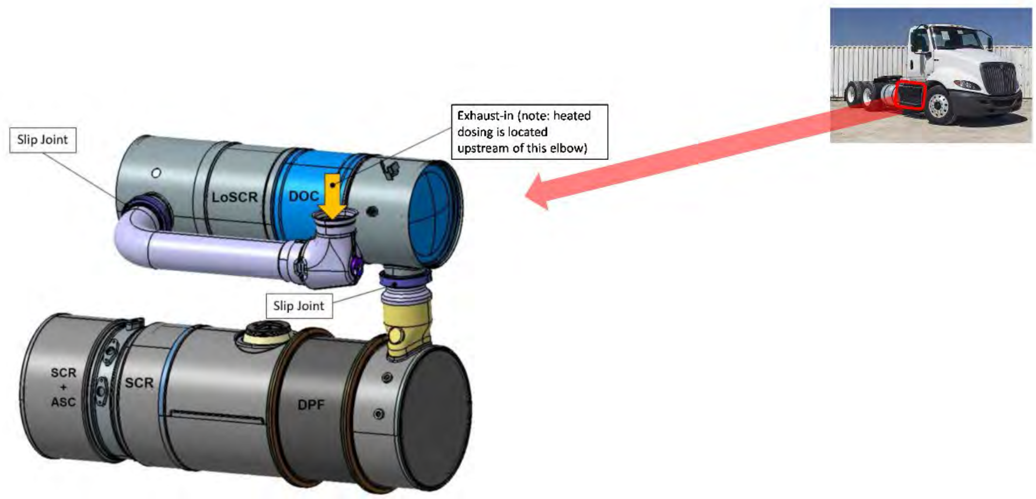

Figure 3-12: MECA "Team B" EAS with light-off SCR integrated into an under-cab mounting position. This system was designed to be installed in a Navistar Daycab which is shown in the upper right.

Table 3-20: Summary of catalyst specifications for developmental "System B" EAS with light-off SCR mounted under-cab.

Component	Substrate Dimensions, Dia. X Length (inches)	Substrate Volume (liters)	Cell Density (cpsi) / wall thickness (mil)	Notes
Light-off SCR/ASC	10.5 X 8	11.4	400/4	High porosity/low-mass substrate with ASC zone-coated to the rearmost 2"
DOC	10.5 X 6	8.5	400/4	High porosity/low-mass substrate
CDPF	13 X 7	15.2	300/7	
SCR+SCR/ASC	13 X 7	30.5	600/3	Two substrates in series - volume is for combined total. ASC is zone-coated to the rearmost 2" of SCR #3

EAS accelerated aging of System A was conducted using a "burner aging" version of the DAAAC.[7,9] This used a burner system fueled with diesel fuel and additized engine lubricant to expose an EAS to both accelerated thermal aging and accelerated chemical aging. The burner was operated over a series of controlled burner exhaust flow rates and burner exhaust temperature setpoints that matched specific engine speed and engine load setpoints during operation of the targeted engine and EAS application (see Figure 3-13). A higher sulfur diesel fuel (>100 ppm) was used during DAAAC burner aging in order to accelerate sulfur exposure.

The DAAAC aging was designed to accelerate thermal and chemical effects by approximately 10 times normal engine operation (i.e., 1 hour of DAAAC is approximately equivalent to 10 hours of actual engine operation). Operation on the DAAAC for 1,000 hours was approximately equivalent to Heavy HDE operation in a truck application to the end of UL (435,000 miles).

Emissions testing with System A was conducted using the developmental X15 engine at accelerated aging equivalents of 435,000 and 650,000 miles of operation. System B was tested using the developmental X15 engine in a “degreened” condition only. While System B demonstrated high NOx reductions over the regulatory cycles in a degreened condition, CO_2 emissions increased by 2% on the SET, due to increased exhaust backpressure compared to the baseline OE EAS. This was likely due to the decision to of use 10.5” diameter substrates for the LO-SCR and DOC within the EAS design. Emissions results for System A in a degreened condition are summarized in Table 3-21, and results after EAS DAAAC aging to an equivalent of 435,000 and 650,000 miles have been added to the docket.[10] Emissions results for System B in a degreened condition are summarized in Table 3-22.

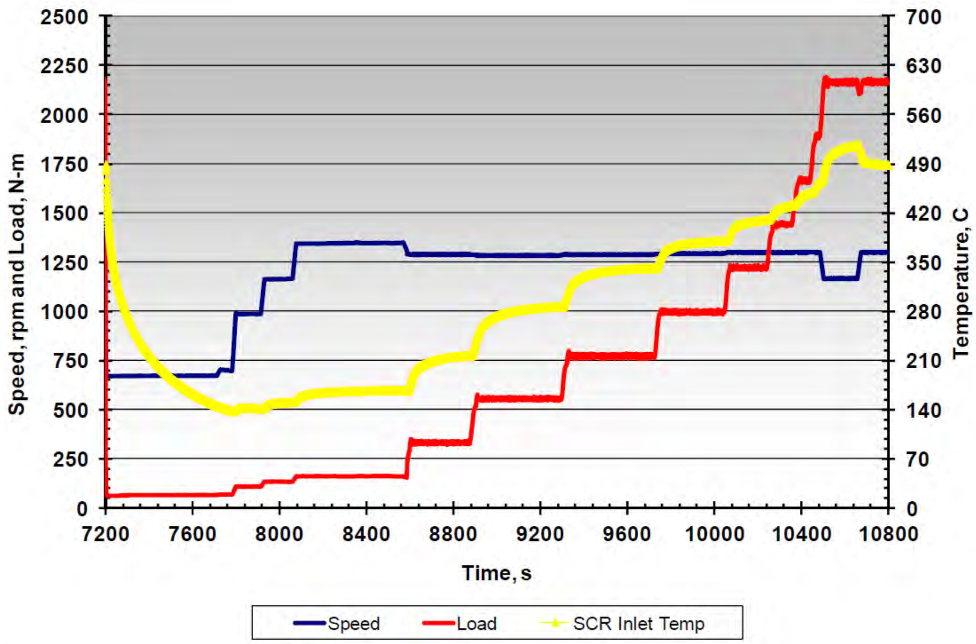

Figure 3-13: Example of engine-speed, engine load, and resulting SCR inlet temperature used over the DAAAC.

Table 3-21: Emissions results for the MECA System A with the EAS in a "degreened" (near 0-hour) condition. The SET (2021) results represent updated 40 CFR 1036.510 SET procedures.

Cycle Results:	FTP cold	95% CI	FTP hot	95% CI	FTP composite	95% CI	SET (2021)	95% CI	LLC	95% CI
NO_X (mg/bhp-hr)	52	10	10	2	16	2	9	1	13	4
PM (mg/bhp-hr)	1	1	1	0	1	0	1	0	2	0
NMHC (mg/bhp-hr)	10	15	4	1	5	2	0	1	8	2
CO (mg/bhp-hr)	135	101	32	13	47	25	5	2	67	40
CO_2 (g/bhp-hr)	544	7	512	3	517	2	454.2	0.5	627	2
N_2O (mg/bhp-hr)	26	2	29	4	28	4	23	164	35	4

For results where the 95% CI is greater than the average, the results are not statistically different from zero based on a 2-sided Student's t-test at α=0.05

Table 3-22: Emissions results for the MECA System B with the EAS in a "degreened" (near 0-hour) condition. The SET (2021) results represent updated 40 CFR 1036.510 SET procedures.

Cycle Results:	FTP cold	95% CI	FTP hot	95% CI	FTP composite	95% CI	SET (2021)	95% CI	LLC	95% CI
NO_X (mg/bhp-hr)	49	1	8	1	14	1	9	0	15	8
PM (mg/bhp-hr)	2	1	1	0	1	0	1	0	4	0
NMHC (mg/bhp-hr)	26	32	10	15	12	16	2	5	16	35
CO (mg/bhp-hr)	178	34	59	45	76	43	42	3	145	148
CO_2 (g/bhp-hr)	541	5	514	4	518	3	459	2	621	10
N_2O (mg/bhp-hr)	25	1	29	6	28	5	29	0	33	8

For results where the 95% CI is greater than the average, the results are not statistically different from zero based on a 2-sided Student's t-test at α=0.05

3.1.1.4 EPA Heavy-duty Diesel CDA Noise, Vibration, and Harshness Study

As discussion in Chapter 3.1.1, EPA has evaluated the use of CDA to lower NO_x emissions by increasing exhaust temperature without increasing CO_2 emission. As show in the results from the EPA and CARB Stage 3 demonstration significant NO_x reductions can be achieve with CDA in combination with EAS improvements. However, one of the concerns with CDA is that by reducing the number of firing cylinders can cause concerns with noise, vibration, and harshness (NVH). To evaluate this concern EPA conducted a study to evaluate the effects of CDA on NVH. This study investigated the impact of fixed CDA on vibration in an on-highway tractor and to determine the acceptable bounds of CDA operation for engine calibration. The conclusions from this study were that there are several ways to reduce NVH through design of the complete system, including, engine conditions where CDA is used, engine mounts, cab mounts, and seat calibration.[11]

3.1.2 Baseline Technology Effectiveness

The basis for our baseline technology assessment is the data provided by manufacturers as part of the heavy-duty, field testing requirements. This data encompasses real world operation from nearly 300 LHD, MHD, and HHD vehicles. Chapter 6 of the RIA describes how the data was used to update the MOVES model emissions rates for HD diesel engines. Chapter 3 of the RIA summarizes the real world emissions performance of these engines.

To assess emissions levels of current production engines on the regulatory cycles we analyzed the certification data submitted to the agency. For this analysis we focused on MY 2019 and newer engines.

Table 3-24 include the data for the certification results of the FTP and SET cycle. The certification results are the test results adjusted for IRAF and DF.

One observation from the data is the range of margin between the certification results and the standard. The margin for NO_X on the FTP cycle is as small as 0.02 g/hp-hr or 10% and as large as 0.15 g/hp-hr or 75% of the standard. For the SET the average compliance margin is slightly larger than the average margin for the FTP. For the other criteria pollutants the margin between the certification results and the applicable standards are much larger than for NO_X.

Table 3-23: Summary of certification data for FTP cycle

	NO_X (g/hp-hr)	**PM (g/hp-hr)**	**NMHC (g/hp-hr)**	**CO (g/hp-hr)**	**N2O (g/hp-hr)**
Standard	0.2	0.01	0.14	15.5	0.1
Average	0.13	0.00	0.01	0.18	0.07
Minimum	0.05	0.00	0.00	0.00	0.04
Maximum	0.18	0.00	0.04	1.10	0.11

Table 3-24: Summary of certification data for SET cycle

	NO_X (g/hp-hr)	**PM (g/hp-hr)**	**NMHC (g/hp-hr)**	**CO (g/hp-hr)**
Standard	0.2	0.01	0.14	15.5
Average	0.11	0.00	0.01	0.00
Minimum	0.00	0.00	0.00	0.00
Maximum	0.18	0.00	0.04	0.20

In addition to analyzing the off-cycle real world data submitted by manufacturers, we also conducted and analyzed engine dynamometer data of three modern HD diesel engines. These engines include a MY 2018 Cummins B6.7, MY 2018 Detroit DD15 and MY 2018 Navistar A26. These engines were tested on cycles that ranged in power demand and included the LLC and the SET cycle defined in 40 CFR 1036.510. These results are summarized in Table 3-25, Table 3-26, and Table 3-27

For two of these engines, both the SET in 40 CFR 1036.510 and 40 CFR 86.1333 were run. As can be seen from the data, there was not a measurable difference between the results of these two cycles. Both of these cycles were also run on the EPA and CARB Stage 3 engine. These results are summarized in Chapter 3.1.1.1. The LLC cycle was also run for these engines to understand the performance of current engines on this cycle. The results from this cycle vary much more than the SET and FTP. The Cummins B6.7, which is the only non-tractor engine that was tested, had the lowest NO_X at 0.35 g/hp-hr. The other two engines including the Cummins X15 shown in Table 3-3 had results that were multiple times higher than the current standards for the FTP and SET.

Table 3-25: MY 2018 Detroit DD15 engine emissions in g/hp-hr

	Cold FTP	Hot FTP	FTP Composite	SET in 40 CFR 86.1333	SET in 40 CFR 1036.510	LLC
CO2	573	550	554	481	472	642
CO	0.54	0.04	0.11	0.01	0.01	0.07
THC	0.02	0.01	0.01	0.01	0.00	0.03
NO_X	0.43	0.05	0.10	0.01	0.01	0.61
NMHC	0.01	0.01	0.01	0.01	0.00	0.02

Table 3-26: MY 2018 Cummins B6.7 engine emissions in g/hp-hr

	Cold FTP	Hot FTP	FTP Composite	SET in 40 CFR 86.1333	SET in 40 CFR 1036.510	LLC
CO2	621	569	576	486	480	908
CO	0.09	0.04	0.05	0.01	0.02	0.03
THC	0.02	0.04	0.04	0.02	0.04	0.04
NO_X	0.48	0.10	0.15	0.05	0.05	0.35
NMHC	0.02	0.04	0.04	0.02	0.04	0.03
PM	0.00	0.00	0.00	0.00	0.00	

Table 3-27: MY 2018 Navistar A26 engine emissions in g/hp-hr

	Cold FTP	Hot FTP	FTP Composite	SET in 40 CFR 86.1333	LLC
CO2	546	527	529	459	710
CO	0.03	0.02	0.02	0.01	0.03
THC	0.01	0.00	0.00	0.00	0.05
NO_X	0.37	0.10	0.14	0.12	0.81
NMHC	0.01	0.00	0.00	0.00	0.05
PM	0.00	0.00	0.00	0.00	

3.1.3 GHG Impacts

The combination of active and passive thermal management anticipated for meeting the final standards can be designed and developed in a manner that does not pose an additional burden for meeting the heavy-duty engine GHG standards in 40 CFR 1036.108 or the heavy-duty vehicles GHG standards in 40 CFR part 1037, subpart B. As described in section 3.1.1, the design and calibration of the CARB and EPA Stage 3 systems achieved significant NO_X reductions that were GHG neutral. The system design and calibration strategy of that system took advantage of

GHG improvements at light-load conditions from CDA and calibration changes at higher load conditions (e.g., injection timing) that approximately offset the impact of increased backpressure from the additional SCR catalyst volume.   The analyses presented in this and in subsequent chapters assume adoption of GHG neutral technologies similar to what was tested at SwRI.

3.1.4 **Estimated Direct Manufacturing Costs for Technology Packages Evaluated**

The final program is based on the need to obtain significant emissions reductions from the heavy-duty transportation sector, and the recognition that there are technically feasible technologies to achieve such reductions in the MY 2027 timeframe with no compromise to vehicle utility or safety. As in many prior mobile source rulemakings, the selected standard is informed by the effectiveness of the emissions control technology, the cost of that emission control technology, and the lead time needed for manufacturers to employ the control technology.

3.1.4.1 Cost Teardown Studies

Publicly available information regarding the engineering cost of new engine and vehicle technologies is a subject of considerable interest. A number of cost analyses in the past few years have utilized supplier price quotes on designated bills of materials as a methodology for estimating the increased cost of vehicle improvements. In general, the actual price quotes provided by suppliers were claimed as confidential business information and have not been released. In addition, supplier price quotes are typically provided for near-term (e.g., 3-5 years) estimation, as these are how contracts between OEMs and suppliers are typically written.

This methodology for estimating technology costs to the consumer has several deficiencies. The lack of transparency regarding the data provided by suppliers does not provide an opportunity for a full public evaluation of the information. In addition, these near-term price quotes may not be appropriate for estimating the long-term costs of a regulatory program implemented in the future. A large fraction of the near-term fixed costs may be recovered and no longer part of the costs to consumers. EPA is required to evaluate the near- and long-term costs to consumers that may result from regulatory requirements. To effectively estimate and communicate those costs, EPA requires a transparent engineering analysis that separates direct and indirect costs for each major component in the technologies it projects will be implemented to meet the new requirements.

EPA previously directed contractual work to develop an analytical methodology that is based on technical knowledge of the engineering, design, and development of advanced vehicle technology components, systems, and subsystems. In addition, the previous contractual work performed pilot studies to demonstrate the methodology on representative vehicle categories.

A key objective for these studies was transparency-- methodologies, assumptions, and inputs well-documented, clearly explained, and releasable to the public, except to the extent that those essential inputs included information claimed as confidential.

3.1.4.1.1 *EPA HDE Cylinder Deactivation and Variable Geometry Valvetrain Teardown Cost Study*

The cost of CDA for Light HDE, Medium HDE, Heavy HDE, and Urban Buses was estimated based upon a detailed, tear-down study of heavy-duty diesel valvetrains, the Heavy-

Duty Engine Valvetrain Technology Cost Assessment (or “FEV Valvetrain Study”).[12] The study was conducted by FEV North America, Inc. under a contract with EPA[B] and was submitted to an independent peer review.[13,14] The FEV Valvetrain Study investigated design modifications to a production engine cylinder head from the Cummins X15 engine. These design modifications allowed the addition of a variable-geometry valvetrain system in one of two different configurations. One configuration was implementation of individual CDA with an integrated exhaust brake. The other configuration was implementation of late intake valve closing (LIVC). The final objective was to evaluate the incremental cost of CDA and LIVC hardware as two distinct technology packages.

The cost of the CDA and LIVC technology packages were evaluated relative to a baseline valvetrain technology represented by a 2019 Cummins X15 engine. FEV also investigated other valvetrain designs used by diesel HDE in order to develop fleet average per-cylinder costs for these valvetrain technologies. The baseline and new technology packages were required to have similar overall performance with respect to service life and other functional objectives. Table 3-1 shows estimated direct manufacturing costs for application of CDA to diesel HDE based on costs derived from the FEV study. For purposes of EPA’s cost analysis of CDA applied to diesel HDE, Light HDE costs were based on application of CDA hardware to 8-cylinder engines while Medium HDE, Heavy HDE, and Urban Bus costs were based on application of CDA hardware to 6-cylinder engines. Both costs, airflow control, and thermal management appeared to roughly comparable for both CDA and LIVC within the analysis, with some advantages with respect to higher BMEP CO_2 reduction for LIVC at slightly higher cost. For more details regarding the FEV Valvetrain Study, please refer to the final report for the study within the docket for this rule. The costs for CDA were assumed as the cost for active thermal management via valvetrain system improvements within EPA’s costs for diesel HDE for the emissions standards in the final rule.

Table 3-28: Summary of CDA Direct Manufacturing Costs from Teardown Study

EPA HD Engine Class	Light HDE	Medium HDE	Heavy HDE	Urban Bus
CDA Valvetrain Hardware - Tier 1 Supplier Cost to Manufacturer (2019 $):	204.16	153.12	214.56	153.12

3.1.4.1.2 *EPA Advanced EAS Teardown Cost Study*

EPA also sponsored an additional study to examine in detail an advanced heavy-duty diesel EAS technology package utilizing a dual-SCR system with heated dosing for the light-off SCR and capable of approximately 90% NO_X reduction relative to post-2010 heavy-duty diesel emission control systems, the Heavy-Duty Vehicles Aftertreatment Systems Cost Assessment (FEV EAS Study)[15]. As with the valvetrain study, this was also conducted by FEV North America, Inc. The FEV EAS Study previously submitted to the docket for the proposal was updated slightly for the final rule to incorporate updated PGM costs.[15,16] The updated FEV EAS Study costs served as the basis for diesel EAS system costs for the final rule.

Within the FEV EAS Study, direct manufacturing costs associated with the advanced EAS technology packages were evaluated relative to a baseline OE (MY 2018) EAS technology

[B] U.S. EPA Contract No. 68HERC19D0008, Task Order No. 68HERH20F0041.

representative of the current state of design. The updated costs from the FEV EAS Study are summarized within Table 3-29. The incremental cost increase for the advanced system capable of meeting the final standards was approximately $108 to 316 higher relative to the costs used for the proposal. For details regarding the FEV EAS Study, please refer to the final report for the updated study within the docket for this rule.

Table 3-29: Summary of Direct Manufacturing Costs from the FEV Exhaust Aftertreatment System Study with PGM Costs Updated for the Final Rule

EPA HD Engine Class	Light HDE	Medium HDE	Heavy HDE	Urban Bus
Total "Baseline" MY2018 EAS Cost	$2,693.69	$2,643.39	$3,919.54	$2,723.17
Total "Advanced" EAS Cost (2019$)	$4,490.44	$4,345.84	$6,080.84	$4,459.33
Total EAS Incremental Cost from 2018 System to Final Rule (2019 $):	$1,796.75	$1,702.45	$2,161.30	$1,736.16
Difference in Incremental Cost Comparing the Final Rule to the Proposal (2019 $)	$ 316.22	$ 206.96	$ 110.13	$ 205.37

Note: Costs from the FEV Study and subsequent updates were originally calculated on a 2020$ basis and were converted to a 2019$ basis for this table to allow direct comparison with other costs within this RIA.

3.1.4.2 Closed Crankcase Systems Technology Costs

We project that manufacturers meeting the final crankcase requirement by closing the crankcase on turbocharged engines that do not have closed crankcase systems already, will rely on engineered closed crankcase ventilation systems. These systems filter oil from the blow-by gases prior to routing them into either the engine intake or the exhaust system downstream of the turbocharger but upstream of the exhaust aftertreatment system. We have estimated the initial direct manufacturing cost for manufacturers of these systems to be approximately $41 (2002$).[17] To estimate the baseline cost, we multiplied $41 (2002$) by the percentage of engines that already have closed crankcase systems, which resulted in a baseline cost of $13 (2002$). We estimated the percentage of engines that already have closed crankcase systems at 32.5%, based on the certification data. For our cost analysis, we converted these estimates to 2017 dollars, which resulted in $18 (2017$) for the baseline cost and the same cost of $37 to implement closed crankcases in the remaining CI engines for our final standards.

3.2 Spark-Ignition Technology Feasibility

3.2.1 **Baseline Technology Effectiveness**

In 2018, EPA evaluated heavy-duty gasoline Class 3 and 4 vehicles from three different manufacturers to better understand the state of criteria pollutant control technology incorporated on gasoline engines used in these applications.[18] Evaluations were conducted using laboratory chassis dynamometer testing and real-world Portable Emissions Measurement System (PEMS) testing.

Most chassis-certified heavy-duty vehicles are subject to EPA's light-duty Tier 3 program and these vehicles have adopted many of the emissions technologies from their light-duty counterparts (79 FR 23414, April 28, 2014). To meet these Tier 3 emission standards, manufacturers reduced the time needed for the catalyst to reach operational temperature by implementing cold-start calibration strategies to reduce light-off time. They have also moved the catalyst closer to the engine. Manufacturers have not widely adopted the same strategies for their heavy-duty engine-certified products. The purpose of this test program was to observe emissions performance for technologies that are available in the market today and establish a baseline to evaluate the performance of advanced technologies to further reduce criteria emissions.

3.2.1.1 Baseline Vehicles Tested for Exhaust Emissions

Three vehicles were chosen for evaluation based on majority market share. Two vehicles, Class 4 (GVWR 14,001 – 16,000 pounds), with powertrains produce by General Motors (GM) and Ford respectively, utilized engines that were dynamometer certified. The third vehicle, produced by Fiat Chrysler Automobiles (FCA), was a Class 3 (GVWR 10,001 – 14,000 pounds) chassis certified truck that meets the Federal HDV2 Tier 3 Bin 570 standards, and has the same powertrain (engine and transmission) that FCA uses in their Class 4 engine certified trucks. The FCA test article had comparable gross combined vehicle weight (GCWR) as the Class 4 vehicles tested but employed aftertreatment technology tailored for Tier 3 chassis certification. Table 3-30 lists the major specifications of the three vehicle/powertrain combinations considered in this study.

Table 3-30: Heavy-Duty Gasoline Vehicle Emissions Investigation Vehicle Specifications

	G.M.	**Ford**	**FCA**
Configuration	Box Truck	Box Truck	Pickup
Certification	HDGE	HDGE	Tier 3 Bin 570
Model Year	2015	2016	2017
Odometer	48,000	37,000	43,000
Eng. System	NA/PFI/TWC	NA/PFI/TWC	NA/PFI/EGR/TWC
Displacement	6.0L V8	6.8L V10	6.4L V8
Power	297 hp @ 4,300 rpm	305 hp @ 4,250 rpm	410 hp @ 4,600 rpm
Torque	372 lbft @ 4,000 rpm	420 lbft @ 3,250 rpm	429 lbft @ 4,000 rpm
Transmission	6 spd Auto	6 spd Auto	6 spd Auto
GVWR (lbs)	14,500	14,500	13,300
GCWR (lbs)	20,500	22,000	19,900

3.2.1.2 Baseline Tests Performed for Exhaust Emissions

As previously stated, two of the vehicles tested were equipped with dynamometer certified engines for use in vehicles with a GVWR over 14,000 lbs while the third vehicle was a chassis certified HDV2. These vehicles were chosen as the engines representing the bulk of the HD SI vehicle market. The lighter HDV2 vehicle was chosen because of its chassis certification resulting in its aftertreatment system more closely resembling what is commonly found on Tier III light-duty vehicles.

The purpose of this particular program was to investigate the current state of HD SI engine criteria emissions performance. Because cold start emissions are not strongly emphasized in HD SI engine test, manufacturers generally locate three-way catalysts for exhaust aftertreatment significantly downstream from the exhaust manifold. Because chassis certification places a higher weighting on cold start results, the HDV2 vehicle we tested was designed to reach light-off temperature sooner. Its catalysts were significantly closer to the exhaust manifold than in an engine-certified configuration. Table 3-31 shows the average distance, in meters, from the outlet of the exhaust manifold to the front face of the catalyst substrate. Where two catalysts are used, one for each bank of cylinders on a V8 or V10 engine, the value represents the average of the two distances.

Table 3-31: Average distance from exhaust manifold to catalyst.

Manufacturer	Average Exhaust Manifold to Catalyst Distance (meters)
G.M./Isuzu	2
Ford	1.6
FCA	0.9

Location of the catalyst relative to the exhaust manifold has a significant impact on overall tailpipe emissions, as a shorter distance enables more rapid heating and catalytic reduction after cold start. A longer distance reduces the maximum catalyst temperature during high load operation, protecting the washcoat from thermal degradation. The assumptions investigated in this test program were as follows:

- Gasoline stoichiometric operation and advanced three-way catalyst can provide a high level of efficiency and nearly zero warmed-up emissions rates.
- Vehicle weights and loads can drive high exhaust gas temperatures.
- High exhaust gas temperatures can drive the need for fuel enrichment and related strategies that protect engine components and the catalyst.
- Location of the catalyst is partially dictated by exhaust gas temperature.
- Rearward catalyst locations can hinder catalyst light-off as well as performance under extended low-load operation.

3.2.1.3 On Road PEMS testing for Exhaust Emissions

Each vehicle was subject to real world emissions testing and driven during the workday on several routes EPA uses to collect real world drive emissions. During these drive evaluations, each vehicle was equipped with a portable emissions measurement system (PEMS). These PEMS units are 40 CFR part 1065 compliant, with independent emissions measurement for CO, CO_2, NO, NO_2 and NMHC emissions. Additionally, data is collected for exhaust flow, GPS location, environmental conditions, and selected CAN signals via the vehicles OBD connector. Temperatures within the exhaust system were also recorded during these drive cycles: exhaust gas temperature at the exhaust manifold outlet and catalyst inlet cone, and catalyst substrate temperature 1-inch reward of the front face of the catalyst.

Where possible, each vehicle was tested across a range of test weights: curb weight plus instrumentation; 90% GVRW (gross vehicle weight rating); and where possible, 90% GCWR (gross combined weight rating, which represents truck and trailer weight.) Each real-world drive schedule was on public roads and subject to varying traffic loads dependent on the time of day as well as all variation as a result of traffic control. When possible, each vehicle was driven on each route on three different days. Table 3-32 describes the routes used for collection of real-world drive emissions.

Table 3-32: Description of Real-world PEMS Testing Routes

Route	Distance (mi)	Avg. Speed (mph)	Description
A	7	21	Low speed, light load
B	12	24	Medium speed, short duration high load
C	32	45	High speed, short duration high load
D	84	63	High speed, sustained high load
E	30	40	Medium speed, high load

3.2.1.4 Laboratory Chassis Testing for Exhaust Emissions

The vehicles described previously were tested at EPA's Ann Arbor, MI National Vehicle and Fuel Emissions Laboratory (NVFEL). Laboratory testing was conducted to remove the variability inherent in real world PEMS testing such as engine coolant start temperature, load, and traffic conditions. Both the Ford and the GM/Isuzu vehicles engines were, as previously noted, certified to HD engine standards on an engine dynamometer, while the FCA vehicle was chassis certified as a complete vehicle to HDV2 standards. By testing engine certified vehicles on the chassis rolls, it was possible to highlight the emphasis each type of certification places on the vehicle, engine, and catalyst system design considerations. For testing purposes, each vehicle was subjected to both cold start and hot start testing using Tier 3 (10% ethanol) certification fuel. Each vehicle was tested at an estimated test weight (ETW) condition that represented the vehicle's curb weight plus the weight of instrumentation, as well as 90% GVWR condition. Because the FCA vehicle was certified as a Tier 3 HDV2 it was tested at curb weight plus 1/2 payload. Table 3-33 are the test weight and dynamometer coefficients used for this testing.

Table 3-33: Test weights and dynamometer coefficients used for NVFEL HD gasoline testing.

	Test #1	Test #2	Test #3 (FCA only)
ETW (lbs)	9,320	14,000	11,000
Target A (lbf)	123.23	87.54	87.54
Target B (lbf/mph)	0	1.399	1.399
Target C (lbf/mph^2)	0.0917	0.1215	0.1215

Each vehicle-engine combination was subject to four distinct test cycles, two of which were cold starts, with the remaining two warm starts. Each test cycle is described below.

Cold Start, Federal Test Procedure (FTP-75). The FTP-75 is used for emissions certification and fuel economy testing of light-duty vehicles in the United States. The FTP-75 consists of three phases: a cold transient phase (ambient temperature 20-30 deg. C), a stabilized phase and a hot start transient phase. The FTP-75 was used to understand the differences in cold start strategies and catalyst architectures resulting from the differences in certification testing between HD dynamometer and HD chassis. Figure 3-14 illustrates the FTP-75 three-phase test cycle.

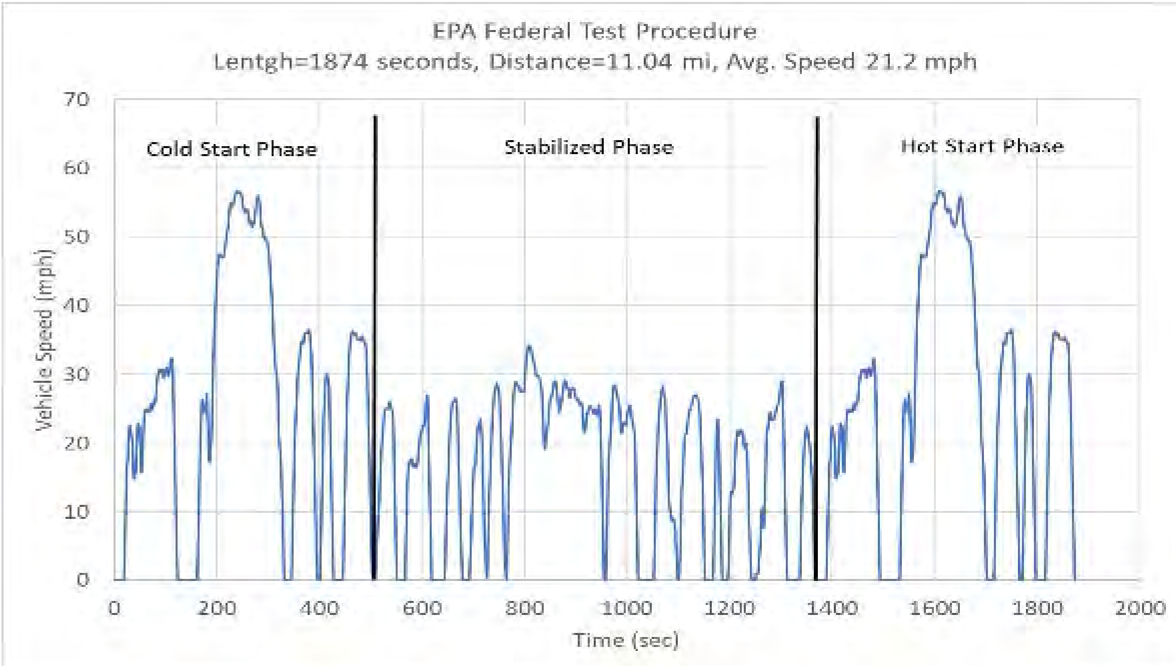

Figure 3-14: FTP-75, Cold start three phase test cycle

The Highway Fuel Economy Driving Schedule (HWFE) was chosen to compare emissions performance under simulated urban driving conditions. Figure 3-15 illustrates the HFE test cycle that was used. This test was run as double HWFE cycles, where the first cycle is used as a warmup, or prep, and there is no emissions sampling or recording conducted.

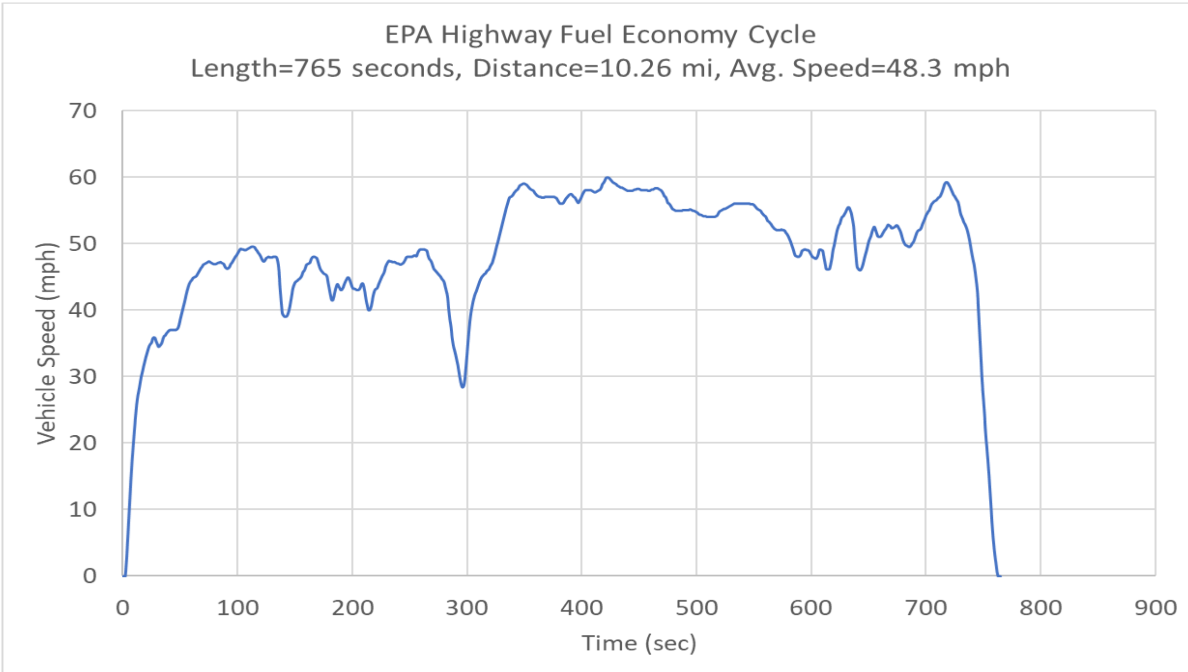

Figure 3-15: EPA Highway Fuel Economy Cycle

Phase LA92 drive cycle. The LA92 drive cycle is a CARB-developed dynamometer schedule. The LA92 was originally developed as an inventory improvement tool, and compared to the FTP, it has a higher top speed, a higher average speed, less idle time, fewer stops per mile, and higher rates of acceleration. For the purposes of this comparison testing, two back-to-back LA 92 cycles were utilized, doubling the distance and creating a 4-phase test. Phase 1, a warm start, was followed by a warmed-up Phase 2, followed by a 30-minute engine off soak. Phase 3 is a warm start after the engine off soak and is a repeat of the earlier Phase 1 drive cycle, and Phase 4 is a repeat of Phase 2.

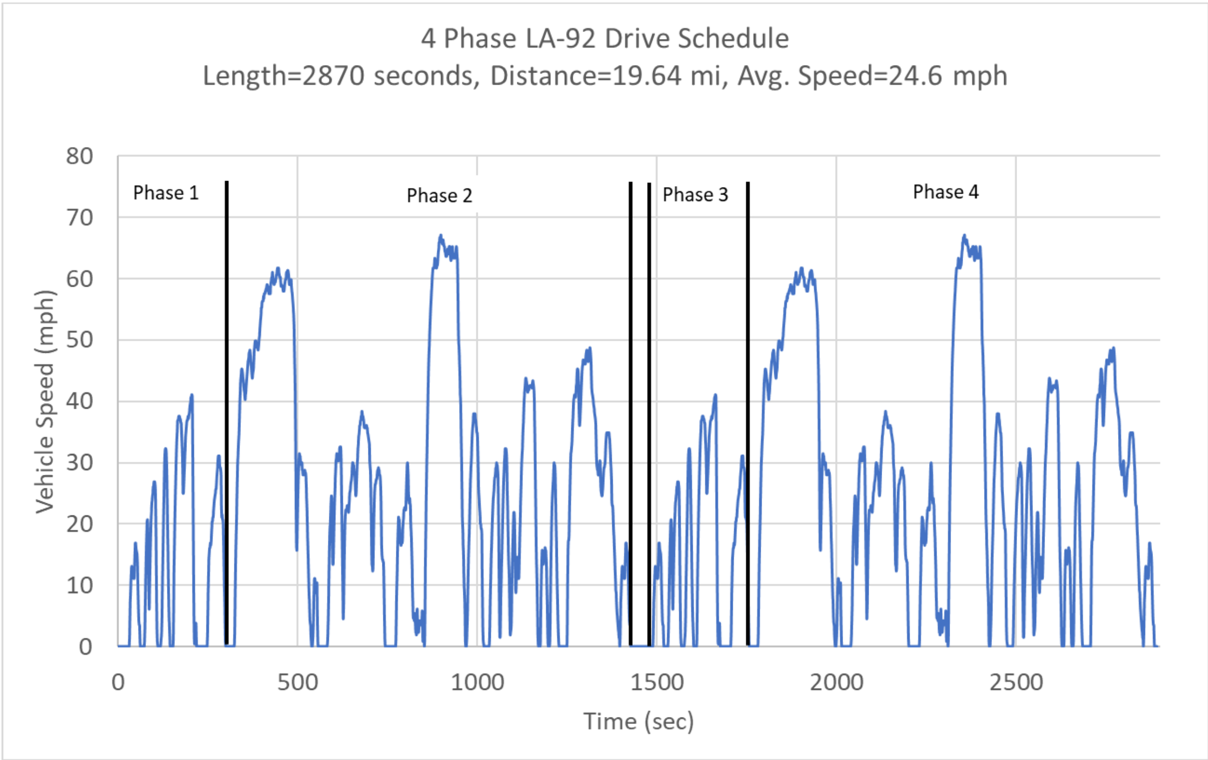

Figure 3-16: EPA 4 phase LA92 test cycle.

HD GEM Cycle (i.e., Super Cycle). This cycle is a composite of the many cycles used in the process of certifying trucks to HD GHG vehicle standards. The super cycle drive cycle consists of a combination of low speed, low load cycles followed by a 10-minute idle, and a return-to-service portion consisting of 55 and 65 mph cruise conditions. Phases 1 and 2 are consecutive ARB Heavy Heavy-Duty Diesel Truck transient modes from the ARB HHDDT schedule, are 14.3 miles in combined length, and represent an average speed 15.4 mph. Phases 1 and 2 are followed immediately by Phase 3, a 10-minute idle. Phase 4 is a return-to-service cycle consisting of an acceleration from idle to a 55-mph cruise, followed by another acceleration to a 65 mph cruise, and a return to idle. Phase 4 for has an average speed of 55.8 mph, is 29.2 miles in length, and both acceleration and deceleration rates are 0.5 mph/sec. The purpose of this cycle was to investigate how the lower exhaust gas temperature resulting from low-load operation affect catalyst activity and the emissions generated during a high-load, return-to-service event.

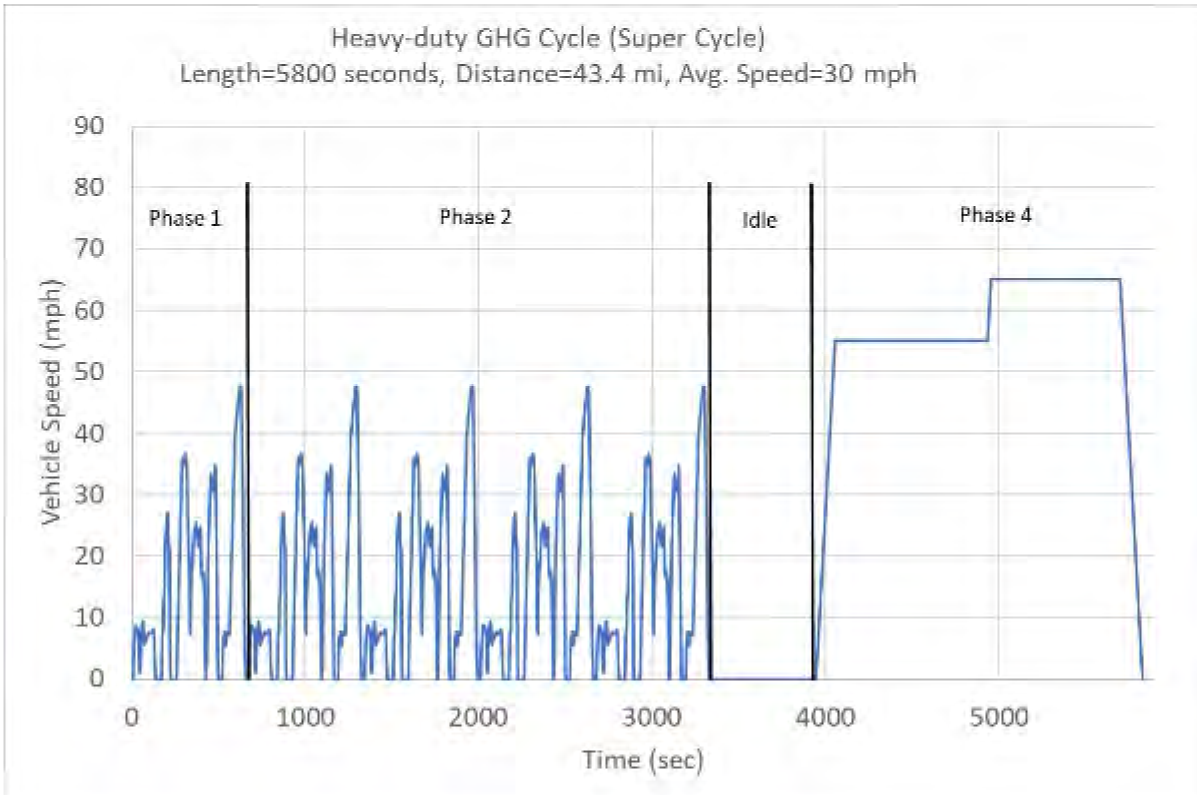

Figure 3-17: Super cycle, GEM greenhouse gas cycle

3.2.1.5 Baseline Exhaust Emissions Results

Due to the inherent variability of real-world driving conditions, as well as the absence of defined test cycles or off-cycle emission standards for HD gasoline engines, direct comparisons of the onroad PEMS and chassis testing results cannot be used to classify the emissions performance of any one vehicle as above or below an applicable HD standard. However, comparisons of emission rates observed under similar conditions, time to catalyst light-off, and overall performance of each configuration relative to the others, can be made.

Most illustrative are the test results and data acquired in the laboratory employing the FTP-75 cold start test procedure. Figure 3-18 and Figure 3-19 show cumulative hydrocarbons and NO_X respectively for each vehicle at each test weight. Each figure clearly shows that once catalyst light-off is achieved, the sharp knee in the curve between 20 seconds and 140 seconds, emissions rates decline significantly and remain so for the remainder of the test. The top lines in the figures also illustrate how quickly the emissions can accumulate if catalyst light-off is delayed, allowing most of the emissions totals to be achieved in the first minutes of operation.

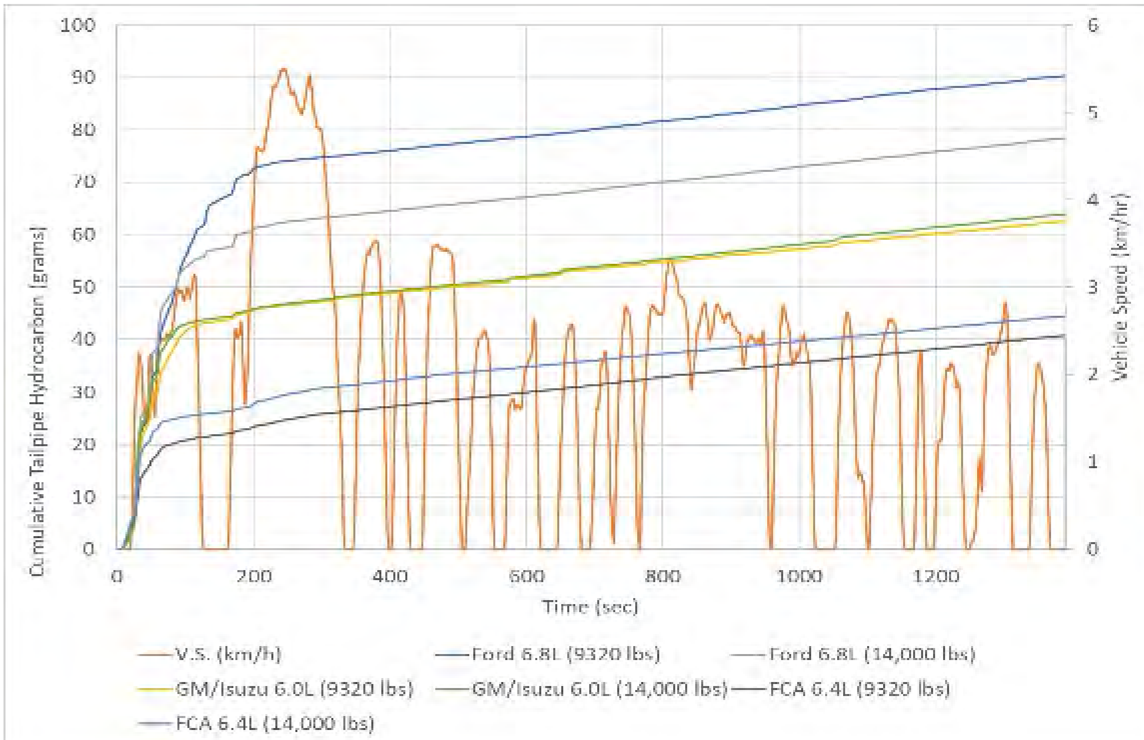

Figure 3-18: FTP-75 Cold start cumulative total HC comparison

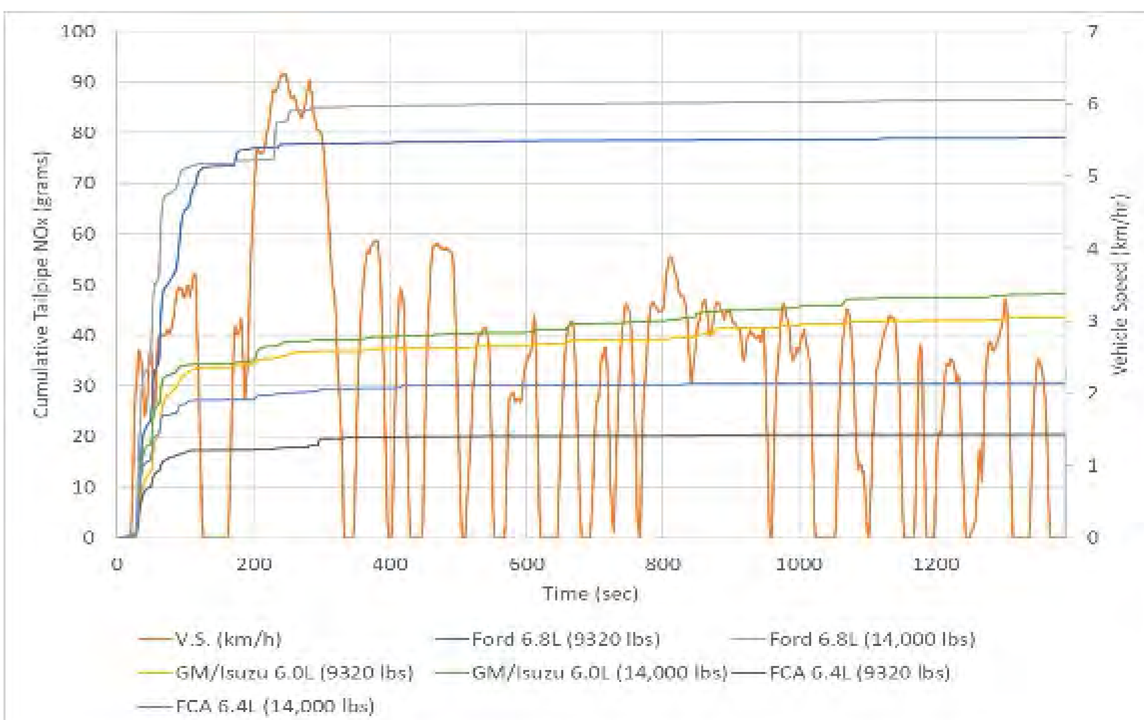

Figure 3-19: FTP-75 Cold start cumulative NO_X comparison

Figure 3-20 sharpens the focus on catalyst architecture as well as possible calibration techniques driven by the particular certification tests. Figure 3-20 shows the effect that engine load has on the exhaust temperature of the Ford 6.8L and the GM/Isuzu 6.0L, both of which are dyno certified with little emphasis on cold start emissions. In contrast the FCA 6.4L which is a Tier 3 Bin 570 chassis certified vehicle shows no impact on catalyst light-off time, exhaust gas temperature, due to increased load. The FCA 6.4L was certified to the FTP-75 which emphasizes cold start emissions. This emphasis results in catalyst architecture, shorter exhaust manifold to catalyst distance, as well as cold start controls and calibrations more closely related to light-duty trucks and passenger cars.

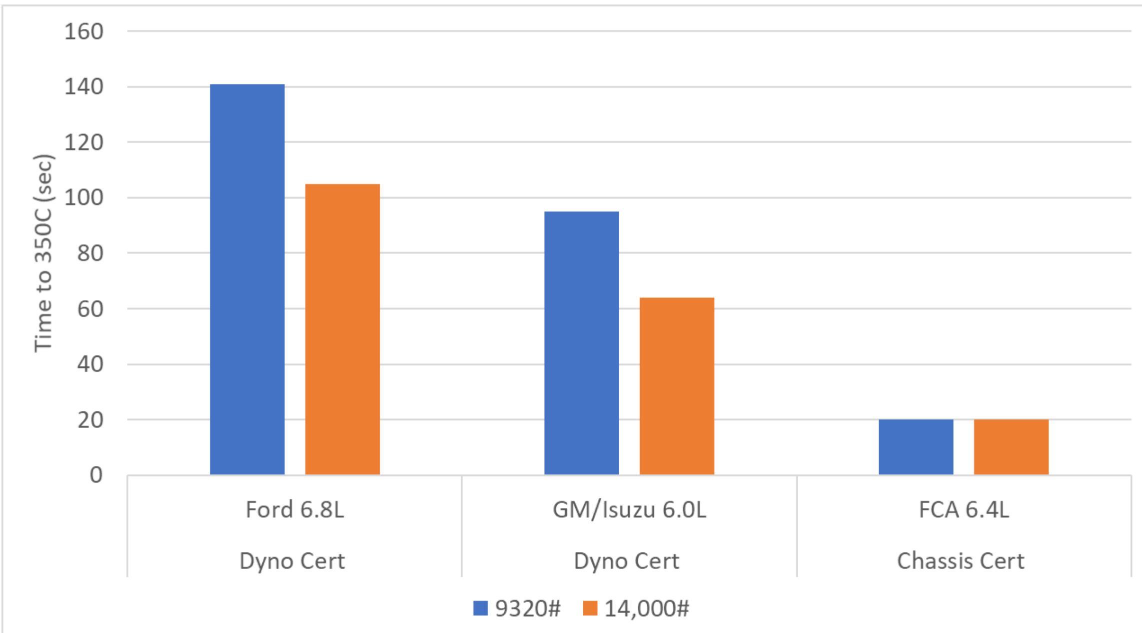

Figure 3-20: FTP-75 Catalyst light-off time comparison

Catalyst location can not only affect light-off but can also affect catalyst temperature during extended periods of idle. If idle conditions are long enough, catalyst temperatures may fall below 350 C, a temperature associated with reduced conversion efficiency. Given this reduced catalyst conversion efficiency, an engine out emissions spike caused by an applied load can become a tailpipe emissions spike. Figure 3-21 illustrates this condition during the 10-minute idle portion of the Super Cycle (Figure 3-17). Each vehicle enters the idle with a catalyst temperature of approximately 500 C. Over the course of the idle, catalyst temperatures decline. Those vehicles with the largest distances from manifold to catalyst (Table 3-31) fall below 300 C.

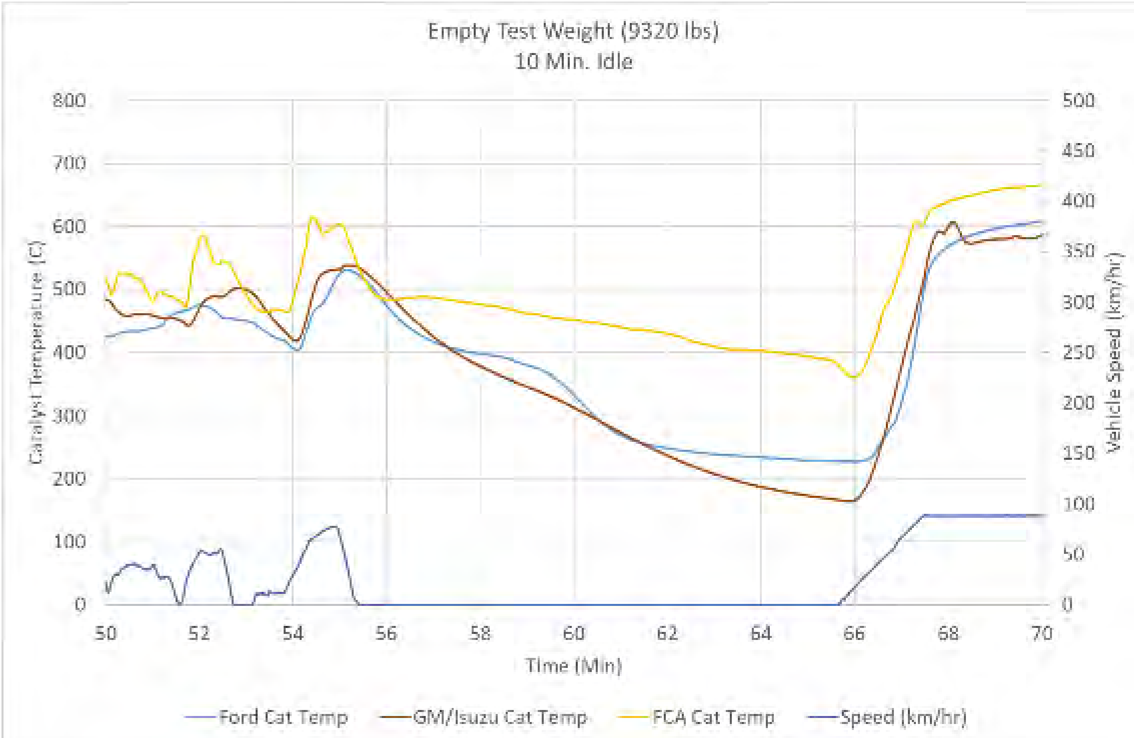

Figure 3-21: Extended idle catalyst cool down comparison

3.2.1.6 Baseline Refueling Emissions

As mentioned in Chapter 2.3.2.4 of this RIA, these vehicles are subject to evaporative emission standards, but have no refueling requirements. We are unaware of any HD SI engines certified for incomplete vehicles that implement ORVR technologies today. For our feasibility analysis, we believe these HD SI engines would not implement ORVR without a regulatory driver and assumed zero adoption of ORVR technology in our baseline.

3.2.2 **Projected Technology Effectiveness**

The emissions performance of the advanced catalyst technologies was evaluated in EPA's HD SI demonstration program. We also evaluated a combination of additional data sources including MY 2019 compliance data and engine mapping data to project the effectiveness of these technologies and inform the level of stringency in our standards. A description of these data and our analysis of them is presented in this section.

We project the effectiveness of implementing ORVR for incomplete HD SI vehicles based on the performance of complete vehicles subject to the Tier 3 evaporative and refueling requirements applying assumptions to account for increased fuel tank sizes.

3.2.2.1 MY 2019 HD SI Compliance Data for FTP Exhaust Emissions Performance

Four engine manufacturers certified HD SI engines in MY 2019. These manufacturers certified six engine families ranging in displacement from 6.0 to 8.8 liters.[19,20] Table 3-34 presents the MY 2019 FTP-based emission levels reported for the three pollutants addressed by a TWC: NO_X, NMHC and CO. We organized the engines by descending NO_X level. One engine, labeled "Cert Engine 6", is below the final NO_X standard for MY 2027 while maintaining relatively low NMHC and CO emissions. While this high performing engine, which is available today, demonstrates that it is possible to meet the new NO_X standard, we acknowledge that these certification results are representative of a shorter useful life period than we are implementing. PM emissions for most of these engines were undetectable and reported as zero for certification, suggesting the 5 mg/hp-hr standard is feasible for HD SI.[C]

Table 3-34: Family Emission Limits Reported for the Six Certified HD SI Engines in MY 2019; NO_X and NMHC values are converted from g/hp-hr to mg/hp-hr to match the units of our new standards

	Cert Engine 1	Cert Engine 2	Cert Engine 3	Cert Engine 4	Cert Engine 5	Cert Engine 6
NOx (mg/hp-hr)	240	160	120	104	70	40
NMHC (mg/hp-hr)	50	50	60	80	80	80
CO (g/hp-hr)	1.5	3.7	6.6	8.6	12.7	3.7
Fraction of MY 2019 HD SI Sales	20%	2%	20%	4%	48%	5%

In order to evaluate the NMHC and CO emissions, we calculated an overall average emission rate for each pollutant that includes all engines. Table 3-35 compares this average with the EPA 2010 standards, the new 2027 standards, and results from the engine family with the best NO_X emission performance of the MY 2019 compliance data.

Table 3-35: Average emission performance for Certified HD SI Engines in MY 2019

Pollutant	EPA 2010 Standard	EPA 2027 Standard	Overall Average	Best NO_X Performance
NO_X (mg/hp-hr)	200	35	122	40
HC (mg/hp-hr)	140	60	67	80
CO (g/hp-hr)	14.4	6	6.1	3.7

Table 3-35 compares the average NO_X, NMHC, and CO emission performance of the six engines and displays the EPA 2010 standard, the EPA 2027 standard, as well as the 2019 cert engine family with the best NOx. When calibrating their engines, SI manufacturers experience a tradeoff in emissions performance for the three pollutants in their TWCs and each manufacturer will optimize their emission controls differently. As expected, the table shows no clear trend in NMHC and CO emissions related to the reduced NO_X. The new 2027 standard levels are aligned with the six certification engines' average emissions for NMHC and CO. The new standards are likely achievable by minor calibration changes, such as incorporation of cold start catalyst light-off strategies and refinement of the catalyst protection fuel enrichment and related strategies. These results support FTP standards of 60 mg NMHC/hp-hr and 6 g CO/hp-hr, consistent with the overall average NMHC and CO levels achieved for 2019. We describe the feasibility of the final standards based on NMHC, CO, NOx, and PM emissions levels achieved in our demonstration program in detail in section 3.2.2.3.

[C] One engine reported a 0.005 g/hp-hr PM FEL.

Four engine manufacturers certified alternative fuels HD SI engines in MY 2019 – 2021 that performed favorably as compared to the MY 2027 standards. These manufacturers certified eight engine families ranging in displacement from 6.0 to 8.8 liters. Four of the engine families selected for comparison were certified with compressed natural gas (CNG) and four engine families were certified with liquified petroleum gas (LPG). Table 3-36 presents the FTP-based emission levels reported for the three pollutants: NO_X, NMHC and CO. Three of the CNG and all of the LPG engine families are below the final NO_X standards for MY 2027. All of the NMHC and CO emissions reported are below the new MY 2027 standards. We acknowledge that these certification results are representative of a shorter useful life period than we are finalizing in this rule; but, PM emissions levels certified for these selected CNG and LPG engines were 1 mg/hp-hr or lower, which supports the5 mg/hp-hr standard for PM.[D]

Table 3-36 Family Emission Limits Reported for Four CNG and Four LPG HD Engines in MY 2019 - 2021; NO_X and NMHC values are converted from g/hp-hr to mg/hp-hr to match the units of our new standards

	Cert E1	Cert E2	Cert E3	Cert E4	Average	MY 2027 Std
Fuel	CNG	CNG	CNG	CNG		
NOx (mg/hphr)	6	20	10	70	27	35
NMHC (mg/hphr)	3	9	22	7	10	60
CO (g/hphr)	4.0	1.3	2.5	4.4	3	6

	Cert E5	Cert E6	Cert E7	Cert E8	Average	MY 2027 Std
Fuel	LPG	LPG	LPG	LPG		
NOx (mg/hphr)	20	20	10	20	18	35
NMHC (mg/hphr)	15	48	42	51	39	60
CO (g/hphr)	2.8	2.7	5	5.6	4	6

3.2.2.2 EPA Engine Mapping Test Program for SET Exhaust Emissions Estimation

To assess the potential for emission reductions in HD gasoline engines over sustained loads, EPA evaluated engine fuel mapping data from a testing program previously performed by EPA as part the of the HD GHG Phase 2 rule. EPA contracted SwRI to test a production MY 2015 Ford 6.8L V10 gasoline engine to assess CO_2 emissions and to evaluate the new fuel mapping test procedures developed for that rulemaking. As part of that work, the engine was run on an early version of 40 CFR 1036.535, which is the steady state fuel mapping procedure that requires the engine to be run at nearly 100 speed and torque points for 90 seconds. The first 60 seconds are for the engine and fuel consumption to reach stability and the last 30 seconds are averaged to create the fuel map.

Since continuous dilute criteria emissions were also collected for the test, we recently directed SwRI to reevaluate those results and create three versions of the data that summarized fuel consumption and emissions (NO_X, CO, NMHC and CO_2) versus engine speed and torque. The first version analyzed conditions where the engine went into power enrichment, consistent with

[D] One engine reported a 0.005 g/hp-hr PM FEL.

strategy used in the production application of the engine. The second version analyzed the conditions where the engine controller activated a catalyst protection fuel enrichment strategy but did so before a power enrichment strategy was activated (this is due to a programmed delay for power enrichment of approximately one minute in the production engine controller). The third version analyzed only conditions where the engine maintained stoichiometric air-fuel ratio, achieved by limiting engine load to keep exhaust temperatures slightly below the level that would activate the thermal protection strategy programmed into the production software.

These three analyses of the data differed only in the peak torque portions of the map. As in other portions, the engines maintained stoichiometric air-fuel ratio control for a majority of the points (below about 90 percent throttle). For each of the maps, the peak torque points were used to calculate the A, B and C speeds as well as the torque values, so there were three unique sets of surrogate SET test points. Emission mass rates for CO, NMHC, NO_X, and CO_2 and fuel consumption were calculated from each map by interpolation of the maps at each of the SET test points. Finally, the results were weighted according to the existing CI-based weighting factors outlined in 40 CFR 1036.512. The engine and emission control components were not aged to the useful life requirements in this analysis.

The data analysis below includes operation at three distinct engine speeds described above and at several different loads, consistent with the approximate test points that would be required to perform the SET test procedure and then calculate a composite emissions level for the engine. The data presented include emission levels, fuel consumption rates, and engine power observed at the required SET test points and while operating in three distinct modes as allowed by production software controls (i.e., power enrichment mode, catalyst protection enrichment mode, and stoichiometric operation).

While not typically observed during the transient FTP test or torque mapping procedure, the engine controller activated a power enrichment mode after approximately one minute when throttle openings were above 90%. The extra fuel resulted in a slight increase in power. Power enrichment is sometimes used on gasoline engines to produce approximately 5% additional power beyond what is made when the air to fuel ratio is maintained at stoichiometry. Stoichiometric operation is the fundamental operating mode needed for three-way catalyst systems to simultaneously reduce HC, CO and NO_X emissions, but as described above, it is not the mode that produces peak power.

Another operating mode observed in the data is catalyst protection fuel enrichment. When the catalyst or other critical engine components are exposed to high exhaust gas temperatures, damage can occur that affects the durability of these components, and manufacturers typically implement control strategies that use a limited amount of fuel enrichment to cool the exhaust gas and protect critical components. The fuel enrichment reduces the amount of excess oxygen that supports the exothermic (heat releasing) reaction in the catalyst and also reduces the temperature of the combustion gases exiting the engine. The combination of these two temperature-reducing strategies effectively provides control of exhaust gas temperatures and protects critical exhaust components from irreversible damage. Other strategies that maintain effective emission control, expand the area of stoichiometric operation, and still provide protection of critical engine and catalyst components are discussed in Chapter 2 of the RIA.

As observed in the composite SET test data below, any enrichment mode, whether for power or catalyst protection purposes, can result in substantial emission increases and higher fuel consumption. As seen in Table 3-37, when the engine is commanded into power enrichment mode and is no longer maintaining stoichiometric operation, the NMHC and CO increase substantially, and the engine consumes more fuel. The NMHC emissions are more than 10 times higher while the CO emissions are almost 50 times higher than the stoichiometric operating mode. NO_X emissions are reduced about 60% in power enrichment mode as expected because of the rich operation, however stoichiometric NO_X emissions can be improved with catalyst design and calibration. Since this is a MY 2015 production engine, it was not designed or calibrated for optimum emissions for sustained high load operation at mid operating speeds such as demonstrated over the SET cycle. Improved NO_X emission control required over the FTP test cycle with this rule is expected to also result in improvements in the NO_X levels over the SET cycle. It is important to note that this power enrichment mode is not typically observed during the transient FTP test due to the short periods of time spent at high and full power loads. The short FTP time at load limits any power related enrichment features from activating like observed in the sustained full power test points in the SET testing described above.

Table 3-37: Comparison of Simulated 6.8L V10 SET Composite Emissions to MY 2027 Standards

	NO_X (mg/hp-hr)	**HC[a] (mg/hp-hr)**	**CO (g/hp-hr)**	**CO_2 (g/hp-hr)**	**BSFC (lb/hp-hr)**
Power Enrichment Allowed	11	110	45.2	587.3	0.479
Catalyst Protection with No Power Enrichment	19	30	11.4	617.7	0.463
Stoichiometric Operation	28	10	0.97	626.6	0.457
Spark-Ignition Exhaust Emission Standards for SET Duty Cycle MY2027 and later	35	60	14.4	-	-

[a] Hydrocarbons measured in the dataset were NMHC.

As discussed above and illustrated in Figure 3-22, NO_X emissions remain reasonably controlled under all operating modes; however, NMHC and CO emissions increases are closely tied to enrichment events. The MY 2027 HC and CO standards for the FTP cycle are achieved in stoichiometric operation, but CO begins to approach today's FTP standard when catalyst protection is enabled. Power enrichment causes drastic spikes in both NMHC and CO. We are including the SET duty cycle to incentivize manufacturers to expand the stoichiometric operation under heavy load conditions of their HD SI engines and maintain the maximum TWC effectiveness. The SET standards for HC and CO will require manufacturers to significantly reduce the frequency of fuel enrichment events, yet allow for some necessary catalyst protection and power enrichment operation. We are applying the same numeric values for FTP and SET duty cycles for HC and NOx standards. We are remaining generally consistent with a fuel neutral approach in the final FTP and SET standards, with the exception of CO for Spark-ignition HDE over the new SET duty cycle. These new SET standards are summarized in Table 3-38.

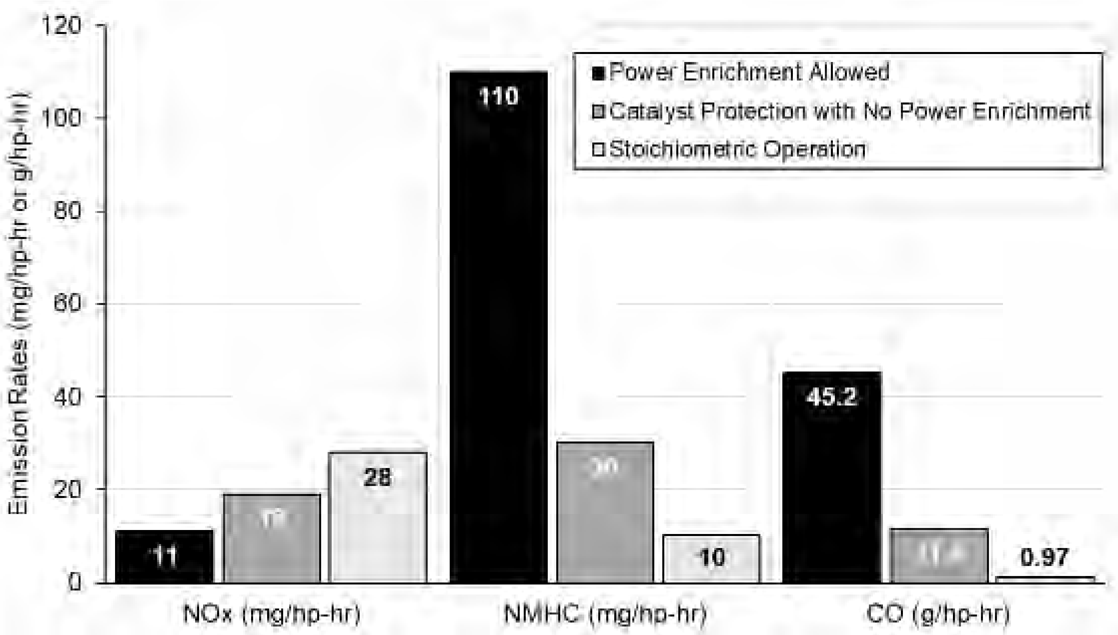

Figure 3-22: Comparison of operating modes in the fuel mapping-based SET

Table 3-38 Exhaust Emission Standards for SET Duty Cycle

	NO_X (mg/hp-hr)	**HC (mg/hp-hr)**	**CO (g/hp-hr)**
MY 2027 and later	35	60	14.4

The standards in Table 3-38 represent technically feasible levels in MY 2027 based on our analysis of the performance of products currently in the market, the use of current high-temperature-tolerant catalyst washcoats, and the design and calibration strategies available to ensure rapid catalyst light-off and reduce HC and CO emissions under high load. As indicated in the results of the SET composite analysis in Table 3-37, where emission results for three distinct modes of fuel control under high load operation were simulated, the emission levels for each can vary significantly.

First, the current power enrichment mode, which is allowed solely for the purpose of providing a modest increase in power, can produce emission results that exceed the MY 2027 composite SET standards (see Table 3-37 above). This fuel enrichment approach increases power but produces higher CO and NMHC emissions as a result, in addition to increasing fuel consumption. Reducing the amount to time spent in this enrichment mode, or eliminating it entirely, provides a significant reduction in emissions.

Second, the catalyst thermal protection mode, where fuel enrichment is used solely to limit temperatures inside the catalyst to a value specified by manufacturers, will drive emissions higher. This enrichment mode, controlled by software-based temperature models in the engine control module (ECM), also results in increased emissions, but is necessary to prevent irreversible damage to the catalyst. As indicated in Chapter 2 technology discussion, catalyst washcoats and other related exhaust components have progressed in recent light-duty applications and are able to tolerate significantly higher exhaust gas temperatures while still achieving acceptable component durability and catalytic deterioration targets. The use of these improved materials, along with the more robust temperature models or temperature measurement devices discussed in Chapter should result in significant reductions in CO emissions and allow engines to meet the new emission standards. Also discussed in Chapter 2.2.1.7 is the use of engine down speeding, which can avoid the high speed, high exhaust gas temperature conditions that typically result in fuel enrichment due to engine component durability and catalyst thermal concerns. With the integration of modern multi-speed electronically controlled transmissions, this down speeding approach is extremely feasible and likely to also reduce engine wear and improve fuel consumption with little perceived effect on performance under commercial and vocational operation. Note that in order to meet GHG and fuel consumption goals, this engine has already implemented some degree of down speeding as evident in the reduced maximum test speeds reported by one manufacturer. The agency believes that the more recent introduction of 10-speed transmissions provides additional opportunities for down speeding that have not yet been explored.

Finally, the third mode of operation, where the ECM maintains a stoichiometric fuel-to-air ratio throughout all of the SET cycle test points, and potentially, under high load real world operation as well, results in the greatest degree of emission control. Under stoichiometric operation, NMHC, CO, and NO_X emissions are simultaneously reduced, as the three-way catalyst can be optimized to reduce all three pollutants. This strategy is discussed in chapter 2.2.1.6, and our analysis of MY 2019 certification data indicates that NMHC and CO emissions are well below the new SET standards and NOX emissions meet the new SET MY 2027 and later standards. This level of NOX control was achieved without any improvements or refinements to the calibration and control strategies that we believe manufacturers will utilize to meet the new FTP standards. As observed in the analysis in Table 3-39, a slight drop in power is observed at the three SET test points as fuel enrichment decreases, however, this slight power loss is also accompanied by a noticeable decrease in fuel consumption, which can be a potentially important operational cost benefit in a commercial vehicle application. Similar to the previous discussion, the agency believes that several engine hardware and control technologies, as well as the additional gear ratios in current transmission designs, will provide the opportunity for maintaining stoichiometric fuel-air control under all load and speed conditions.

Table 3-39 SET Operation Mode Power Comparison

	Power (kW)			**Torque (Nm)**		
	SET Set Points			**SET Set Points**		
	A	**B**	**C**	**A**	**B**	**C**
Power Enrichment Allowed	211	187	145	546	572	547
Catalyst Protection with No Power Enrichment	211	182	141	542	554	524
Stoichiometric Operation	201	179	137	522	551	526

3.2.2.3 Spark-Ignition Technology Demonstration Program

EPA initiated a program with Southwest Research Institute to better understand the emissions performance limitations of current heavy-duty SI engines as well as investigate the feasibility of advanced three-way catalyst aftertreatment and technologies and strategies to meet new exhaust emission standards[21]. In addition to investigating emission performance on the FTP duty cycle, the test program evaluated the SET duty cycle that is now required for certification. This section describes the results of the SI demonstration program. See Chapter 1.2 for an expanded description of these and other technologies and strategies to address exhaust emissions for HD SI engines.

A MY 2019-certified heavy-duty gasoline engine was used for this evaluation. This particular engine was chosen because it represented the newest design among the three most-common engines in the market and included technologies not normally found in HD SI engines, such as variable valve timing (VVT) and cooled EGR. Additional considerations for selecting this engine were the availability of chassis-certified trucks with the options and driveline configuration desired, as well as the ability to install and operate the engine in a dynamometer test cell. Table 3-40 describes the HD SI engine that was used for this evaluation.

Table 3-40 Major engine specifications of the MY2019 HD SI gasoline engine used for the EPA demonstration program

Engine Component	Specification
Engine Displacement (L)	6.4
Configuration / Type	90° PushrodV-8
Bore (mm)	103.9
Stroke (mm)	94.8
Aspiration	Naturally aspirated
Injection	Sequential multi-port fuel injected
Compression Ratio	10.0:1
Engine Block Material	Cast Iron
Cylinder Head Material	Cast Aluminum, Hemispherical combustion chamber
Valve Train	2 valve per cylinder, Cam-in-block, VVT, Hydraulic roller lifters
Ignition	8 individual coils, 16 spark plugs, 2 per cylinder
Exhaust Gas Recirculation	Cooled EGR
Fuel Requirement	89 Octane recommended
Peak Horsepower	360 HP @ 4715 rpm
Peak Torque	408 lb-ft @ 4000 rpm

This program includes a baseline evaluation of emissions performance as well as a demonstration of the reductions possible through the application of advanced catalyst designs that included decreased substrate wall thickness and increased cell density, a washcoat formulation that is more tolerant of high exhaust gas temperatures, and forward placement of the catalyst substrate (i.e., moving a portion of the total catalyst volume closer to the engine). The catalysts were artificially aged to represent performance equivalent of 250,000 miles of real-world operation in a manner approved by the engine manufacturer.

We also investigated the impact of engine down-speeding and calibration changes to demonstrate further emission reduction potential of both the baseline and advanced catalyst configurations on the FTP and SET. As noted in Chapter 1.2, this engine down-speeding strategy is currently used by at least one HD gasoline engine manufacturer and this lower speed is made possible by transmission strategies preventing over-speeding, which allows the emission controls to operate in a much more desirable and lower emitting area of engine operation. For the down-speed testing in our demonstration program, the maximum test speed (MTS) was lowered from the manufacturer's stated MTS of 4715 rpm to 4000 rpm.

Finally, engine calibration parameters affecting air-fuel enrichments and biasing (lambda < 1.0) were manipulated to further reduce CO emissions on the SET. Because of the limited abilities of aftermarket vehicle engine control module programmers and the complexity of OEM engine calibrations and control strategies, this effort was met with limited success. We do however believe that manufacturers with access to the latest tools for calibration and complete access to engine control strategies and calibrations will be able to optimize lambda biasing as well as any necessary air-fuel enrichments for catalyst and engine protection. With the capabilities previously mentioned we believe that through a combination of engine down-speeding and calibration optimization the final emissions standards are achievable.

Installation of the engine in the test cell included instrumenting the engine's aftertreatment with thermocouples at exhaust manifold exit, catalyst inlet, and 1-inch rearward of the catalyst front face. For all engine tests, Controller Area Network (CAN) data from the engine control module, including, but not limited to engine speed, short and long-term fuel correction, and spark advance were recorded.

We evaluated the following test procedures, performing three repeats of each cycle:

- HD SET (40 CFR 1036.510)
- HD SI FTP cycle (40 CFR 1036.512(a)(1))
- Engine mapping (40 CFR 1036.535 and 1036.540)

In all tests, we measured NO_X, CO, PM, and NMHC, as well as the GHG-related parameters of brake-specific fuel consumption (BSFC), CH_4, and CO_2. Emissions were measured from two locations throughout each test cycle: before the catalytic aftertreatment and at the tailpipe.

Table 3-41 and Table 3-42 present results representing three technology packages over the FTP and SET duty cycles, respectively: advanced catalyst technology, advanced catalyst with engine down-speeding, and a combination of advanced catalyst, engine down-speeding, and calibration. Over both duty cycles, the results show NO_X and NMHC to be at or below the final MY 2027 standards. For the FTP results in Table 3-41, CO is below the final MY 2027 standards with the advanced catalyst alone and further reduced by downspeeding to 4000 rpm MTS. Figure 3-23 illustrates the CO breakthrough associated with the 4715 MTS. The lambda excursions seen in Figure 3-23 are a direct result of catalyst protection lambda enrichment specifically associated with higher engine speed operation as observed in Table 3-41 below. Please see the discussion in Chapter 1.2 regarding engine operating modes and possible calibration philosophy to address excess CO emissions. We applied one set of calibration changes to create a richer lambda bias and avoid throttle-based enrichment that led to higher SET CO levels. Those calibration changes resulted in slight increases in NO_X and CO over the FTP compared to the unmodified calibration, but the emission levels remained below the final standards.

Table 3-41 Spark-Ignition Demonstration Program FTP Results

	NO_X (mg/hp-hr)	CO (g/hp-hr)	NMHC (mg/hp-hr)	PM (mg/hp-hr)	BSFC (lb/hp-hr)
New Standards MY 2027 and later	35	6.0	60	5	
250k Catalysts 4715 RPM MTS	19	4.9	32	4.8	0.456
250k Catalysts 4000 RPM MTS	18	0.25	35	4.5	0.448
250k Catalysts 4000 RPM MTS Modified Cal	21	0.99	1	4.4	0.448

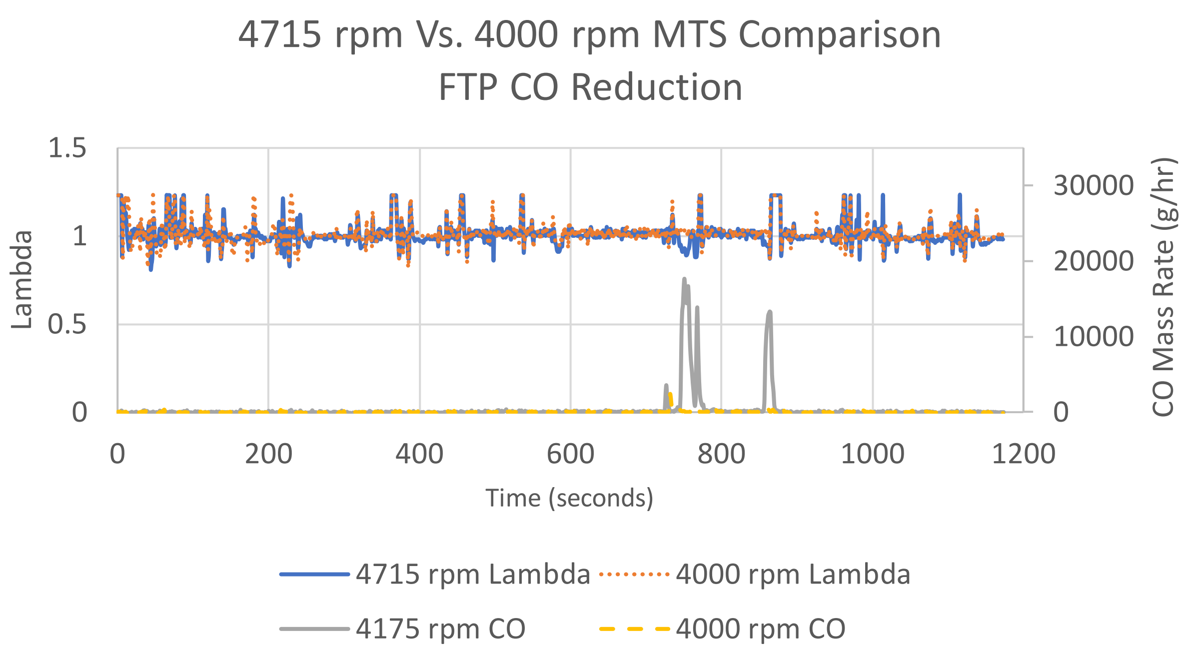

Figure 3-23: Engine RPM Down-Speeding FTP CO Comparison

Table 3-42 compares the emissions results over the SET cycle at MTSs of 4715 rpm and 4000 rpm. Like the FTP results NMHC and NO_X remained low for all three technology packages. Unlike the FTP results, the advanced catalyst alone did not reduce CO below the new standard. Although CO at 4000 rpm MTS meets the new standard, calibration changes, described above, were applied to attempt further reducing the CO. The calibration changes effectively reduced NOx, but did not have the desired effect of reducing CO. We note that some of the catalyst separated from the mat during the 4715 rpm test. The catalyst continued to control NOx, CO, and NMHC, but some of the separated material was captured by the particulate filter resulting in the 7 mg/hp-hr PM measurement shown in Table 3-42. In the absence of the separated catalyst material, we expect the PM level would be below the 5 mg/hp-hr standard we are finalizing.

Table 3-42 Spark-Ignition Demonstration Program SET Results

	NO_X (mg/hp-hr)	CO (g/hp-hr)	NMHC (mg/hp-hr)	PM (mg/hp-hr)	BSFC (lb/hp-hr)
New Standards MY 2027and later	35	14.4	60	5	
250k Catalysts 4715 RPM MTS	8	36.7	6	7	0.462
250k Catalysts 4000 RPM MTS	5	7.21	1	3	0.437
250k Catalysts 4000 RPM MTS Modified Cal	1	9.65	1	3	0.438

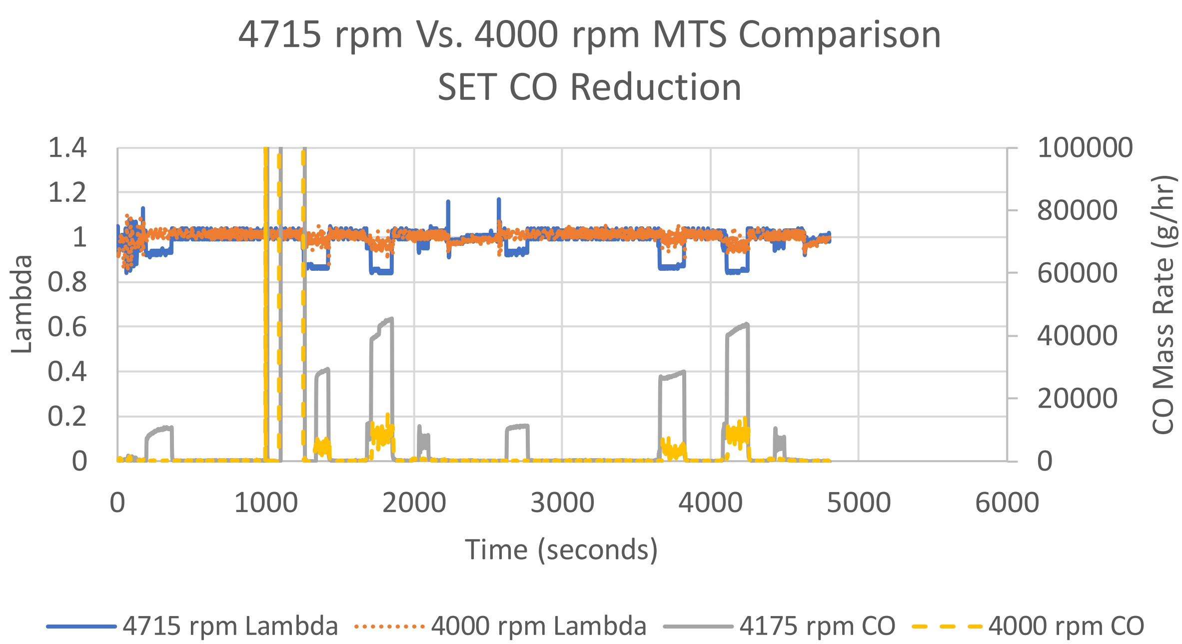

Figure 3-24: Engine RPM Down-Speeding SET CO Comparison

As mentioned previously, SwRI undertook a calibration effort to address the level of CO witnessed, 7.21 g/hp-hr, during SET testing and lower NO_X for both the FTP and SET. Because of the limited abilities of aftermarket vehicle engine control module programmers and the complexity of OEM engine calibrations and control strategies, this effort was met with limited success. We do however believe that manufacturers with access to the latest tools for calibration and complete access to engine control strategies and calibrations will be able to optimize lambda biasing as well as any necessary air-fuel enrichments for catalyst and engine protection. With the capabilities previously mentioned we believe that through a combination of engine down-speeding and calibration optimization the SET CO emission standard of 14.4 g/hp-hr is achievable.

3.2.2.4 Refueling Emissions Technology Effectiveness

As described Chapter 2.3.2.4 of this RIA, HD SI engines certified as incomplete heavy-duty vehicles are not currently required to meet ORVR. The technology package we considered for these engines is based on the technologies implemented by chassis-certified complete vehicles to meet the evaporative and refueling requirements of Tier 3. The technology package includes four main equipment components and strategies that incomplete heavy-duty vehicles may need to update to implement ORVR: increased working capacity of the carbon canister to handle additional vapors volumes, flow control valves to manage vapor flow pathway during refueling, filler pipe and seal to prevent vapors from escaping, and the purge system and management of the additional stored fuel vapors. Chapter 1.2.3 includes descriptions of these technologies. The assumptions we applied to account for the larger fuel tanks and other considerations for larger incomplete vehicles are summarized in Chapter 3.2.3.2 where we present our projected direct manufacturing costs.

The final refueling standards are projected to result in 27.8% lower VOC and Benzene by 2030, 80.2% lower by 2040 and 88.5% lower by 2045 for heavy duty gasoline vehicles over 14,000 lbs. See the discussion and results in Chapter 5.3.

3.2.3 Estimated Direct Manufacturing Costs for Technology Packages Evaluated

For this analysis of the aftertreatment costs, heavy-duty spark-ignition (HD SI) engines are categorized by the type of fuel they use: liquid fuels (i.e., gasoline, gasoline-ethanol blends, and ethanol) or gaseous fuels (i.e., compressed natural gas and liquified petroleum gas). The gaseous-fueled category includes engines derived from SI platforms and engines converted to SI from heavy-duty CI engines. The heavy-duty SI engine category is further divided into heavy heavy-duty (HHD) and urban bus. We projected the costs of achieving the HD SI engine exhaust emission standards based on the technologies we evaluated in our demonstration program (see Chapter 3.2.2.3).

3.2.3.1 Spark Ignition Exhaust Aftertreatment System Direct Manufacturing Cost Analysis

Manufacturers will optimize the design of their aftertreatment systems specific to their different vehicles. Manufacturers' primary considerations include cost, light-off performance, warmed-up conversion efficiency, and the exhaust temperatures encountered by the vehicle during high-load operation. Vehicles having low power-to-weight ratios will tend to have higher exhaust gas temperatures and exhaust gas flow which will result in a different design when compared to vehicles having higher power-to-weight ratios.

Manufacturers and catalyst suppliers perform detailed studies evaluating the cost and emission performance of aftertreatment systems. It is anticipated that manufacturers will optimize their aftertreatment systems to achieve the heavy-duty emission standards and meet the durability criteria for all vehicle classes.

Similar to the CI engine cost analysis, costs for baseline HD SI engine aftertreatment systems were estimated using cost data published by Dallman et al.[22], Pasoda et al.[23] as well as data from manufacturer's technical descriptions of aftertreatment catalyst components submitted as part of engine certification packages for MY 2019. Manufacturer's data were then combined into projected sales-weighted averages by type of fuel (liquid and gaseous fuels), including two categories for gaseous-fueled engines identified as heavy heavy-duty and urban bus that have distinctly different aftertreatment demands. Costs in this study are driven by the catalyst and precious metal loading. No significant labor costs were identified. The direct manufacturing cost for these technology packages is equal to the catalyst piece cost.

Baseline projected sales-weighted average engine displacements, catalyst volumes, PGM loadings and costs are shown in Table 3-43 for both liquid and gaseous fueled SI engines.[E] As mentioned previously, these are based on certification data from MY 2019. These MY 2019 engine and aftertreatment costs estimates are used as the MY 2027 baseline cost presented in RIA Chapter 3.2.3.1 after conversion to 2021 dollars.

Table 3-43: 2019 MY Sales-Weighted Baseline SI Engine Direct Manufacturing Costs (2021$)

	Liquid Fueled SI Engine	**Gaseous Fueled SI Engine**
Engine Displacement (L)	6.6	7.2
Total TWC Volume (L)	4.3	5.1
CATv/ENGd Ratio (L/L)	0.65	0.72
Total Pt ($)	$0	$0
Total Pd ($)	$140	$178
Total Rh ($)	$2,371	$2,214
Substrate cost ($)	$56	$68
Washcoat cost ($)	$26	$31
Canning cost ($)	$15	$19
Total direct manufacturing cost ($2019)	*$2,371*	*$2,510*

We separately evaluated two distinct categories for the three gaseous-fueled HD SI engines that certified to California's optional and more stringent 0.02 g/hp-hr NO_X standard in MY 2019: HHD and urban bus. One engine is derived from a traditional SI gasoline-fueled engine and two are converted from CI diesel-fueled engines. Given the small number of engines in each of these categories, we are not publicly releasing the component-level details for MY 2019 HD SI, HHD and urban bus engines, but summarize the total costs for these categories. Table 3-44 shows the baseline aftertreatment cost for HHD and urban bus gaseous-fueled engines.

Table 3-44: 2019 MY HHD and Urban Bus Gaseous-fueled Direct Manufacturing Baseline Costs (2021$)

	Gaseous Fueled SI Engine	**Gaseous Fueled HHD Engine**	**Gaseous Fueled Urban Bus Engine**
Engine Displacement (L)	6.8	11.9	8.9
Total direct manufacturing cost ($2019)	*$5,770*	*$8,474*	*$6,356*

As mentioned previously, the three MY2019 gaseous-fueled HD SI, HHD and urban bus engines currently meet a 0.02 g/hp-hr NO_X standard, and we assumed that additional technology would not be needed for these engines to meet the standards in future model years. However, it is

[E] PGM costs were determined by taking the average price across all regions from 8/31/2020 to 8/31/2022 from Johnson Matthey's PGM management website. https://matthey.com/products-and-markets/pgms-and-circularity/pgm-management/

reasonable to believe that improvements in the materials and design of the catalyst substrate support structure (e.g., can material, mat, seals, etc.) will be needed to achieve durability over the longer useful life and we estimated a nominal addition per-engine cost for these engine categories.

For the other gaseous fueled engines category, we assumed the same technologies would be used to meet the MY2027 and later standards. Table 3-45 shows the MY 2027 and later gaseous-fueled engine direct manufacturing costs adjusted for improved catalyst and component durability.

Table 3-45: Projected Gaseous Fueled Engine Direct Manufacturing Cost (2021$)

	Gaseous Fueled SI Engine	**Gaseous Fueled HHD Engine**	**Gaseous Fueled Urban Bus Engine**
MY 2027 and later total direct manufacturing cost ($2019)	*$5,770*	*$8,474*	*$6,356*

The MY 2027 technology cost for the liquid fueled SI engines are based on the demonstration engine described in Chapter 3.2.2.3. Costs were estimated using the same Dallman et al.[24] and Pasoda et al.[25] data as our baseline estimates and data from the specific aftertreatment catalyst components used for the HD SI demonstration program. We did not make any specific cost adjustments to account for the lengthened useful life, since the aftertreatment system used in the demonstration program represented catalysts aged to 250,000 miles. Table 3-46 contains the details of this analysis.

Table 3-46: Projected Liquid Fueled SI Engine Piece Cost (2021$)

Technology Description	**MY 2027 and later Liquid Fueled SI Engine**
Total TWC Volume (L)	5.8
CATv/ENGd Ratio (L/L)	0.91
Light-off Catalyst	
Number of Catalysts	2
L.O. Catalyst Volume (L)	0.82
Total Pt ($)	$0.0
Total Pd ($)	$707
Total Rh ($)	$496
Substrate Cost ($)	$21
Washcoat Cost ($)	$10
Canning Cost ($)	$6
Underfloor Catalyst	
Number of Catalyst	2
U.F. Catalyst Volume (L)	2.1
Total Pt ($)	$0
Total Pd ($)	$494
Total Rh ($)	$1,270
Substrate Cost ($)	$55
Washcoat Cost ($)	$25
Canning Cost ($)	$18
Total Demonstration TWC Cost ($2019)	*$3,101*

Table 3-47 summarizes the costs for each of the HD SI engine categories evaluated in this analysis. These direct manufacturing costs are used in the analysis to determine the overall costs of the program, as detailed in Chapter 7 of this RIA.

Table 3-47: Summary of HD SI Engine Direct Manufacturing Cost Comparison

Cost Packages (2021$)	**Liquid Fueled SI Engine**	**Gaseous Fueled**		
		SI Engine	**SI HHD**	**SI Urban Bus**
Baseline Technology	$2,371	$2,510	$8,447	$6,335
MY 2027 Technology	$3,101	$5,770	$8,474	$6,336
MY 2027 Incremental	*$730*	*$3,260*	*$28*	*$21*

3.2.3.2 Onboard Refueling Vapor Recovery Anticipated Costs

As described in Chapter 2.3.2.4 of this RIA, HD SI engines certified as incomplete heavy-duty vehicles are not currently required to meet ORVR. There are four main equipment components and strategies incomplete heavy-duty vehicles need to update to implement ORVR: increased working capacity of the carbon canister to handle additional vapors volumes, flow control valves to manage vapor flow pathway during refueling, filler pipe and seal to prevent vapors from escaping, and the purge system and management of the additional stored fuel vapors. Chapter 1.2.3 includes more information on these technologies. The associated direct manufacturing costs for these updates are summarized below. No labor cost was identified so the direct manufacturing cost is equal to the piece cost plus tooling cost (per piece). ORVR requirements will be extended to heavy-duty gasoline engines in incomplete vehicles starting in model year 2027. For our cost analysis, we assumed all heavy-duty gasoline engines that are identified as LHD, MHD and HHD in MOVES will have an average of a 70-gallon fuel tank.

Capturing the increased vapor volume from the vapor displaced during a refueling event will require canisters to increase vapor or "working" capacity approximately 15%-40% depending on the individual vehicle systems (i.e., fuel tank size). This can be achieved by increasing the canister volume using conventional carbon, the fundamental material used to store fuel vapors. A typical Tier 3 canister has approximately 5.1 liters of conventional carbon to capture overnight diurnal evaporative emissions for a 70-gallon fuel tank. An increase in required capacity to allow refueling vapors to be captured results in the need for an additional 1.9 liters of conventional carbon. A change in canister volume to accommodate additional carbon includes increased costs for retooling and additional canister plastic material, as well as design considerations to fit the larger canister on the vehicle.

An alternative to retooling for a larger single canister would be to add a second canister for the extra canister volume to avoid the re-tooling costs. Several smaller volume canisters are available on the market today. Another approach, based on discussions with canister and carbon manufacturers, can be achieved by using a higher adsorption carbon along with modifications to compartmentalization within the existing canister plastic shell that will increase the canister working capacity without requiring a larger canister size.

Additionally, there are two primary technologies used to prevent vapors from escaping into the atmosphere through the filler neck and around the fuel nozzle area when the vehicle is refueling that can affect the canister vapor capacity design requirements: a mechanical seal which makes direct physical contact with the refueling nozzle to create a nozzle to filler neck seal; or a liquid seal further down in the filler pipe which uses the liquid fuel mass flowing down the filler pipe and entering the tank to hydraulically prevent vapors from migrating back up the fill pipe. There is approximately a 20% reduction in carbon volume required if a mechanical seal is used at the filler neck versus a liquid seal approach. While mechanical seals are not currently the preferred technology, manufacturers facing the choices available for the larger volume fuel tanks and the need for a larger matching carbon containing canister to handle these large quantities of fuel vapors, may opt for more a mechanical seal design to avoid excess canister carbon requirements and possible retooling charges. We share our assumptions and cost estimates for both seal options in Table 3-48 and Table 3-49. A mechanical seal approach costs approximately $10.00 per seal. A dual tank may require two seals if dual filler necks are used instead of a single filler neck and transfer pump to move fuel between the two tanks.

The second required equipment update would be to install flow control valves, which may be integrated into existing roll-over/vapor lines. The flow control valves are needed to manage the vapors during the refueling event by providing a low restriction pathway for vapors to enter the canister for adsorption and storage on the carbon materials. We anticipate vehicles would require on average one valve per vehicle which would be approximately $6.50 per valve. A dual tank system may require a flow control valve system per tank depending on the design approach.

Thirdly, as mentioned above, a filler pipe and seal system would be needed for each filler nozzle to keep the vapors contained during refueling. Manufacturers have the option of a mechanical seal that costs approximately $10.00 per seal, or a liquid seal which in itself costs nothing but requires approximately $15 of new hardware modifications to provide enough back pressure to stop the refueling nozzle fuel flow when tank reaches full capacity.

Lastly, the engine control of the canister purge rates would need to be addressed. This update will include calibration improvements and potentially additional hardware to ensure adequate purge volumes are achieved as required to maintain an appropriate canister state to manage vapors generated during diurnal and subsequent refueling events. If required for a dual tank system, an extra purge valve may be needed if the two-tank system maintains independent canisters instead of a single common cannister as observed in dual-tank, single canister light-duty applications.

Table 3-48 shows our calculations estimating the amount of extra canister size for conventional carbon for a 70-gallon tank, using Tier 3 requirements as a baseline. Currently under Tier 3 requirements the canister and purge strategy are sized for the diurnal test and designed to meet the Bleed Emissions Test Procedure (BETP) requirements. During the diurnal test, the canister is loaded with hydrocarbons over two or three days, allowing the hydrocarbons to load a conventional carbon canister (1500 GWC, gasoline working capacity) at a 70 g/L efficacy. During a refueling event, which takes place over a few minutes, the vapor from the gas tank is quickly loaded onto the carbon in the canister with an ORVR system, causing the efficiency of the canister loading to drop to 50 g/L efficacy mainly because of the high volume of fuel vapors and the composition of those vapors required to be adsorbed in the short period of a refueling event. Typically, a design safety margin adds an extra 10% carbon to ensure adequate performance over the life of the system. Therefore, even though there is typically less fuel vapor mass generated and managed during a refueling event than is generated over a three-day diurnal time period, the amount of carbon that is necessary to contain the vapor is higher for a refueling event.

In order for carbon in the canisters to be effective at managing vapors for diurnals and refueling events, the vehicle engine must sufficiently purge the canister during engine operation in preparation for future events that will require vapor adsorbing capacity. The purge requirements are shown in Table 3-48. The diurnal drive cycle is only 30 minutes and targets 200 bed volumes of purge to clean the canister before the evaporative emissions test. When the bed volumes of purge are multiplied by the canister volume, the total purge volume can be calculated. The total purge volume divided by the number of minutes driving gives us the average purge rate. An ORVR test requires proper conditioning for a very clean canister in order to pass the ORVR test. To clean out the canister over the 97 minutes of driving cycles for the ORVR prep, a much higher amount of bed volumes is necessary; therefore, the purge rate required is also higher. Table 3-49 shows cost estimations for the different approaches. For our direct manufacturing cost we used $25 (2019 dollars), which is the average of all approaches considered, as the cost estimate for the additional canister capacity and hardware to meet the refueling standard. These costs were converted to 2021 dollars as described in the cost analysis of Chapter 7.

Table 3-48: Assumptions for gasoline-fueled heavy-duty spark-ignition vehicles for conventional carbon requirements to meet the refueling standard

	Tier 3 Baseline for Evaporative Standards	Updates for Refueling Standards	
		Mechanical Seal	Liquid Seal
	Diurnal	ORVR	
Diurnal Heat Build	72-96°F	80°F	
RVP	9 psi		
Nominal Tank Volume	70 gallons		
Fill Volume	40%	10% to 100%	
Air Ingestion Rate		0%	13.50%
Mass Vented per heat build, g/day	120		
Mass Vented per refueling event		255	315
Hot Soak Vapor Load	5		
Mass Vented over 48-hour test	227.2		
Mass Vented over 72-hour test	323.3		
1500 GWC, g/L [a]	70	50	50
Excess Capacity	10%	10%	10%
Canister Volume, liters[b]			
48-hour	3.6		
72-hour	5.1		
ORVR[c]		5.6	6.9
Limiting Drive Cycle, minutes	30	97	97
Bed Volumes Purge	200	646	646
Total Purge Volume, liters[d]	1020	3618	4457
Average Purge Rate, LPM[e]	34	37	46
BETP Purge		37	46

[a] Storage capability of conventional carbon

[b] Canister Volume = 1.1(mass vented)/ 1500 GWC (Efficiency)

[c] ORVR adds .5 liters and 1.8 liters for Mechanical Seal and Liquid Seal respectively

[d] Total Purge Volume, liters = canister volume, liters * Bed Volumes Purge represent the potential volume of purge for the 97 minute drive cycle used for the ORVR test procedure. Required purge volume to clean out the canister of fuel vapors for the larger ORVR canisters is likely much lower, approximately the ratio of new canister volume to the previous canister volume multiplied by the target bed volumes (220 bed volumes for a mechanical seal and 271 bed volumes for a liquid seal)

[e] Average Purge Rate, LPM = Total Purge Volume, liters / Limiting Drive Cycle, minutes however as noted in (d), this is not necessarily the required purge volumes or rates

Table 3-49: Estimated Direct Manufacturing Costs for ORVR Over Tier 3 as Baseline

	Liquid Seal		Mechanical Seal	
	New Canister	**Dual Existing Canisters in Series**	**New Canister**	**Dual Existing Canisters in Series**
Additional Canister Costs	$20	$15	$8	$8
Additional Tooling[a]	$0.50		$0.50	
Flow Control Valves	$6.50		$6.50	
Seal	$0	$0	$10	
Total[b]	$27	$22	$25	

[a] Assumes the retooling costs will be spread over a five-year period
[b] Possible additional hardware for spitback requirements

Chapter 3 References

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Chapter 4 Health and Environmental Impacts

4.1 Health Effects Associated with Exposure to Pollutants

Heavy duty vehicles emit pollutants that contribute to ambient concentrations of ozone, PM, NO_2, CO, and air toxics. A discussion of the health effects associated with exposure to these pollutants is presented in this section of the RIA. The following discussion of health impacts is mainly focused on describing the effects of air pollution on the population in general.

Additionally, because children have increased vulnerability and susceptibility for adverse health effects related to air pollution exposures, EPA's findings regarding adverse effects for children related to exposure to pollutants that are impacted by this rule are noted in this section. The increased vulnerability and susceptibility of children to air pollution exposures may arise because infants and children generally breathe more relative to their size than adults do, and consequently may be exposed to relatively higher amounts of air pollution.[1] Children also tend to breathe through their mouths more than adults and their nasal passages are less effective at removing pollutants, which leads to greater lung deposition of some pollutants, such as PM.[2,3] Furthermore, air pollutants may pose health risks specific to children because children's bodies are still developing.[A] For example, during periods of rapid growth such as fetal development, infancy, and puberty, their developing systems and organs may be more easily harmed.[4,5] EPA's America's Children and the Environment is a tool which presents national trends on air pollutants and other contaminants and environmental health of children.[6]

4.1.1 **Ozone**

4.1.1.1 Background on Ozone

Ground-level ozone pollution forms in areas with high concentrations of ambient nitrogen oxides (NOx) and volatile organic compounds (VOCs) when solar radiation is high. Major U.S. sources of NO_X are highway and nonroad motor vehicles and engines, power plants, and other industrial sources, with natural sources, such as soil, vegetation, and lightning, serving as smaller sources. Vegetation is the dominant source of VOCs in the U.S. Volatile consumer and commercial products, such as propellants and solvents, highway and nonroad vehicles, engines, fires, and industrial sources also contribute to the atmospheric burden of VOCs at ground-level.

The processes underlying ozone formation, transport, and accumulation are complex. Ground-level ozone is produced and destroyed by an interwoven network of free radical reactions involving the hydroxyl radical (OH), NO, NO_2, and complex reaction intermediates derived from VOCs. Many of these reactions are sensitive to temperature and available sunlight. High ozone events most often occur when ambient temperatures and sunlight intensities remain high for several days under stagnant conditions. Ozone and its precursors can also be transported hundreds of miles downwind, which can lead to elevated ozone levels in areas with otherwise low VOC or NO_X emissions. As an air mass moves and is exposed to changing ambient

[A] Children's environmental health includes conception, infancy, early childhood and through adolescence until 21 years of age as described in the EPA Memorandum: Issuance of EPA's 2021 Policy on Children's Health. October 5, 2021. Available at https://www.epa.gov/system/files/documents/2021-10/2021-policy-on-childrens-health.pdf.

concentrations of NO_X and VOCs, the ozone photochemical regime (relative sensitivity of ozone formation to NO_X and VOC emissions) can change.

When ambient VOC concentrations are high, comparatively small amounts of NO_X catalyze rapid ozone formation. Without available NO_X, ground-level ozone production is severely limited, and VOC reductions would have little impact on ozone concentrations. Photochemistry under these conditions is said to be "NO_X-limited." When NO_X levels are sufficiently high, faster NO_2 oxidation consumes more radicals, dampening ozone production. Under these "VOC-limited" conditions (also referred to as "NO_X-saturated" conditions), VOC reductions are effective in reducing ozone, and NO_X can react directly with ozone resulting in suppressed ozone concentrations near NO_X emission sources. Under these NO_X-saturated conditions, NO_X reductions can actually increase local ozone under certain circumstances, but overall ozone production (considering downwind formation) decreases. Even in VOC-limited areas, NO_X reductions are not expected to increase ozone levels if the NO_X reductions are sufficiently large - large enough to become NO_X-limited.

4.1.1.2 Health Effects Associated with Exposure to Ozone

This section provides a summary of the health effects associated with exposure to ambient concentrations of ozone.[B] The information in this section is based on the information and conclusions in the April 2020 Integrated Science Assessment for Ozone (Ozone ISA).[7] The Ozone ISA concludes that human exposures to ambient concentrations of ozone are associated with a number of adverse health effects and characterizes the weight of evidence for these health effects. [C] The discussion below highlights the Ozone ISA's conclusions pertaining to health effects associated with both short-term and long-term periods of exposure to ozone.

For short-term exposure to ozone, the Ozone ISA concludes that respiratory effects, including lung function decrements, pulmonary inflammation, exacerbation of asthma, respiratory-related hospital admissions, and mortality, are causally associated with ozone exposure. It also concludes that metabolic effects, including metabolic syndrome (i.e., changes in insulin or glucose levels, cholesterol levels, obesity and blood pressure) and complications due to diabetes are likely to be causally associated with short-term exposure to ozone. The evidence is also suggestive of a causal relationship between short-term exposure to ozone and cardiovascular effects, central nervous system effects, and total mortality.

For long-term exposure to ozone, the Ozone ISA concludes that respiratory effects, including new onset asthma, pulmonary inflammation, and injury, are likely to be causally related with ozone exposure. The Ozone ISA characterizes the evidence as suggestive of a causal relationship for associations between long-term ozone exposure and cardiovascular effects, metabolic effects, reproductive and developmental effects, central nervous system effects and total mortality. The

[B] Human exposure to ozone varies over time due to changes in ambient ozone concentration and because people move between locations which have notable different ozone concentrations. Also, the amount of ozone delivered to the lung is not only influenced by the ambient concentrations but also by the breathing route and rate.

[C] The ISA evaluates evidence and draws conclusions on the causal relationship between relevant pollutant exposures and health effects, assigning one of five "weight of evidence" determinations: causal relationship, likely to be a causal relationship, suggestive of a causal relationship, inadequate to infer a causal relationship, and not likely to be a causal relationship. For more information on these levels of evidence, please refer to Table II in the Preamble of the ISA.

evidence is inadequate to infer a causal relationship between chronic ozone exposure and increased risk of cancer.

Finally, interindividual variation in human responses to ozone exposure can result in some groups being at increased risk for detrimental effects in response to exposure. In addition, some groups are at increased risk of exposure due to their activities, such as outdoor workers and children. The Ozone ISA identified several groups that are at increased risk for ozone-related health effects. These groups are people with asthma, children and older adults, individuals with reduced intake of certain nutrients (i.e., Vitamins C and E), outdoor workers, and individuals having certain genetic variants related to oxidative metabolism or inflammation. Ozone exposure during childhood can have lasting effects through adulthood. Such effects include altered function of the respiratory and immune systems. Children absorb higher doses (normalized to lung surface area) of ambient ozone, compared to adults, due to their increased time spent outdoors, higher ventilation rates relative to body size, and a tendency to breathe a greater fraction of air through the mouth.[D] Children also have a higher asthma prevalence compared to adults. Recent epidemiologic studies provide generally consistent evidence that long-term ozone exposure is associated with the development of asthma in children. Studies comparing age groups reported higher magnitude associations for short-term ozone exposure and respiratory hospital admissions and emergency room visits among children than among adults. Panel studies also provide support for experimental studies with consistent associations between short-term ozone exposure and lung function and pulmonary inflammation in healthy children. Additional children's vulnerability and susceptibility factors are listed in Section XIII.B of the Preamble.

4.1.2 **Particulate Matter**

4.1.2.1 Background on Particulate Matter

Particulate matter (PM) is a complex mixture of solid particles and liquid droplets distributed among numerous atmospheric gases which interact with solid and liquid phases. Particles in the atmosphere range in size from less than 0.01 to more than 10 micrometers (μm) in diameter.[8] Atmospheric particles can be grouped into several classes according to their aerodynamic diameter and physical sizes. Generally, the three broad classes of particles include ultrafine particles (UFPs, generally considered as particles with a diameter less than or equal to 0.1 μm [typically based on physical size, thermal diffusivity or electrical mobility]), "fine" particles ($PM_{2.5}$; particles with a nominal mean aerodynamic diameter less than or equal to 2.5 μm), and "thoracic" particles (PM_{10}; particles with a nominal mean aerodynamic diameter less than or equal to 10 μm). Particles that fall within the size range between $PM_{2.5}$ and PM_{10}, are referred to as "thoracic coarse particles" ($PM_{10-2.5}$, particles with a nominal mean aerodynamic diameter

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[D] Children are more susceptible than adults to many air pollutants because of differences in physiology, higher per body weight breathing rates and consumption, rapid development of the brain and bodily systems, and behaviors that increase chances for exposure. Even before birth, the developing fetus may be exposed to air pollutants through the mother that affect development and permanently harm the individual.
Infants and children breathe at much higher rates per body weight than adults, with infants under one year of age having a breathing rate up to five times that of adults. In addition, children breathe through their mouths more than adults and their nasal passages are less effective at removing pollutants, which leads to a higher deposition fraction in their lungs.

greater than 2.5 µm and less than or equal to 10 µm). EPA currently has standards that regulate $PM_{2.5}$ and PM_{10}.[E]

Most particles are found in the lower troposphere, where they can have residence times ranging from a few hours to weeks. Particles are removed from the atmosphere by wet deposition, such as when they are carried by rain or snow, or by dry deposition, when particles settle out of suspension due to gravity. Atmospheric lifetimes are generally longest for $PM_{2.5}$, which often remains in the atmosphere for days to weeks before being removed by wet or dry deposition.[9] In contrast, atmospheric lifetimes for UFP and $PM_{10-2.5}$ are shorter. Within hours, UFP can undergo coagulation and condensation that lead to formation of larger particles, or can be removed from the atmosphere by evaporation, deposition, or reactions with other atmospheric components. $PM_{10-2.5}$ are also generally removed from the atmosphere within hours, through wet or dry deposition.[10]

Particulate matter consists of both primary and secondary particles. Primary particles are emitted directly from sources, such as combustion-related activities (e.g., industrial activities, motor vehicle operation, biomass burning), while secondary particles are formed through atmospheric chemical reactions of gaseous precursors (e.g., sulfur oxides (SO_X), NO_X and VOCs).

4.1.2.2 Health Effects Associated with Exposure to Particulate Matter

Scientific evidence spanning animal toxicological, controlled human exposure, and epidemiologic studies shows that exposure to ambient PM is associated with a broad range of health effects. These health effects are discussed in detail in the Integrated Science Assessment for Particulate Matter, which was finalized in December 2019 (PM ISA).[11] In addition, there is a more targeted evaluation of studies published since the literature cutoff date of the 2019 PM ISA in the Supplement to the Integrated Science Assessment for PM (Supplement).[12] The PM ISA characterizes the causal nature of relationships between PM exposure and broad health categories (e.g., cardiovascular effects, respiratory effects, etc.) using a weight-of-evidence approach.[F] Within this characterization, the PM ISA summarizes the health effects evidence for short-term (i.e., hours up to one month) and long-term (i.e., one month to years) exposures to $PM_{2.5}$, $PM_{10-2.5}$, and ultrafine particles, and concludes that exposures to ambient $PM_{2.5}$ are associated with a number of adverse health effects. The discussion below highlights the PM ISA's conclusions,

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[E] Regulatory definitions of PM size fractions, and information on reference and equivalent methods for measuring PM in ambient air, are provided in 40 CFR parts 50, 53, and 58. With regard to national ambient air quality standards (NAAQS) which provide protection against health and welfare effects, the 24-hour PM_{10} standard provides protection against effects associated with short-term exposure to thoracic coarse particles (i.e., $PM_{10-2.5}$).
[F] The causal framework draws upon the assessment and integration of evidence from across scientific disciplines, spanning atmospheric chemistry, exposure, dosimetry and health effects studies (i.e., epidemiologic, controlled human exposure, and animal toxicological studies), and assess the related uncertainties and limitations that ultimately influence our understanding of the evidence. This framework employs a five-level hierarchy that classifies the overall weight-of-evidence with respect to the causal nature of relationships between criteria pollutant exposures and health and welfare effects using the following categorizations: causal relationship; likely to be causal relationship; suggestive of, but not sufficient to infer, a causal relationship; inadequate to infer the presence or absence of a causal relationship; and not likely to be a causal relationship (U.S. EPA. (2019). Integrated Science Assessment for Particulate Matter (Final Report). U.S. Environmental Protection Agency, Washington, DC, EPA/600/R-19/188, Section P. 3.2.3).

and summarizes additional information from the Supplement where appropriate, pertaining to the health effects evidence for both short- and long-term PM exposures. Further discussion of PM-related health effects can also be found in the 2022 Policy Assessment for the review of the PM NAAQS.[13]

EPA has concluded that recent evidence in combination with evidence evaluated in the 2009 PM ISA supports a "causal relationship" between both long- and short-term exposures to $PM_{2.5}$ and premature mortality and cardiovascular effects and a "likely to be causal relationship" between long- and short-term $PM_{2.5}$ exposures and respiratory effects.[14] Additionally, recent experimental and epidemiologic studies provide evidence supporting a "likely to be causal relationship" between long-term $PM_{2.5}$ exposure and nervous system effects, and long-term $PM_{2.5}$ exposure and cancer. Because of remaining uncertainties and limitations in the evidence base, EPA determined the evidence is "suggestive of, but not sufficient to infer, a causal relationship" for long-term $PM_{2.5}$ exposure and reproductive and developmental effects (i.e., male/female reproduction and fertility; pregnancy and birth outcomes), long- and short-term exposures and metabolic effects, and short-term exposure and nervous system effects.

As discussed extensively in the 2019 PM ISA and the Supplement, recent studies continue to support a "causal relationship" between short- and long-term $PM_{2.5}$ exposures and mortality.[15] For short-term $PM_{2.5}$ exposure, multi-city studies evaluated in the PM ISA, in combination with single- and multi-city studies evaluated in the 2009 PM ISA, provide evidence of consistent, positive associations across studies conducted in different geographic locations, populations with different demographic characteristics, and studies using different exposure assignment techniques. Additionally, the consistent and coherent evidence across scientific disciplines for cardiovascular morbidity, particularly ischemic events and heart failure, and to a lesser degree for respiratory morbidity, including exacerbations of chronic obstructive pulmonary disease (COPD) and asthma, provide biological plausibility for cause-specific mortality and ultimately total mortality. Recent epidemiologic studies evaluated in the Supplement, including studies that employed alternative methods for confounder control, provide additional support to the evidence base that contributed to the 2019 PM ISA conclusion for short-term $PM_{2.5}$ exposure and mortality.

The 2019 PM ISA concluded a "causal relationship" between long-term $PM_{2.5}$ exposure and mortality. In addition to reanalyses and extensions of the American Cancer Society (ACS) and Harvard Six Cities (HSC) cohorts, multiple new cohort studies conducted in the U.S. and Canada, consisting of people employed in a specific job (e.g., teacher, nurse) and that apply different exposure assignment techniques, provide evidence of positive associations between long-term $PM_{2.5}$ exposure and mortality. Biological plausibility for mortality due to long-term $PM_{2.5}$ exposure is provided by the coherence of effects across scientific disciplines for cardiovascular morbidity, particularly for coronary heart disease, stroke and atherosclerosis, and for respiratory morbidity, particularly for the development of COPD. Additionally, recent studies provide evidence indicating that as long-term $PM_{2.5}$ concentrations decrease there is an increase in life expectancy. Recent cohort studies evaluated in the Supplement, as well as epidemiologic studies that conducted accountability analyses or employed alternative methods for confounder controls, support and extend the evidence base that contributed to the 2019 PM ISA conclusion for long-term $PM_{2.5}$ exposure and mortality.

A large body of studies examining both short- and long-term $PM_{2.5}$ exposure and cardiovascular effects builds on the evidence base evaluated in the 2009 PM ISA. The strongest evidence for cardiovascular effects in response to short-term $PM_{2.5}$ exposures is for ischemic heart disease and heart failure. The evidence for short-term $PM_{2.5}$ exposure and cardiovascular effects is coherent across scientific disciplines and supports a continuum of effects ranging from subtle changes in indicators of cardiovascular health to serious clinical events, such as increased emergency department visits and hospital admissions due to cardiovascular disease and cardiovascular mortality. For long-term $PM_{2.5}$ exposure, there is strong and consistent epidemiologic evidence of a relationship with cardiovascular mortality. This evidence is supported by epidemiologic and animal toxicological studies demonstrating a range of cardiovascular effects including coronary heart disease, stroke, impaired heart function, and subclinical markers (e.g., coronary artery calcification, atherosclerotic plaque progression), which collectively provide coherence and biological plausibility. Recent epidemiologic studies evaluated in the Supplement, as well as studies that conducted accountability analyses or employed alternative methods for confounder control, support and extend the evidence base that contributed to the 2019 PM ISA conclusion for both short- and long-term $PM_{2.5}$ exposure and cardiovascular effects.

Studies evaluated in the 2019 PM ISA continue to provide evidence of a "likely to be causal relationship" between both short- and long-term $PM_{2.5}$ exposure and respiratory effects. Epidemiologic studies provide consistent evidence of a relationship between short-term $PM_{2.5}$ exposure and asthma exacerbation in children and COPD exacerbation in adults, as indicated by increases in emergency department visits and hospital admissions, which is supported by animal toxicological studies indicating worsening allergic airways disease and subclinical effects related to COPD. Epidemiologic studies also provide evidence of a relationship between short-term $PM_{2.5}$ exposure and respiratory mortality. However, there is inconsistent evidence for respiratory effects, specifically lung function declines and pulmonary inflammation, in controlled human exposure studies. With respect to long term $PM_{2.5}$ exposure, epidemiologic studies conducted in the U.S. and abroad provide evidence of a relationship with respiratory effects, including consistent changes in lung function and lung function growth rate, increased asthma incidence, asthma prevalence, and wheeze in children; acceleration of lung function decline in adults; and respiratory mortality. The epidemiologic evidence is supported by animal toxicological studies, which provide coherence and biological plausibility for a range of effects including impaired lung development, decrements in lung function growth, and asthma development.

Since the 2009 PM ISA, a growing body of scientific evidence examined the relationship between long-term $PM_{2.5}$ exposure and nervous system effects, resulting for the first time in a causality determination for this health effects category of a "likely to be causal relationship". The strongest evidence for effects on the nervous system come from epidemiologic studies that consistently report cognitive decrements and reductions in brain volume in adults. The effects observed in epidemiologic studies in adults are supported by animal toxicological studies demonstrating effects on the brain of adult animals including inflammation, morphologic changes, and neurodegeneration of specific regions of the brain. There is more limited evidence for neurodevelopmental effects in children with some studies reporting positive associations with autism spectrum disorder and others providing limited evidence of an association with cognitive function. While there is some evidence from animal toxicological studies indicating effects on the brain (i.e., inflammatory and morphological changes) to support a biologically plausible pathway for neurodevelopmental effects, epidemiologic studies are limited due to their lack of control for potential confounding by copollutants, the small number of studies conducted, and uncertainty regarding critical exposure windows.

Building off the decades of research demonstrating mutagenicity, DNA damage, and other endpoints related to genotoxicity due to whole PM exposures, recent experimental and epidemiologic studies focusing specifically on $PM_{2.5}$ provide evidence of a relationship between long-term $PM_{2.5}$ exposure and cancer. Epidemiologic studies examining long-term $PM_{2.5}$ exposure and lung cancer incidence and mortality provide evidence of generally positive associations in cohort studies spanning different populations, locations, and exposure assignment techniques. Additionally, there is evidence of positive associations with lung cancer incidence and mortality in analyses limited to never smokers. In addition, experimental and epidemiologic studies of genotoxicity, epigenetic effects, carcinogenic potential, and that $PM_{2.5}$ exhibits several characteristics of carcinogens, provide biological plausibility for cancer development. This collective body of evidence contributed to the conclusion of a "likely to be causal relationship."

For the additional health effects categories evaluated for $PM_{2.5}$ in the 2019 PM ISA, experimental and epidemiologic studies provide limited and/or inconsistent evidence of a relationship with $PM_{2.5}$ exposure. As a result, the 2019 PM ISA concluded that the evidence is "suggestive of, but not sufficient to infer a causal relationship" for short-term $PM_{2.5}$ exposure and metabolic effects and nervous system effects, and long-term $PM_{2.5}$ exposures and metabolic effects as well as reproductive and developmental effects.

In addition to evaluating the health effects attributed to short- and long-term exposure to $PM_{2.5}$, the 2019 PM ISA also conducted an extensive evaluation as to whether specific components or sources of $PM_{2.5}$ are more strongly related with specific health effects than $PM_{2.5}$ mass. An evaluation of those studies resulted in the 2019 PM ISA concluding that "many $PM_{2.5}$ components and sources are associated with many health effects, and the evidence does not indicate that any one source or component is consistently more strongly related to health effects than $PM_{2.5}$ mass."[16]

For both $PM_{10-2.5}$ and UFPs, for all health effects categories evaluated, the 2019 PM ISA concluded that the evidence was "suggestive of, but not sufficient to infer, a causal relationship" or "inadequate to determine the presence or absence of a causal relationship." For $PM_{10-2.5}$, although a Federal Reference Method (FRM) was instituted in 2011 to measure $PM_{10-2.5}$ concentrations nationally, the causality determinations reflect that the same uncertainty identified in the 2009 PM ISA persists with respect to the method used to estimate $PM_{10-2.5}$ concentrations in epidemiologic studies. Specifically, across epidemiologic studies, different approaches are used to estimate $PM_{10-2.5}$ concentrations (e.g., direct measurement of $PM_{10-2.5}$, difference between PM_{10} and $PM_{2.5}$ concentrations), and it remains unclear how well correlated $PM_{10-2.5}$ concentrations are both spatially and temporally across the different methods used.

For UFPs, which have often been defined as particles <0.1 µm, the uncertainty in the evidence for the health effect categories evaluated across experimental and epidemiologic studies reflects the inconsistency in the exposure metric used (i.e., particle number concentration, surface area concentration, mass concentration) as well as the size fractions examined. In epidemiologic studies the size fraction examined can vary depending on the monitor used and exposure metric, with some studies examining number count over the entire particle size range, while experimental studies that use a particle concentrator often examine particles up to 0.3 µm. Additionally, due to the lack of a monitoring network, there is limited information on the spatial and temporal variability of UFPs within the U.S., as well as population exposures to UFPs, which adds uncertainty to epidemiologic study results.

The 2019 PM ISA cites extensive evidence indicating that “both the general population as well as specific populations and lifestages are at risk for $PM_{2.5}$-related health effects.”[17] For example, in support of its “causal” and “likely to be causal” determinations, the ISA cites substantial evidence for (1) PM-related mortality and cardiovascular effects in older adults; (2) PM-related cardiovascular effects in people with pre-existing cardiovascular disease; (3) PM-related respiratory effects in people with pre-existing respiratory disease, particularly asthma exacerbations in children; and (4) PM-related impairments in lung function growth and asthma development in children. The ISA additionally notes that stratified analyses (i.e., analyses that directly compare PM-related health effects across groups) provide strong evidence for racial and ethnic differences in $PM_{2.5}$ exposures and in the risk of $PM_{2.5}$-related health effects, specifically within Hispanic and non-Hispanic Black populations with some evidence of increased risk for populations of low socioeconomic status. Recent studies evaluated in the Supplement support the conclusion of the 2019 PM ISA with respect to disparities in both $PM_{2.5}$ exposure and health risk by race and ethnicity and provide additional support for disparities for populations of lower socioeconomic status. Additionally, evidence spanning epidemiologic studies that conducted stratified analyses, experimental studies focusing on animal models of disease or individuals with pre-existing disease, dosimetry studies, as well as studies focusing on differential exposure suggest that populations with pre-existing cardiovascular or respiratory disease, populations that are overweight or obese, populations that have particular genetic variants, and current/former smokers could be at increased risk for adverse $PM_{2.5}$-related health effects. The 2022 Policy Assessment for the review of the PM NAAQS also highlights that factors that may contribute to increased risk of $PM_{2.5}$-related health effects include lifestage (children and older adults), pre-existing diseases (cardiovascular disease and respiratory disease), race/ethnicity, and socioeconomic status.[18]

4.1.3 **Nitrogen Oxides**

4.1.3.1 Background on Nitrogen Oxides

Oxides of nitrogen (NO_X) refers to nitric oxide (NO) and nitrogen dioxide (NO_2). Most NO_2 is formed in the air through the oxidation of nitric oxide (NO) that is emitted when fuel is burned at a high temperature. NO_X is a major contributor to secondary $PM_{2.5}$ formation, and NO_X along with VOCs are the two major precursors of ozone. The health effects of PM and ozone are discussed in Sections 4.1.1and 4.1.2 respectively.

4.1.3.2 Health Effects Associated with Exposure to Nitrogen Oxides

The most recent review of the health effects of oxides of nitrogen completed by EPA can be found in the 2016 Integrated Science Assessment for Oxides of Nitrogen - Health Criteria (ISA for Oxides of Nitrogen).[19] The primary source of NO_2 is motor vehicle emissions, and ambient NO_2 concentrations tend to be highly correlated with other traffic-related pollutants. Thus, a key issue in characterizing the causality of NO_2-health effect relationships consists of evaluating the extent to which studies supported an effect of NO_2 that is independent of other traffic-related pollutants. EPA concluded that the findings for asthma exacerbation integrated from epidemiologic and controlled human exposure studies provided evidence that is sufficient to infer a causal relationship between respiratory effects and short-term NO_2 exposure. The strongest evidence supporting an independent effect of NO_2 exposure comes from controlled human exposure studies demonstrating increased airway responsiveness in individuals with asthma following ambient-relevant NO_2 exposures. The coherence of this evidence with epidemiologic findings for asthma hospital admissions and emergency department visits as well as lung function decrements and increased pulmonary inflammation in children with asthma describe a plausible pathway by which NO_2 exposure can cause an asthma exacerbation. The 2016 ISA for Oxides of Nitrogen also concluded that there is likely to be a causal relationship between long-term NO_2 exposure and respiratory effects. This conclusion is based on new epidemiologic evidence for associations of NO_2 with asthma development in children combined with biological plausibility from experimental studies.

In evaluating a broader range of health effects, the 2016 ISA for Oxides of Nitrogen concluded that evidence is "suggestive of, but not sufficient to infer, a causal relationship" between short-term NO_2 exposure and cardiovascular effects and mortality and between long-term NO_2 exposure and cardiovascular effects and diabetes, birth outcomes, and cancer. In addition, the scientific evidence is inadequate (insufficient consistency of epidemiologic and toxicological evidence) to infer a causal relationship for long-term NO_2 exposure with fertility, reproduction, and pregnancy, as well as with postnatal development. The ISA states that a key uncertainty in understanding the relationship between these non-respiratory health effects and short- or long-term exposure to NO_2 is copollutant confounding, particularly by other roadway pollutants. The available evidence for non-respiratory health effects does not adequately address whether NO_2 has an independent effect or whether it primarily represents effects related to other or a mixture of traffic-related pollutants.

The 2016 ISA for Oxides of Nitrogen concluded that people with asthma, children, and older adults are at increased risk for NO_2-related health effects. In these groups and lifestages, NO_2 is consistently related to larger effects on outcomes related to asthma exacerbation, for which there is confidence in the relationship with NO_2 exposure.

4.1.4 **Carbon Monoxide**

4.1.4.1 Background on Carbon Monoxide

Carbon monoxide (CO) is a colorless, odorless gas emitted from combustion processes. Nationally, particularly in urban areas, the majority of CO emissions to ambient air come from mobile sources.

4.1.4.2 Health Effects Associated with Exposure to Carbon Monoxide

Information on the health effects of carbon monoxide (CO) can be found in the January 2010 Integrated Science Assessment for Carbon Monoxide (CO ISA).[20] The CO ISA presents conclusions regarding the presence of causal relationships between CO exposure and categories of adverse health effects.[G] This section provides a summary of the health effects associated with exposure to ambient concentrations of CO, along with the CO ISA conclusions.[H]

Controlled human exposure studies of subjects with coronary artery disease show a decrease in the time to onset of exercise-induced angina (chest pain) and electrocardiogram changes following CO exposure. In addition, epidemiologic studies observed associations between short-term CO exposure and cardiovascular morbidity, particularly increased emergency room visits and hospital admissions for coronary heart disease (including ischemic heart disease, myocardial infarction, and angina). Some epidemiologic evidence is also available for increased hospital admissions and emergency room visits for congestive heart failure and cardiovascular disease as a whole. The CO ISA concludes that a causal relationship is likely to exist between short-term exposures to CO and cardiovascular morbidity. It also concludes that available data are inadequate to conclude that a causal relationship exists between long-term exposures to CO and cardiovascular morbidity.

Animal studies show various neurological effects with in-utero CO exposure. Controlled human exposure studies report central nervous system and behavioral effects following low-level CO exposures, although the findings have not been consistent across all studies. The CO ISA concludes that the evidence is suggestive of a causal relationship with both short- and long-term exposure to CO and central nervous system effects.

A number of studies cited in the CO ISA have evaluated the role of CO exposure in birth outcomes such as preterm birth or cardiac birth defects. There is limited epidemiologic evidence of a CO-induced effect on preterm births and birth defects, with weak evidence for a decrease in birth weight. Animal toxicological studies have found perinatal CO exposure to affect birth weight, as well as other developmental outcomes. The CO ISA concludes that the evidence is suggestive of a causal relationship between long-term exposures to CO and developmental effects and birth outcomes.

Epidemiologic studies provide evidence of associations between short-term CO concentrations and respiratory morbidity such as changes in pulmonary function, respiratory symptoms, and hospital admissions. A limited number of epidemiologic studies considered copollutants such as ozone, SO_2, and PM in two-pollutant models and found that CO risk estimates were generally robust, although this limited evidence makes it difficult to disentangle effects attributed to CO itself from those of the larger complex air pollution mixture. Controlled human exposure studies have not extensively evaluated the effect of CO on respiratory morbidity. Animal studies at levels of 50-100 ppm CO show preliminary evidence of altered pulmonary vascular remodeling and oxidative injury. The CO ISA concludes that the evidence is suggestive of a causal relationship between short-term CO exposure and respiratory morbidity,

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[G] The ISA evaluates the health evidence associated with different health effects, assigning one of five “weight of evidence” determinations: causal relationship, likely to be a causal relationship, suggestive of a causal relationship, inadequate to infer a causal relationship, and not likely to be a causal relationship. For definitions of these levels of evidence, please refer to Section 1.6 of the ISA.

[H] Personal exposure includes contributions from many sources, and in many different environments. Total personal exposure to CO includes both ambient and non-ambient components; and both components may contribute to adverse health effects.

and inadequate to conclude that a causal relationship exists between long-term exposure and respiratory morbidity.

Finally, the CO ISA concludes that the epidemiologic evidence is suggestive of a causal relationship between short-term concentrations of CO and mortality. Epidemiologic evidence suggests an association exists between short-term exposure to CO and mortality, but limited evidence is available to evaluate cause-specific mortality outcomes associated with CO exposure. In addition, the attenuation of CO risk estimates that was often observed in copollutant models contributes to the uncertainty as to whether CO is acting alone or as an indicator for other combustion-related pollutants. The CO ISA also concludes that there is not likely to be a causal relationship between relevant long-term exposures to CO and mortality.

4.1.5 **Diesel Exhaust**

4.1.5.1 Background on Diesel Exhaust

Diesel exhaust is a complex mixture composed of particulate matter, carbon dioxide, oxygen, nitrogen, water vapor, carbon monoxide, nitrogen compounds, sulfur compounds and numerous low-molecular-weight hydrocarbons. A number of these gaseous hydrocarbon components are individually known to be toxic, including aldehydes, benzene and 1,3-butadiene. The diesel particulate matter present in diesel exhaust consists mostly of fine particles (< 2.5 µm), of which a significant fraction is ultrafine particles (< 0.1 µm). These particles have a large surface area which makes them an excellent medium for adsorbing organics and their small size makes them highly respirable. Many of the organic compounds present in the gases and on the particles, such as polycyclic organic matter, are individually known to have mutagenic and carcinogenic properties.

Diesel exhaust varies significantly in chemical composition and particle sizes between different engine types (heavy-duty, light-duty), engine operating conditions (idle, acceleration, deceleration), and fuel formulations (high/low sulfur fuel). Also, there are emissions differences between on-road and nonroad engines because the nonroad engines are generally of older technology. After being emitted in the engine exhaust, diesel exhaust undergoes dilution as well as chemical and physical changes in the atmosphere. The lifetimes of the components present in diesel exhaust range from seconds to days.

4.1.5.2 Health Effects Associated with Exposure to Diesel Exhaust

In EPA’s 2002 Diesel Health Assessment Document (Diesel HAD), exposure to diesel exhaust was classified as likely to be carcinogenic to humans by inhalation from environmental exposures, in accordance with the revised draft 1996/1999 EPA cancer guidelines.[21,22] A number of other agencies (National Institute for Occupational Safety and Health, the International Agency for Research on Cancer, the World Health Organization, California EPA, and the U.S. Department of Health and Human Services) made similar hazard classifications prior to 2002. EPA also concluded in the 2002 Diesel HAD that it was not possible to calculate a cancer unit risk for diesel exhaust due to limitations in the exposure data for the occupational groups or the absence of a dose-response relationship.

In the absence of a cancer unit risk, the Diesel HAD sought to provide additional insight into the significance of the diesel exhaust cancer hazard by estimating possible ranges of risk that might be present in the population. An exploratory analysis was used to characterize a range of possible lung cancer risk. The outcome was that environmental risks of cancer from long-term diesel exhaust exposures could plausibly range from as low as 10^{-5} to as high as 10^{-3}. Because of uncertainties, the analysis acknowledged that the risks could be lower than 10^{-5}, and a zero risk from diesel exhaust exposure could not be ruled out.

Noncancer health effects of acute and chronic exposure to diesel exhaust emissions are also of concern to EPA. EPA derived a diesel exhaust reference concentration (RfC) from consideration of four well-conducted chronic rat inhalation studies showing adverse pulmonary effects. The RfC is 5 $\mu g/m^3$ for diesel exhaust measured as diesel particulate matter. This RfC does not consider allergenic effects such as those associated with asthma or immunologic or the potential for cardiac effects. There was emerging evidence in 2002, discussed in the Diesel HAD, that exposure to diesel exhaust can exacerbate these effects, but the exposure-response data were lacking at that time to derive an RfC based on these then-emerging considerations. The Diesel HAD states, "With [diesel particulate matter] being a ubiquitous component of ambient PM, there is an uncertainty about the adequacy of the existing [diesel exhaust] noncancer database to identify all of the pertinent [diesel exhaust]-caused noncancer health hazards." The Diesel HAD also notes "that acute exposure to [diesel exhaust] has been associated with irritation of the eye, nose, and throat, respiratory symptoms (cough and phlegm), and neurophysiological symptoms such as headache, lightheadedness, nausea, vomiting, and numbness or tingling of the extremities." The Diesel HAD notes that the cancer and noncancer hazard conclusions applied to the general use of diesel engines then on the market and as cleaner engines replace a substantial number of existing ones, the applicability of the conclusions would need to be reevaluated.

It is important to note that the Diesel HAD also briefly summarizes health effects associated with ambient PM and discusses EPA's then-annual $PM_{2.5}$ NAAQS of 15 $\mu g/m^3$.[I] There is a large and extensive body of human data showing a wide spectrum of adverse health effects associated with exposure to ambient PM, of which diesel exhaust is an important component. The $PM_{2.5}$ NAAQS is designed to provide protection from the noncancer health effects and premature mortality attributed to exposure to $PM_{2.5}$. The contribution of diesel PM to total ambient PM varies in different regions of the country and also, within a region, from one area to another. The contribution can be high in near-roadway environments, for example, or in other locations where diesel engine use is concentrated.

Since 2002, several new studies have been published which continue to report increased lung cancer risk associated with occupational exposure to diesel exhaust from older engines. Of particular note since 2011 are three new epidemiology studies that have examined lung cancer in occupational populations, including, truck drivers, underground nonmetal miners, and other diesel motor-related occupations. These studies reported increased risk of lung cancer related to exposure to diesel exhaust, with evidence of positive exposure-response relationships to varying degrees.[23,24,25] These newer studies (along with others that have appeared in the scientific literature) add to the evidence EPA evaluated in the 2002 Diesel HAD and further reinforce the concern that diesel exhaust exposure likely poses a lung cancer hazard. The findings from these newer studies do not necessarily apply to newer technology diesel engines (i.e., heavy-duty

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[I] See Section 6.1.2 for discussion of the current $PM_{2.5}$ NAAQS standard.

highway engines from 2007 and later model years) since the newer engines have large reductions in the emission constituents compared to older technology diesel engines.

In light of the growing body of scientific literature evaluating the health effects of exposure to diesel exhaust, in June 2012 the World Health Organization's International Agency for Research on Cancer (IARC), a recognized international authority on the carcinogenic potential of chemicals and other agents, evaluated the full range of cancer-related health effects data for diesel engine exhaust. IARC concluded that diesel exhaust should be regarded as "carcinogenic to humans."[26] This designation was an update from its 1988 evaluation that considered the evidence to be indicative of a "probable human carcinogen."

4.1.6 **Air Toxics**

Heavy-duty engine emissions contribute to ambient levels of air toxics that are known or suspected human or animal carcinogens, or that have noncancer health effects. These compounds include, but are not limited to, benzene, formaldehyde, acetaldehyde, and naphthalene. These compounds were identified as national or regional cancer risk drivers or contributors in the 2018 AirToxScreen Assessment and have significant inventory contributions from mobile sources.[27,28]

4.1.6.1 Health Effects Associated with Exposure to Benzene

EPA's Integrated Risk Information System (IRIS) database lists benzene as a known human carcinogen (causing leukemia) by all routes of exposure, and concludes that exposure is associated with additional health effects, including genetic changes in both humans and animals and increased proliferation of bone marrow cells in mice.[29,30,31] EPA states in its IRIS database that data indicate a causal relationship between benzene exposure and acute lymphocytic leukemia and suggest a relationship between benzene exposure and chronic non-lymphocytic leukemia and chronic lymphocytic leukemia. EPA's IRIS documentation for benzene also lists a range of 2.2×10^{-6} to 7.8×10^{-6} per $\mu g/m^3$ as the unit risk estimate (URE) for benzene.[J,32] The IARC has determined that benzene is a human carcinogen, and the U.S. Department of Health and Human Services (DHHS) has characterized benzene as a known human carcinogen.[33,34]

A number of adverse noncancer health effects, including blood disorders such as preleukemia and aplastic anemia, have also been associated with long-term exposure to benzene.[35,36] The most sensitive noncancer effect observed in humans, based on current data, is the depression of the absolute lymphocyte count in blood.[37,38] EPA's inhalation reference concentration (RfC) for benzene is 30 $\mu g/m^3$. The RfC is based on suppressed absolute lymphocyte counts seen in humans under occupational exposure conditions. In addition, studies sponsored by the Health Effects Institute (HEI) provide evidence that biochemical responses occur at lower levels of benzene exposure than previously known.[39,40,41,42] EPA's IRIS program has not yet evaluated these new data. EPA does not currently have an acute reference concentration for benzene. The Agency for Toxic Substances and Disease Registry (ATSDR) Minimal Risk Level (MRL) for acute exposure to benzene is 29 $\mu g/m^3$ for 1-14 days exposure.[43,K]

[J] A unit risk estimate is defined as the increase in the lifetime risk of cancer of an individual who is exposed for a lifetime to 1 $\mu g/m^3$ benzene in air.

[K] A minimal risk level (MRL) is defined as an estimate of the daily human exposure to a hazardous substance that is likely to be without appreciable risk of adverse noncancer health effects over a specified duration of exposure.

There is limited information from two studies regarding an increased risk of adverse effects to children whose parents have been occupationally exposed to benzene.[44,45] Data from animal studies have shown benzene exposures result in damage to the hematopoietic (blood cell formation) system during development.[46,47,48] Also, key changes related to the development of childhood leukemia occur in the developing fetus.[49] Several studies have reported that genetic changes related to eventual leukemia development occur before birth. For example, there is one study of genetic changes in twins who developed T cell leukemia at nine years of age.[50]

4.1.6.2 Health Effects Associated with Exposure to Formaldehyde

In 1991, EPA concluded that formaldehyde is a Class B1 probable human carcinogen based on limited evidence in humans and sufficient evidence in animals.[51] An Inhalation URE for cancer and a Reference Dose for oral noncancer effects were developed by EPA and posted on the IRIS database. Since that time, the NTP and IARC have concluded that formaldehyde is a known human carcinogen.[52,53,54]

The conclusions by IARC and NTP reflect the results of epidemiologic research published since 1991 in combination with previous animal, human and mechanistic evidence. Research conducted by the National Cancer Institute reported an increased risk of nasopharyngeal cancer and specific lymphohematopoietic malignancies among workers exposed to formaldehyde.[55,56,57] A National Institute of Occupational Safety and Health study of garment workers also reported increased risk of death due to leukemia among workers exposed to formaldehyde.[58] Extended follow-up of a cohort of British chemical workers did not report evidence of an increase in nasopharyngeal or lymphohematopoietic cancers, but a continuing statistically significant excess in lung cancers was reported.[59] Finally, a study of embalmers reported formaldehyde exposures to be associated with an increased risk of myeloid leukemia but not brain cancer.[60]

Health effects of formaldehyde in addition to cancer were reviewed by the ATSDR in 1999, supplemented in 2010, and by the World Health Organization. [61,62,63] These organizations reviewed the scientific literature concerning health effects linked to formaldehyde exposure to evaluate hazards and dose response relationships and defined exposure concentrations for minimal risk levels (MRLs). The health endpoints reviewed included sensory irritation of eyes and respiratory tract, reduced pulmonary function, nasal histopathology, and immune system effects. In addition, research on reproductive and developmental effects and neurological effects were discussed along with several studies that suggest that formaldehyde may increase the risk of asthma – particularly in the young.

In June 2010, EPA released a draft Toxicological Review of Formaldehyde – Inhalation Assessment through the IRIS program for peer review by the National Research Council (NRC) and public comment.[64] That draft assessment reviewed more recent research from animal and human studies on cancer and other health effects. The NRC released their review report in April 2011.[65] EPA's draft assessment, which addresses NRC recommendations, was suspended in 2018.[66] The draft assessment was unsuspended in March 2021, and an external review draft was released in April 2022.[67] This draft assessment is now undergoing review by the National Academy of Sciences.[68]

4.1.6.3 Health Effects Associated with Exposure to Acetaldehyde

Acetaldehyde is classified in EPA's IRIS database as a probable human carcinogen, based on nasal tumors in rats, and is considered toxic by the inhalation, oral, and intravenous routes.[69] The URE in IRIS for acetaldehyde is 2.2×10^{-6} per $\mu g/m^3$.[70] Acetaldehyde is reasonably anticipated to be a human carcinogen by the NTP in the 14th Report on Carcinogens and is classified as possibly carcinogenic to humans (Group 2B) by the IARC.[71,72]

The primary noncancer effects of exposure to acetaldehyde vapors include irritation of the eyes, skin, and respiratory tract.[73] In short-term (4 week) rat studies, degeneration of olfactory epithelium was observed at various concentration levels of acetaldehyde exposure.[74,75] Data from these studies were used by EPA to develop an inhalation reference concentration of 9 $\mu g/m^3$. Some asthmatics have been shown to be a sensitive subpopulation to decrements in functional expiratory volume (FEV1 test) and bronchoconstriction upon acetaldehyde inhalation.[76] Children, especially those with diagnosed asthma, may be more likely to show impaired pulmonary function and symptoms of asthma than are adults following exposure to acetaldehyde.[77]

4.1.6.4 Health Effects Associated with Exposure to Naphthalene

Naphthalene is found in small quantities in gasoline and diesel fuels. Naphthalene emissions have been measured in larger quantities in both gasoline and diesel exhaust compared with evaporative emissions from mobile sources, indicating it is primarily a product of combustion.

Acute (short-term) exposure of humans to naphthalene by inhalation, ingestion, or dermal contact is associated with hemolytic anemia and damage to the liver and the nervous system.[78] Chronic (long term) exposure of workers and rodents to naphthalene has been reported to cause cataracts and retinal damage.[79] Children, especially neonates, appear to be more susceptible to acute naphthalene poisoning based on the number of reports of lethal cases in children and infants (hypothesized to be due to immature naphthalene detoxification pathways).[80] EPA released an external review draft of a reassessment of the inhalation carcinogenicity of naphthalene based on a number of recent animal carcinogenicity studies.[81] The draft reassessment completed external peer review.[82] Based on external peer review comments received, EPA is developing a revised draft assessment that considers inhalation and oral routes of exposure, as well as cancer and noncancer effects.[83] The external review draft does not represent official agency opinion and was released solely for the purposes of external peer review and public comment. The NTP listed naphthalene as "reasonably anticipated to be a human carcinogen" in 2004 on the basis of bioassays reporting clear evidence of carcinogenicity in rats and some evidence of carcinogenicity in mice.[84] California EPA has released a new risk assessment for naphthalene, and the IARC has reevaluated naphthalene and re-classified it as Group 2B: possibly carcinogenic to humans.[85]

Naphthalene also causes a number of non-cancer effects in animals following chronic and less-than-chronic exposure, including abnormal cell changes and growth in respiratory and nasal tissues.[86] The current EPA IRIS assessment includes noncancer data on hyperplasia and metaplasia in nasal tissue that form the basis of the inhalation RfC of 3 $\mu g/m^3$.[87] The ATSDR MRL for acute and intermediate duration oral exposure to naphthalene is 0.6 mg/kg/day based on maternal toxicity in a developmental toxicology study in rats.[88] ATSDR also derived an ad hoc reference value of 6×10^{-2} mg/m^3 for acute (≤24-hour) inhalation exposure to naphthalene in a Letter Health Consultation dated March 24, 2014 to address a potential exposure concern in Illinois.[89] The ATSDR acute inhalation reference value was based on a qualitative identification of an exposure level interpreted not to cause pulmonary lesions in mice. More recently, EPA developed acute RfCs for 1-, 8-, and 24-hour exposure scenarios; the ≤24-hour reference value is 2×10^{-2} mg/m^3.[90] EPA's acute RfCs are based on a systematic review of the literature, benchmark dose modeling of naphthalene-induced nasal lesions in rats, and application of a PBPK (physiologically based pharmacokinetic) model.

4.1.6.5 Health Effects Associated with Exposure to Other Air Toxics

In addition to the compounds described above, other compounds found in gaseous hydrocarbon and PM emissions from engines will be affected by this rule. Mobile source air toxics that will potentially be affected include acrolein, ethylbenzene, propionaldehyde, toluene, and xylene. Information regarding the health effects of these compounds can be found in EPA's IRIS database.[91]

4.1.7 <u>**Exposure and Health Effects Associated with Traffic**</u>

Locations in close proximity to major roadways generally have elevated concentrations of many air pollutants emitted from motor vehicles. Hundreds of studies have been published in peer-reviewed journals, concluding that concentrations of CO, CO_2, NO, NO_2, benzene, aldehydes, PM, black carbon, and many other compounds are elevated in ambient air within approximately 300-600 meters (about 1,000-2,000 feet) of major roadways. The highest concentrations of most pollutants emitted directly by motor vehicles are found at locations within 50 meters (about 165 feet) of the edge of a roadway's traffic lanes.

A large-scale review of air quality measurements in the vicinity of major roadways between 1978 and 2008 concluded that the pollutants with the steepest concentration gradients in vicinities of roadways were CO, UFPs, metals, elemental carbon (EC), NO, NO_X, and several VOCs.[92] These pollutants showed a large reduction in concentrations within 100 meters downwind of the roadway. Pollutants that showed more gradual reductions with distance from roadways included benzene, NO_2, $PM_{2.5}$, and PM_{10}. In reviewing the literature, Karner et al., (2010) reported that results varied based on the method of statistical analysis used to determine the gradient in pollutant concentration. More recent studies continue to show significant concentration gradients of traffic-related air pollution around major roads.[93,94,95,96,97; 98,99,100] There is evidence that EPA's regulations for vehicles have lowered the near-road concentrations and gradients.[101] Starting in 2010, EPA required through the NAAQS process that air quality monitors be placed near high-traffic roadways for determining concentrations of CO, NO_2, and $PM_{2.5}$ (in addition to those existing monitors located in neighborhoods and other locations farther away from pollution sources). The monitoring data for NO_2 indicate that in urban areas, monitors near roadways often report the highest concentrations of NO_2.[102] More recent studies of traffic-related air pollutants continue to report sharp gradients around roadways, particularly within several hundred meters.[103,104]

For pollutants with relatively high background concentrations relative to near-road concentrations, detecting concentration gradients can be difficult. For example, many carbonyls have high background concentrations as a result of photochemical breakdown of precursors from many different organic compounds. However, several studies have measured carbonyls in multiple weather conditions and found higher concentrations of many carbonyls downwind of roadways.[105,106] These findings suggest a substantial roadway source of these carbonyls.

In the past 30 years, many studies have been published with results reporting that populations who live, work, or go to school near high-traffic roadways experience higher rates of numerous adverse health effects, compared to populations far away from major roads.[L] In addition, numerous studies have found adverse health effects associated with spending time in traffic, such as commuting or walking along high-traffic roadways, including studies among children.[107,108,109,110] The health outcomes with the strongest evidence linking them with traffic-associated air pollutants are respiratory effects, particularly in asthmatic children, and cardiovascular effects. Commenters on the NPRM stressed the importance of consideration of the impacts of traffic-related air pollution, especially NOx, on children's health.

Numerous reviews of this body of health literature have been published. In a 2022 final report, an expert panel of the Health Effects Institute (HEI) employed a systematic review focusing on selected health endpoints related to exposure to traffic-related air pollution.[111] The HEI panel concluded that there was a high level of confidence in evidence between long-term exposure to traffic-related air pollution and health effects in adults, including all-cause, circulatory, and ischemic heart disease mortality.[112] The panel also found that there is a moderate-to-high level of confidence in evidence of associations with asthma onset and acute respiratory infections in children and lung cancer and asthma onset in adults. This report follows on an earlier expert review published by HEI in 2010, where it found strongest evidence for asthma-related traffic impacts. Other literature reviews have been published with conclusions generally similar to the HEI panels'.[113,114,115,116] Additionally, in 2014, researchers from the U.S. Centers for Disease Control and Prevention (CDC) published a systematic review and meta-analysis of studies evaluating the risk of childhood leukemia associated with traffic exposure and reported positive associations between "postnatal" proximity to traffic and leukemia risks, but no such association for "prenatal" exposures.[117] The U.S. Department of Health and Human Services' National Toxicology Program (NTP) published a monograph including a systematic review of traffic-related air pollution and its impacts on hypertensive disorders of pregnancy. The NTP concluded that exposure to traffic-related air pollution is "presumed to be a hazard to pregnant women" for developing hypertensive disorders of pregnancy.[118]

Health outcomes with few publications suggest the possibility of other effects still lacking sufficient evidence to draw definitive conclusions. Among these outcomes with a small number of positive studies are neurological impacts (e.g., autism and reduced cognitive function) and reproductive outcomes (e.g., preterm birth, low birth weight).[119,120,121,122,123]

In addition to health outcomes, particularly cardiopulmonary effects, conclusions of numerous studies suggest mechanisms by which traffic-related air pollution affects health. For example, numerous studies indicate that near-roadway exposures may increase systemic inflammation, affecting organ systems, including blood vessels and lungs.[124,125,126,127] Additionally, long-term

[L] In the widely used PubMed database of health publications, between January 1, 1990 and December 31, 2021, 1,979 publications contained the keywords "traffic, pollution, epidemiology," with approximately half the studies published after 2015.

exposures in near-road environments have been associated with inflammation-associated conditions, such as atherosclerosis and asthma.[128,129,130]

Several studies suggest that some factors may increase susceptibility to the effects of traffic-associated air pollution. Several studies have found stronger adverse health associations in children experiencing chronic social stress, such as in violent neighborhoods or in homes with low incomes or high family stress.[131,132,133,134]

The risks associated with residence, workplace, or schools near major roads are of potentially high public health significance due to the large population in such locations. The 2013 U.S. Census Bureau's American Housing Survey (AHS) was the last AHS that included whether housing units were within 300 feet of an "airport, railroad, or highway with four or more lanes."[M] The 2013 survey reports that 17.3 million housing units, or 13 percent of all housing units in the U.S., were in such areas. Assuming that populations and housing units are in the same locations, this corresponds to a population of more than 41 million U.S. residents in close proximity to high-traffic roadways or other transportation sources. According to the Central Intelligence Agency's World Factbook, based on data collected between 2012-2014, the United States had 6,586,610 km of roadways, 293,564 km of railways, and 13,513 airports. As such, highways represent the overwhelming majority of transportation facilities described by this factor in the AHS.

EPA also conducted a study to estimate the number of people living near truck freight routes in the United States.[135] Based on a population analysis using the U.S. Department of Transportation's (USDOT) Freight Analysis Framework 4 (FAF4) and population data from the 2010 decennial census, an estimated 72 million people live within 200 meters of these freight routes.[N,O] In addition, relative to the rest of the population, people of color and those with lower incomes are more likely to live near FAF4 truck routes. They are also more likely to live in metropolitan areas. The EPA's Exposure Factor Handbook also indicates that, on average, Americans spend more than an hour traveling each day, bringing nearly all residents into a high-exposure microenvironment for part of the day.[136]

As described in Section 4.3, we estimate that about 10 million students attend schools within 200 meters of major roads.[137] Research into the impact of traffic-related air pollution on school performance is tentative. A review of this literature found some evidence that children exposed to higher levels of traffic-related air pollution show poorer academic performance than those exposed to lower levels of traffic-related air pollution.[138] However, this evidence was judged to be weak due to limitations in the assessment methods.

While near-roadway studies focus on residents near roads or others spending considerable time near major roads, the duration of commuting results in another important

[M] The variable was known as "ETRANS" in the questions about the neighborhood.

[N] FAF4 is a model from the USDOT's Bureau of Transportation Statistics (BTS) and Federal Highway Administration (FHWA), which provides data associated with freight movement in the U.S. It includes data from the 2012 Commodity Flow Survey (CFS), the Census Bureau on international trade, as well as data associated with construction, agriculture, utilities, warehouses, and other industries. FAF4 estimates the modal choices for moving goods by trucks, trains, boats, and other types of freight modes. It includes traffic assignments, including truck flows on a network of truck routes. *https://ops.fhwa.dot.gov/freight/freight_analysis/faf/*.

[O] The same analysis estimated the population living within 100 meters of a FAF4 truck route is 41 million.

contributor to overall exposure to traffic-related air pollution. Studies of health that address time spent in transit have found evidence of elevated risk of cardiac impacts.[139,140,141] Studies have also found that school bus emissions can increase student exposures to diesel-related air pollutants, and that programs that reduce school bus emissions may improve health and reduce school absenteeism.[142,143,144,145]

4.2 Environmental Effects Associated with Exposure to Pollutants

This section discusses the environmental effects associated with pollutants affected by this rule, specifically PM, ozone, NO_X and air toxics.

4.2.1 Visibility Degradation

Visibility can be defined as the degree to which the atmosphere is transparent to visible light.[146] Visibility impairment is caused by light scattering and absorption by suspended particles and gases. It is dominated by contributions from suspended particles except under pristine conditions. Fine particles with significant light-extinction efficiencies include sulfates, nitrates, organic carbon, elemental carbon, sea salt, and soil.[147,148] Visibility is important because it has direct significance to people's enjoyment of daily activities in all parts of the country. Individuals value good visibility for the well-being it provides them directly, where they live and work, and in places where they enjoy recreational opportunities. Visibility is also highly valued in significant natural areas, such as national parks and wilderness areas, and special emphasis is given to protecting visibility in these areas. For more information on visibility see the final 2019 PM ISA.[149]

The extent to which any amount of light extinction affects a person's ability to view a scene depends on both scene and light characteristics. For example, the appearance of a nearby object (e.g., a building) is generally less sensitive to a change in light extinction than the appearance of a similar object at a greater distance. See Figure 4-1 for an illustration of the important factors affecting visibility.[150]

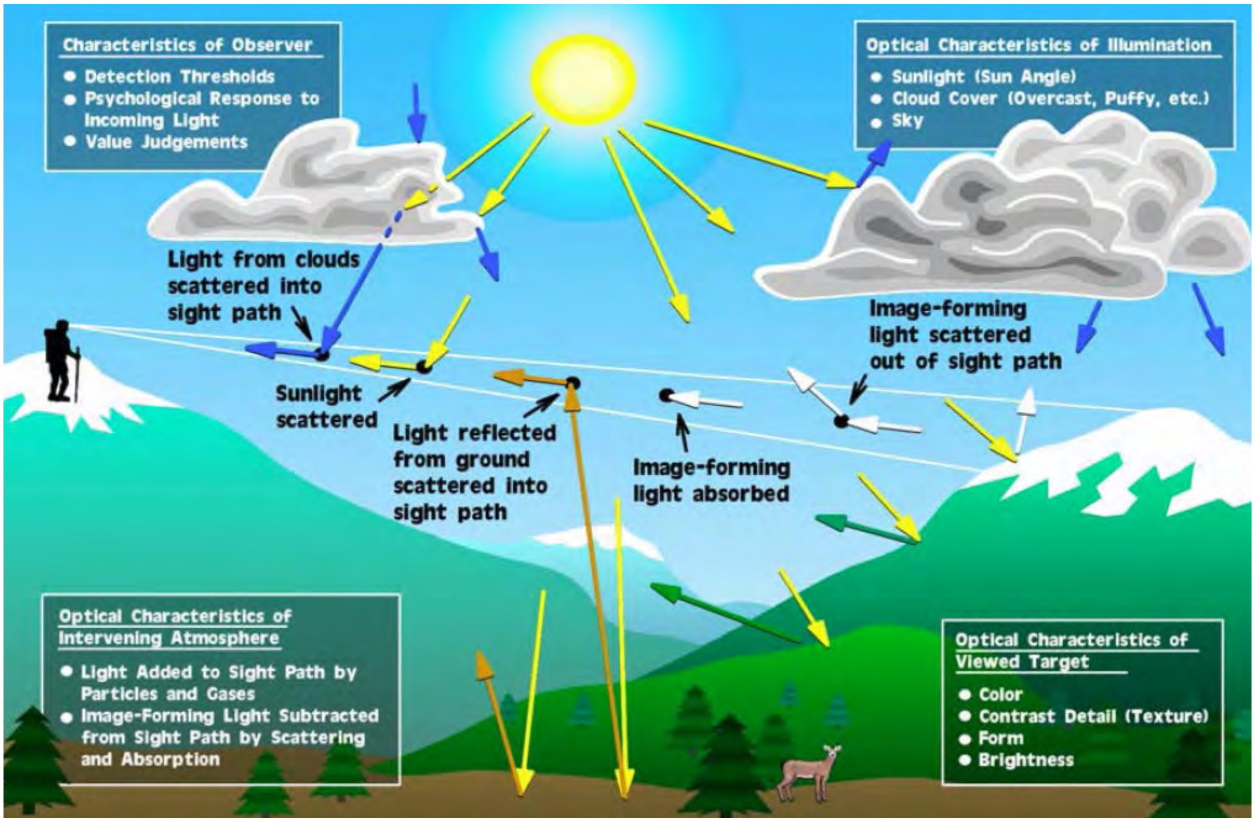

Figure 4-1: Important Factors Involved in Seeing a Scenic Vista (Malm, 2016)

EPA is working to address visibility impairment. Reductions in air pollution from implementation of various programs called for in the Clean Air Act Amendments of 1990 (CAAA) have resulted in substantial improvements in visibility and will continue to do so in the future. Nationally, because trends in haze are closely associated with trends in particulate sulfate and nitrate emissions due to the relationship between their concentration and light extinction, visibility trends have improved as emissions of SO_2 and NO_X have decreased over time due to air pollution regulations such as the Acid Rain Program.[151] However, in the western part of the country, changes in total light extinction were smaller, and the contribution of particulate organic matter to atmospheric light extinction was increasing due to increasing wildfire emissions.[152]

In the Clean Air Act Amendments of 1977, Congress recognized visibility's value to society by establishing a national goal to protect national parks and wilderness areas from visibility impairment caused by manmade pollution.[153] In 1999, EPA finalized the regional haze program (64 FR 35714) to protect the visibility in Mandatory Class I Federal areas. There are 156 national parks, forests and wilderness areas categorized as Mandatory Class I Federal areas (62 FR 38680-38681, July 18, 1997). These areas are defined in CAA Section 162 as those national parks exceeding 6,000 acres, wilderness areas and memorial parks exceeding 5,000 acres, and all international parks that were in existence on August 7, 1977. Figure 4-2 shows the location of the 156 Mandatory Class I Federal areas.

Figure 4-2: Mandatory Class I Federal Areas in the U.S.

EPA has also concluded that $PM_{2.5}$ causes adverse effects on visibility in other areas that are not targeted by the Regional Haze Rule, such as urban areas, depending on $PM_{2.5}$ concentrations and other factors such as dry chemical composition and relative humidity (i.e., an indicator of the water composition of the particles). The secondary (welfare-based) PM NAAQS provide protection against visibility effects. In recent PM NAAQS reviews, EPA evaluated a target level of protection for visibility impairment that is expected to be met through attainment of the existing secondary PM standards.

4.2.1.1 Visibility Monitoring

In conjunction with the U.S. National Park Service, the U.S. Forest Service, other Federal land managers, and State organizations in the U.S., EPA has supported visibility monitoring in national parks and wilderness areas since 1988. The monitoring network was originally established at 20 sites, but it has now been expanded to 152 sites that represent all but one of the 156 Mandatory Federal Class I areas across the country (see Figure 4-2). This long-term visibility monitoring network is known as IMPROVE (Interagency Monitoring of Protected Visual Environments).

IMPROVE provides direct measurement of particles that contribute to visibility impairment. The IMPROVE network employs aerosol measurements at all sites, and optical and scene measurements at some of the sites. Aerosol measurements are taken for PM_{10} and $PM_{2.5}$ mass, and for key constituents of $PM_{2.5}$, such as sulfate, nitrate, organic and elemental carbon (OC and EC), and other elements that can be used to estimate soil dust and sea salt contributions. Measurements for specific aerosol constituents are used to calculate "reconstructed" aerosol light extinction by multiplying the mass for each constituent by its empirically-derived scattering and/or absorption efficiency, with adjustment for the relative humidity. The IMPROVE program utilizes both an “original” and a “revised” reconstruction formula for this purpose, with the latter explicitly accounting for sea salt concentrations. Knowledge of the main constituents of a site's light extinction "budget" is critical for source apportionment and control strategy development. In addition to this indirect method of assessing light extinction, there are optical measurements which directly measure light extinction or its components. Such measurements are made principally with a nephelometer to measure light scattering; some sites also include an aethalometer for light absorption; and a few sites use a transmissometer, which measures total light extinction. Scene characteristics are typically recorded using digital or video photography and are used to determine the quality of visibility conditions (such as effects on color and contrast) associated with specific levels of light extinction as measured under both direct and aerosol-related methods. Directly measured light extinction is used under the IMPROVE protocol to cross check that total light extinction calculated from the IMPROVE reconstruction formula are consistent with directly measured extinction. Aerosol-derived light extinction from the IMPROVE equation is used to document spatial and temporal trends and to determine how changes in atmospheric constituents would affect future visibility conditions.

Annual average visibility conditions (reflecting light extinction due to both anthropogenic and non-anthropogenic sources) vary regionally across the U.S. Figures 13-1 through 13-14 in the PM ISA detail the percent contributions to particulate light extinction for ammonium nitrate and sulfate, EC and OC, and coarse mass and fine soil, by month.[154]

4.2.2 **Plant and Ecosystem Effects of Ozone**

The welfare effects of ozone include effects on ecosystems, which can be observed across a variety of scales, i.e., subcellular, cellular, leaf, whole plant, population and ecosystem. When ozone effects that begin at small spatial scales, such as the leaf of an individual plant, occur at sufficient magnitudes (or to a sufficient degree), they can result in effects being propagated along a continuum to higher and higher levels of biological organization. For example, effects at the individual plant level, such as altered rates of leaf gas exchange, growth and reproduction, can, when widespread, result in broad changes in ecosystems, such as productivity, carbon storage, water cycling, nutrient cycling, and community composition.

Ozone can produce both acute and chronic injury in sensitive plant species depending on the concentration level and the duration of the exposure.[155] In those sensitive species[P], effects from repeated exposure to ozone throughout the growing season of the plant can tend to accumulate, so that even relatively low concentrations experienced for a longer duration have the potential to create chronic stress on vegetation.[156,Q] Ozone damage to sensitive plant species includes impaired photosynthesis and visible injury to leaves. The impairment of photosynthesis, the process by which the plant makes carbohydrates (its source of energy and food), can lead to reduced crop yields, timber production, and plant productivity and growth. Impaired photosynthesis can also lead to a reduction in root growth and carbohydrate storage below ground, resulting in other, more subtle plant and ecosystems impacts.[157] These latter impacts include increased susceptibility of plants to insect attack, disease, harsh weather, interspecies

P Only a small percentage of all the plant species growing within the U.S. (over 43,000 species have been catalogued in the USDA PLANTS database) have been studied with respect to ozone sensitivity.

[Q] The concentration at which ozone levels overwhelm a plant's ability to detoxify or compensate for oxidant exposure varies. Thus, whether a plant is classified as sensitive or tolerant depends in part on the exposure levels being considered.

competition, and overall decreased plant vigor. The adverse effects of ozone on areas with sensitive species could potentially lead to species shifts and loss from the affected ecosystems[R], resulting in a loss or reduction in associated ecosystem goods and services.[158] Additionally, visible ozone injury to leaves can result in a loss of aesthetic value in areas of special scenic significance like national parks and wilderness areas and reduced use of sensitive ornamentals in landscaping.[159] In addition to ozone effects on vegetation, newer evidence suggests that ozone affects interactions between plants and insects by altering chemical signals (e.g., floral scents) that plants use to communicate to other community members, such as attraction of pollinators.

The Ozone ISA presents more detailed information on how ozone affects vegetation and ecosystems.[160] The Ozone ISA reports causal and likely causal relationships between ozone exposure and a number of welfare effects and characterizes the weight of evidence for different effects associated with ozone.[S] The Ozone ISA concludes that visible foliar injury effects on vegetation, reduced vegetation growth, reduced plant reproduction, reduced productivity in terrestrial ecosystems, reduced yield and quality of agricultural crops, alteration of below-ground biogeochemical cycles, and altered terrestrial community composition are causally associated with exposure to ozone. It also concludes that increased tree mortality, altered herbivore growth and reproduction, altered plant-insect signaling, reduced carbon sequestration in terrestrial ecosystems, and alteration of terrestrial ecosystem water cycling are likely to be causally associated with exposure to ozone.

4.2.3 **Deposition**

Deposited airborne pollutants contribute to adverse effects on ecosystems, and to soiling and materials damage. These welfare effects result mainly from exposure to excess amounts of specific chemical species, regardless of their source or predominant form (particle, gas or liquid). Nitrogen and sulfur tend to comprise a large portion of PM in many locations; however, gas-phase forms of oxidized nitrogen and sulfur also cause adverse ecological effects. The following characterizations of the nature of these environmental effects are based on information contained in the 2019 PM ISA, and the 2020 Integrated Science Assessment for Oxides of Nitrogen, Oxides of Sulfur, and Particulate Matter - Ecological Criteria.[161,162] This rule will reduce emissions of nitrogen and PM but will not change emissions of sulfur.

4.2.3.1 Deposition of Nitrogen and Sulfur

Nitrogen and sulfur interactions in the environment are highly complex, as shown in Figure 4-3.[163] Both nitrogen and sulfur are essential, and sometimes limiting, nutrients needed for growth and productivity of ecosystem components (e.g., algae, plants). In terrestrial and aquatic ecosystems, excesses of nitrogen or sulfur can lead to acidification and nutrient enrichment.[161] In addition, in aquatic ecosystems, sulfur deposition can increase mercury methylation.

[R] Per footnote above, ozone impacts could be occurring in areas where plant species sensitive to ozone have not yet been studied or identified.

[S] The Ozone ISA evaluates the evidence associated with different ozone related health and welfare effects, assigning one of five "weight of evidence" determinations: causal relationship, likely to be a causal relationship, suggestive of a causal relationship, inadequate to infer a causal relationship, and not likely to be a causal relationship. For more information on these levels of evidence, please refer to Table II of the ISA.

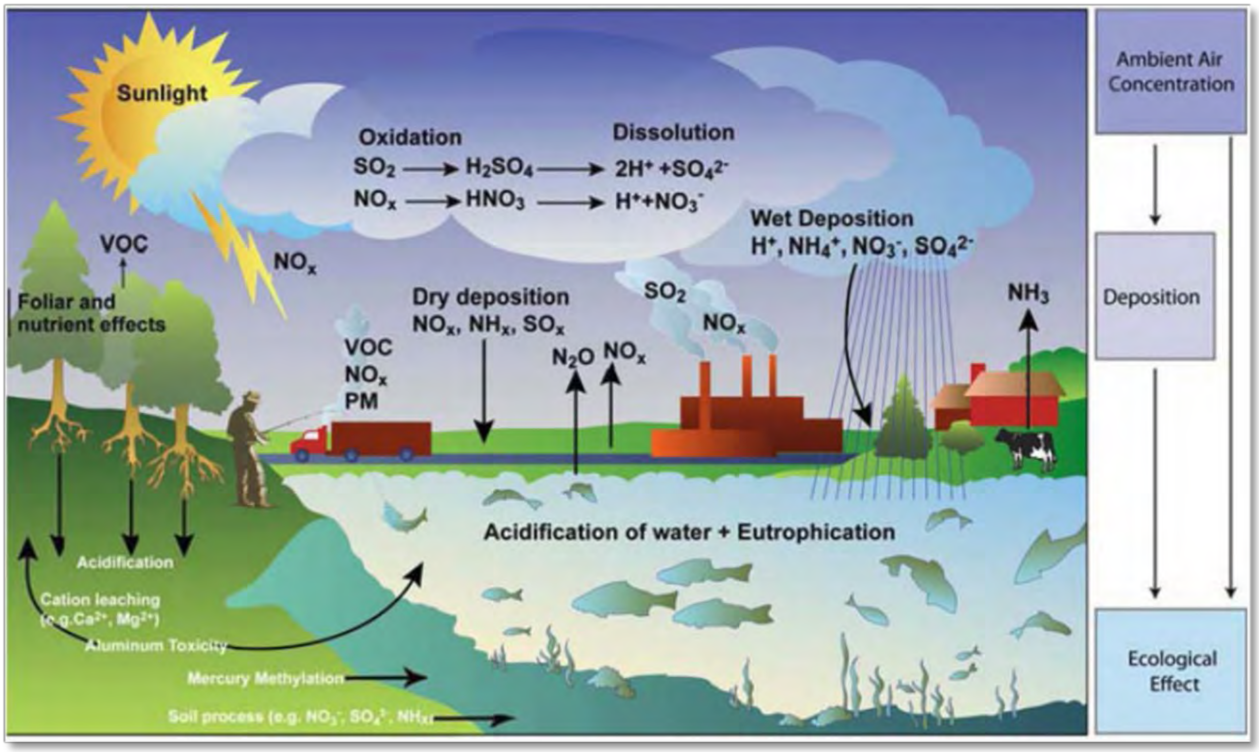

Figure 4-3: Nitrogen and Sulfur Cycling, and Interactions in the Environment

4.2.3.1.1 *Ecological Effects of Acidification*

Deposition of nitrogen and sulfur can cause acidification, which alters biogeochemistry and affects animal and plant life in terrestrial and aquatic ecosystems across the U.S. Soil acidification is a natural process, but is often accelerated by acidifying deposition, which can decrease concentrations of exchangeable base cations in soils.[161] Biological effects of acidification in terrestrial ecosystems are generally linked to aluminum toxicity and decreased ability of plant roots to take up base cations.[161] Decreases in the acid neutralizing capacity and increases in inorganic aluminum concentration contribute to declines in zooplankton, macro invertebrates, and fish species richness in aquatic ecosystems.[161]

Geology (particularly surficial geology) is the principal factor governing the sensitivity of terrestrial and aquatic ecosystems to acidification from nitrogen and sulfur deposition.[161] Geologic formations having low base cation supply generally underlie the watersheds of acid-sensitive lakes and streams. Other factors contribute to the sensitivity of soils and surface waters to acidifying deposition, including topography, soil chemistry, land use, and hydrologic flow path.[161]

4.2.3.1.1.1 *Aquatic Acidification*

Aquatic effects of acidification have been well studied in the U.S. and elsewhere at various trophic levels. These studies indicate that aquatic biota have been affected by acidification at virtually all levels of the food web in acid sensitive aquatic ecosystems. Effects have been most clearly documented for fish, aquatic insects, other invertebrates, and algae. Biological effects are primarily attributable to a combination of low pH and high inorganic aluminum concentrations. Such conditions occur more frequently during rainfall and snowmelt that cause high flows of water, and less commonly during low-flow conditions, except where chronic acidity conditions are severe. Biological effects of episodes include reduced fish condition factor, changes in species composition and declines in aquatic species richness across multiple taxa, ecosystems and regions.

Because acidification primarily affects the diversity and abundance of aquatic biota, it also affects the ecosystem services, e.g., recreational and subsistence fishing, that are derived from the fish and other aquatic life found in these surface waters. In the northeastern United States, the surface waters affected by acidification are a source of food for some recreational and subsistence fishermen and for other consumers with particularly high rates of self-caught fish consumption, such as the Hmong and Chippewa ethnic groups.[164,165]

4.2.3.1.1.2 *Terrestrial Acidification*

Acidifying deposition has altered major biogeochemical processes in the U.S. by increasing the nitrogen and sulfur content of soils, accelerating nitrate and sulfate leaching from soil to drainage waters, depleting base cations (especially calcium and magnesium) from soils, and increasing the mobility of aluminum. Inorganic aluminum is toxic to some tree roots. Plants affected by high levels of aluminum from the soil often have reduced root growth, which restricts the ability of the plant to take up water and nutrients, especially calcium.[161] These direct effects can, in turn, influence the response of these plants to climatic stresses such as droughts and cold temperatures. They can also influence the sensitivity of plants to other stresses, including insect pests and disease leading to increased mortality of canopy trees.[166] In the U.S., terrestrial effects of acidification are best described for forested ecosystems (especially red spruce and sugar maple ecosystems) with additional information on other plant communities, including shrubs and lichen.[161]

Both coniferous and deciduous forests throughout the eastern U.S. are experiencing gradual losses of base cation nutrients from the soil due to accelerated leaching from acidifying deposition. This change in nutrient availability may reduce the quality of forest nutrition over the long term. Evidence suggests that red spruce and sugar maple in some areas in the eastern U.S. have experienced declining health because of this deposition. For red spruce (Picea rubens), dieback or decline has been observed across high elevation landscapes of the northeastern U.S. and, to a lesser extent, the southeastern U.S., and acidifying deposition has been implicated as a causal factor.[167]

4.2.3.1.2 Ecological Effects from Nitrogen Enrichment

4.2.3.1.2.1 *Aquatic Enrichment*

Eutrophication in estuaries is associated with a range of adverse ecological effects including low dissolved oxygen (DO), harmful algal blooms (HABs), loss of submerged aquatic vegetation (SAV), and low water clarity. Low DO disrupts aquatic habitats, causing stress to fish and shellfish, which, in the short-term, can lead to episodic fish kills and, in the long-term, can damage overall growth in fish and shellfish populations. Low DO also degrades the aesthetic qualities of surface water. In addition to often being toxic to fish and shellfish and leading to fish kills and aesthetic impairments of estuaries, HABs can, in some instances, also be harmful to human health. SAV provides critical habitat for many aquatic species in estuaries and, in some instances, can also protect shorelines by reducing wave strength; therefore, declines in SAV due to nutrient enrichment are an important source of concern. Low water clarity is in part the result of accumulations of both algae and sediments in estuarine waters. In addition to contributing to declines in SAV, high levels of turbidity also degrade the aesthetic qualities of the estuarine environment.

An assessment of estuaries nationwide by the National Oceanic and Atmospheric Administration (NOAA) concluded that 64 estuaries (out of 99 with available data) suffered from moderate or high levels of eutrophication due to excessive inputs of both nitrogen (N) and phosphorus.[168] For estuaries in the Mid-Atlantic region, the contribution of atmospheric deposition to total N loads is estimated to range between 10 percent and 58 percent.[169] Estuaries in the eastern United States are an important source of food production, in particular for fish and shellfish production. The estuaries are capable of supporting large stocks of resident commercial species, and they serve as the breeding grounds and interim habitat for several migratory species. Eutrophication in estuaries may also affect the demand for seafood (after well-publicized toxic blooms), water-based recreation, and erosion protection provided by SAV.

4.2.3.1.2.2 *Terrestrial Enrichment*

Terrestrial enrichment occurs when terrestrial ecosystems receive N loadings in excess of natural background levels, through either atmospheric deposition or direct application. Atmospheric N deposition is associated with changes in the types and number of species and biodiversity in terrestrial systems. Nitrogen enrichment occurs over a long time period; as a result, it may take as many as 50 years or more to see changes in ecosystem conditions and indicators. One of the main provisioning services potentially affected by N deposition is grazing opportunities offered by grasslands for livestock production in the Central U.S. Although N deposition on these grasslands can offer supplementary nutritive value and promote overall grass production, there are concerns that fertilization may favor invasive grasses and shift the species composition away from native grasses. This process may ultimately reduce the productivity of grasslands for livestock production.

Terrestrial enrichment also affects habitats, for example the Coastal Sage Scrub (CSS) and Mixed Conifer Forest (MCF) habitats which are an integral part of the California landscape. Together the ranges of these habitats include the densely populated and valuable coastline and the mountain areas. Numerous threatened and endangered species at both the state and federal levels reside in CSS and MCF. Nutrient enrichment of the CSS and MCF also affects the regulating service of fire, by encouraging the growth of more flammable grasses and thus increasing fuel loads and altering the fire cycle.

4.2.3.1.3 *Vegetation Effects Associated with Gaseous Sulfur Dioxide, Nitric Oxide, Nitrogen Dioxide, Peroxyacetyl Nitrate, and Nitric Acid*

Uptake of gaseous pollutants in a plant canopy is a complex process involving adsorption to surfaces (leaves, stems, and soil) and absorption into leaves. These pollutants penetrate into leaves through the stomata, although there is evidence for limited pathways via the cuticle.[161] Pollutants must be transported from the bulk air to the leaf boundary layer in order to reach the stomata. When the stomata are closed, as occurs under dark or drought conditions, resistance to gas uptake is very high and the plant has a very low degree of susceptibility to injury. In contrast, mosses and lichens do not have a protective cuticle barrier to gaseous pollutants or stomates and are generally more sensitive to gaseous sulfur and nitrogen than vascular plants.[161]

Acute foliar injury from SO_2 usually happens within hours of exposure, involves a rapid absorption of a toxic dose, and involves collapse or necrosis of plant tissues. Another type of visible injury is termed chronic injury and is usually a result of variable SO_2 exposures over the growing season. Besides foliar injury, chronic exposure to low SO_2 concentrations can result in reduced photosynthesis, growth, and yield of plants.[161] These effects are cumulative over the season and are often not associated with visible foliar injury. As with foliar injury, these effects vary among species and growing environment. SO_2 is also considered the primary factor causing the death of lichens in many urban and industrial areas.[170]

Similarly, in sufficient concentrations, nitric oxide (NO), nitrogen dioxide (NO_2), peroxyacetyl nitrate (PAN), and nitric acid (HNO_3) can have phytotoxic effects on plants such as decreasing photosynthesis and inducing visible foliar injury. It is also known that these gases can alter the N cycle in some ecosystems, especially in the western U.S., and contribute to N saturation. Further, there are several lines of evidence that past and current HNO_3 concentrations may be contributing to the decline in lichen species in the Los Angeles basin.[171]

4.2.3.1.4 *Mercury Methylation*

Mercury is a persistent, bioaccumulative toxic metal that is emitted in three forms: gaseous elemental Hg (Hg^0), oxidized Hg compounds (Hg^{+2}), and particle-bound Hg (HgP). Methylmercury (MeHg) is formed by microbial action in the top layers of sediment and soils after Hg has precipitated from the air and deposited into waterbodies or land. Once formed, MeHg is taken up by aquatic organisms and bioaccumulates up the aquatic food web. Larger predatory fish may have MeHg concentrations many times higher, typically on the order of one million times, than the concentrations in the freshwater body in which they live. The NO_X SO_X ISA—Ecological Criteria concluded that evidence is sufficient to infer a causal relationship between sulfur deposition and increased mercury methylation in wetlands and aquatic environments.[161] Specifically, there appears to be a relationship between SO_4^{2-} deposition and mercury methylation; however, the rate of mercury methylation varies according to several spatial and biogeochemical factors whose influence has not been fully quantified. Therefore, the correlation between SO_4^{2-} deposition and MeHg cannot yet be quantified for the purpose of interpolating the association across waterbodies or regions. Nevertheless, because changes in MeHg in ecosystems represent changes in significant human and ecological health risks, the association between sulfur and mercury cannot be neglected.[161]

4.2.3.2 Deposition of Metallic and Organic Constituents of PM

Several significant ecological effects are associated with the deposition of chemical constituents of ambient PM such as metals and organics.[161] The trace metal constituents of PM include cadmium, copper, chromium, mercury, nickel, zinc, and lead. The organics include persistent organic pollutants (POPs), polyaromatic hydrocarbons (PAHs) and polybromiated diphenyl ethers (PBDEs). Direct effect exposures to PM occur via deposition (e.g., wet, dry or occult) to vegetation surfaces, while indirect effects occur via deposition to ecosystem soils or surface waters where the deposited constituents of PM then interact with biological organisms. While both fine and coarse-mode particles may affect plants and other organisms, more often the chemical constituents drive the ecosystem response to PM.[172] Ecological effects of PM include direct effects to metabolic processes of plant foliage; contribution to total metal loading resulting in alteration of soil biogeochemistry and microbiology, plant and animal growth and reproduction; and contribution to total organics loading resulting in bioaccumulation and biomagnification.

Particulate matter can adversely impact plants and ecosystem services provided by plants by deposition to vegetative surfaces.[161] Particulates deposited on the surfaces of leaves and needles can block light, altering the radiation received by the plant. PM deposition near sources of heavy deposition can obstruct stomata (limiting gas exchange), damage leaf cuticles and increase plant temperatures.[161] Plants growing on roadsides exhibit impact damage from near-road PM deposition, having higher levels of organics and heavy metals, and accumulating salt from road de-icing during winter months.[161] In addition, atmospheric PM can convert direct solar radiation to diffuse radiation, which is more uniformly distributed in a tree canopy, allowing radiation to reach lower leaves.[161] Decreases in crop yields (a provisioning service) due to reductions in solar radiation have been attributed to regional scale air pollution in counties with especially severe regional haze.[173]

In addition to damage to plant surfaces, deposited PM can be taken up by plants from soil or foliage. Copper, zinc, and nickel have been shown to be directly toxic to vegetation under field conditions.[161] The ability of vegetation to take up heavy metals is dependent upon the amount, solubility and chemical composition of the deposited PM. Uptake of PM by plants from soils and vegetative surfaces can disrupt photosynthesis, alter pigments and mineral content, reduce plant vigor, decrease frost hardiness and impair root development.

Particulate matter can also contain organic air toxic pollutants, including PAHs, which are a class of polycyclic organic matter (POM). PAHs can accumulate in sediments and bioaccumulate in freshwater, flora and fauna. The uptake of organic air toxic pollutants depends on the plant species, site of deposition, physical and chemical properties of the organic compound and prevailing environmental conditions.[161] Different species can have different uptake rates of PAHs. PAHs can accumulate to high enough concentrations in some coastal environments to pose an environmental health threat that includes cancer in fish populations, toxicity to organisms living in the sediment and risks to those (e.g., migratory birds) that consume these organisms.[174,175] Atmospheric deposition of particles is thought to be the major source of PAHs in the sediments of Lake Michigan, Chesapeake Bay, Tampa Bay and other coastal areas of the U.S.[176]

Contamination of plant leaves by heavy metals can lead to elevated concentrations in the soil. Trace metals absorbed into the plant, frequently by binding to the leaf tissue, and then are shed when the leaf drops. As the fallen leaves decompose, the heavy metals are transferred into the soil.[177,178] Many of the major indirect plant responses to PM deposition are chiefly soil-mediated and depend on the chemical composition of individual components of deposited PM. Upon entering the soil environment, PM pollutants can alter ecological processes of energy flow and nutrient cycling, inhibit nutrient uptake to plants, change microbial community structure, and affect biodiversity. Accumulation of heavy metals in soils depends on factors such as local soil characteristics, geologic origin of parent soils, and metal bioavailability. Heavy metals such as zinc, copper, and cadmium, and some pesticides can interfere with microorganisms that are responsible for decomposition of soil litter, an important regulating ecosystem service that serves as a source of soil nutrients.[161] Surface litter decomposition is reduced in soils having high metal concentrations. Soil communities have associated bacteria, fungi, and invertebrates that are essential to soil nutrient cycling processes. Changes to the relative species abundance and community composition are associated with deposited PM to soil biota.[161]

Atmospheric deposition can be the primary source of some organics and metals to watersheds. Deposition of PM to surfaces in urban settings increases the metal and organic component of storm water runoff.[161] This atmospherically-associated pollutant burden can then be toxic to aquatic biota. The contribution of atmospherically deposited PAHs to aquatic food webs was demonstrated in high elevation mountain lakes with no other anthropogenic contaminant sources.[161] Metals associated with PM deposition limit phytoplankton growth, affecting aquatic trophic structure. Long-range atmospheric transport of 47 pesticides and degradation products to the snowpack in seven national parks in the Western U.S. was recently quantified indicating PM-associated contaminant inputs in receiving waters during spring snowmelt. The recently completed Western Airborne Contaminants Assessment Project (WACAP) is the most comprehensive database on contaminant transport and PM depositional effects on sensitive ecosystems in the Western U.S.[179] In this project, the transport, fate, and ecological impacts of anthropogenic contaminants from atmospheric sources were assessed from 2002 to 2007 in seven ecosystem components (air, snow, water, sediment, lichen, conifer needles and fish) in eight core national parks. The study concluded that bioaccumulation of semi-volatile organic compounds occurred throughout park ecosystems, an elevational gradient in PM deposition exists with greater accumulation in higher altitude areas, and contaminants accumulate in proximity to individual agriculture and industry sources, which is counter to the original working hypothesis that most of the contaminants would originate from Eastern Europe and Asia.

4.2.3.3 Materials Damage and Soiling

Building materials including metals, stones, cements, and paints undergo natural weathering processes from exposure to environmental elements (e.g., wind, moisture, temperature fluctuations, sunlight, etc.). Pollution can worsen and accelerate these effects. Deposition of PM is associated with both physical damage (materials damage effects) and impaired aesthetic qualities (soiling effects). Wet and dry deposition of PM can physically affect materials, adding to the effects of natural weathering processes, by potentially promoting or accelerating the corrosion of metals, degrading paints and deteriorating building materials such as stone, concrete and marble.[180] The effects of PM are exacerbated by the presence of acidic gases and can be additive or synergistic depending on the complex mixture of pollutants in the air and surface characteristics of the material. Acidic deposition has been shown to have an effect on materials including zinc/galvanized steel and other metal, carbonate stone (such as monuments and building facings), and surface coatings (paints).[181] The effects on historic buildings and outdoor works of art are of particular concern because of the uniqueness and irreplaceability of many of these objects. In addition to aesthetic and functional effects on metals, stone and glass, altered energy efficiency of photovoltaic panels by PM deposition is also becoming an important consideration for impacts of air pollutants on materials.

4.2.4 Environmental Effects of Air Toxics

Emissions from producing, transporting and combusting fuel contribute to ambient levels of pollutants that contribute to adverse effects on vegetation. VOCs, some of which are considered air toxics, have long been suspected to play a role in vegetation damage.[182] In laboratory experiments, a wide range of tolerance to VOCs has been observed.[183] Decreases in harvested seed pod weight have been reported for the more sensitive plants, and some studies have reported effects on seed germination, flowering and fruit ripening. Effects of individual VOCs or their role in conjunction with other stressors (e.g., acidification, drought, temperature extremes) have not been well studied. In a recent study of a mixture of VOCs including ethanol and toluene on herbaceous plants, significant effects on seed production, leaf water content and photosynthetic efficiency were reported for some plant species.[184]

Research suggests an adverse impact of vehicle exhaust on plants, which has in some cases been attributed to aromatic compounds and in other cases to NO_X.[185,186,187] The impacts of VOCs on plant reproduction may have long-term implications for biodiversity and survival of native species near major roadways. Most of the studies of the impacts of VOCs on vegetation have focused on short-term exposure and few studies have focused on long-term effects of VOCs on vegetation and the potential for metabolites of these compounds to affect herbivores or insects.

4.3 Environmental Justice

EPA's 2016 "Technical Guidance for Assessing Environmental Justice in Regulatory Analysis" provides recommendations on conducting the highest quality analysis feasible, recognizing that data limitations, time and resource constraints, and analytic challenges will vary by media and regulatory context.[188] When assessing the potential for disproportionately high and adverse health or environmental impacts of regulatory actions on people of color, low-income populations, Tribes, and/or indigenous peoples, the EPA strives to answer three broad questions: (1) Is there evidence of potential environmental justice (EJ) concerns in the baseline (the state of the world absent the regulatory action)? Assessing the baseline will allow the EPA to determine whether pre-existing disparities are associated with the pollutant(s) under consideration (e.g., if the effects of the pollutant(s) are more concentrated in some population groups). (2) Is there evidence of potential EJ concerns for the regulatory option(s) under consideration? Specifically, how are the pollutant(s) and its effects distributed for the regulatory options under consideration? And, (3) do the regulatory option(s) under consideration exacerbate or mitigate EJ concerns relative to the baseline? It is not always possible to quantitatively assess these questions.

EPA's 2016 Technical Guidance does not prescribe or recommend a specific approach or methodology for conducting an environmental justice analysis, though a key consideration is consistency with the assumptions underlying other parts of the regulatory analysis when evaluating the baseline and regulatory options. Where applicable and practicable, the Agency endeavors to conduct such an analysis. EPA is committed to conducting environmental justice analysis for rulemakings based on a framework similar to what is outlined in EPA's Technical Guidance, in addition to investigating ways to further weave environmental justice into the fabric of the rulemaking process.

There is evidence that communities with EJ concerns are disproportionately impacted by the emissions sources controlled in this final rule.[189] Numerous studies have found that environmental hazards such as air pollution are more prevalent in areas where people of color and low-income populations represent a higher fraction of the population compared with the general population.[190,191,192] Consistent with this evidence, a recent study found that most anthropogenic sources of $PM_{2.5}$, including industrial sources and light- and heavy-duty vehicle sources, disproportionately affect people of color.[193] In addition, compared to non-Hispanic Whites, some other racial groups experience greater levels of health problems during some life stages. For example, in 2018-2020, about 12 percent of non-Hispanic Black; 9 percent of non-Hispanic American Indian/Alaska Native; and 7 percent of Hispanic children were estimated to currently have asthma, compared with 6 percent of non-Hispanic White children.[194] Nationally, on average, non-Hispanic Black and Non-Hispanic American Indian or Alaska Native people also have lower than average life expectancy based on 2019 data, the latest year for which CDC estimates are available.[195]

As discussed in Chapter 4.1.7 of this document, concentrations of many air pollutants are elevated near high-traffic roadways, and populations who live, work, or go to school near high-traffic roadways experience higher rates of numerous adverse health effects, compared to populations far away from major roads.

We conducted an analysis of the populations living in close proximity to truck freight routes as identified in USDOT's FAF4.[196] FAF4 is a model from the USDOT's Bureau of Transportation Statistics (BTS) and Federal Highway Administration (FHWA), which provides data associated with freight movement in the U.S.[T] Relative to the rest of the population, people living near FAF4 truck routes are more likely to be people of color and have lower incomes than the general population. People living near FAF4 truck routes are also more likely to live in metropolitan areas. Even controlling for region of the country, county characteristics, population density, and household structure, race, ethnicity, and income are significant determinants of whether someone lives near a FAF4 truck route. We note that we did not analyze the population living near warehousing, distribution centers, transshipment, or intermodal freight facilities.

We additionally analyzed national databases that allowed us to evaluate whether homes and schools were located near a major road and whether disparities in exposure may be occurring in these environments. Until 2009, the U.S. Census Bureau's American Housing Survey (AHS) included descriptive statistics of over 70,000 housing units across the nation and asked about transportation infrastructure near respondents' homes.[197,U] We also analyzed the U.S. Department of Education's Common Core of Data (CCD), which includes enrollment and location information for schools across the U.S.[198]

[T] FAF4 includes data from the 2012 Commodity Flow Survey (CFS), the Census Bureau on international trade, as well as data associated with construction, agriculture, utilities, warehouses, and other industries. FAF4 estimates the modal choices for moving goods by trucks, trains, boats, and other types of freight modes. It includes traffic assignments, including truck flows on a network of truck routes. *https://ops.fhwa.dot.gov/freight/freight_analysis/faf/*.

[U] The 2013 AHS again included the "etrans" question about highways, airports, and railroads within half a block of the housing unit but has not maintained the question since then.

In analyzing the 2009 AHS, we focused on whether a housing unit was located within 300 feet of a “4-or-more lane highway, railroad, or airport” (this distance was used in the AHS analysis).[V] We analyzed whether there were differences between households in such locations compared with those in locations farther from these transportation facilities.[199] We included other variables, such as land use category, region of country, and housing type. We found that homes with a non-White householder were 22-34 percent more likely to be located within 300 feet of these large transportation facilities than homes with White householders. Homes with a Hispanic householder were 17-33 percent more likely to be located within 300 feet of these large transportation facilities than homes with non-Hispanic householders. Households near large transportation facilities were, on average, lower in income and educational attainment and more likely to be a rental property and located in an urban area compared with households more distant from transportation facilities.

In examining schools near major roadways, we used the CCD from the U.S. Department of Education, which includes information on all public elementary and secondary schools and school districts nationwide.[200] To determine school proximities to major roadways, we used a geographic information system (GIS) to map each school and roadways based on the U.S. Census’s TIGER roadway file.[201] We estimated that about 10 million students attend schools within 200 meters of major roads, about 20 percent of the total number of public school students in the U.S.[W] About 800,000 students attend public schools within 200 meters of primary roads, or about 2 percent of the total. We found that students of color were overrepresented at schools within 200 meters of primary roadways, and schools within 200 meters of primary roadways had a disproportionate population of students eligible for free or reduced-price lunches.[X] Black students represent 22 percent of students at schools located within 200 meters of a primary road, compared to 17 percent of students in all U.S. schools. Hispanic students represent 30 percent of students at schools located within 200 meters of a primary road, compared to 22 percent of students in all U.S. schools.

We also reviewed existing scholarly literature examining the potential for disproportionate exposure among people of color and people with low socioeconomic status (SES). Numerous studies evaluating the demographics and socioeconomic status of populations or schools near roadways have found that they include a greater percentage of residents of color, as well as lower SES populations (as indicated by variables such as median household income). Locations in these studies include Los Angeles, CA; Seattle, WA; Wayne County, MI; Orange County, FL;

[V] This variable primarily represents roadway proximity. According to the Central Intelligence Agency’s World Factbook, in 2010, the United States had 6,506,204 km of roadways, 224,792 km of railways, and 15,079 airports. Highways thus represent the overwhelming majority of transportation facilities described by this factor in the AHS.
[W] Here, "major roads" refer to those TIGER classifies as either "Primary" or "Secondary." The Census Bureau describes primary roads as "generally divided limited-access highways within the Federal interstate system or under state management." Secondary roads are "main arteries, usually in the U.S. highway, state highway, or county highway system."
[X] For this analysis we analyzed a 200-meter distance based on the understanding that roadways generally influence air quality within a few hundred meters from the vicinity of heavily traveled roadways or along corridors with significant trucking traffic. See U.S. EPA, 2014. Near Roadway Air Pollution and Health: Frequently Asked Questions. EPA-420-F-14-044.

and the State of California, and nationally.[202,203,204,205,206,207,208] Such disparities may be due to multiple factors.[209,210,211,212,213]

People with low SES often live in neighborhoods with multiple stressors and health risk factors, including reduced health insurance coverage rates, higher smoking and drug use rates, limited access to fresh food, visible neighborhood violence, and elevated rates of obesity and some diseases such as asthma, diabetes, and ischemic heart disease. Although questions remain, several studies find stronger associations between air pollution and health in locations with such chronic neighborhood stress, suggesting that populations in these areas may be more susceptible to the effects of air pollution.[214,215,216,217]

Several publications report nationwide analyses that compare the demographic patterns of people who do or do not live near major roadways.[218,219,220,221,222,223] Three of these studies found that people living near major roadways are more likely to be people of color or low in SES.[224,225,226] They also found that the outcomes of their analyses varied between regions within the U.S. However, only one such study looked at whether such conclusions were confounded by living in a location with higher population density and how demographics differ between locations nationwide.[227] In general, it found that higher density areas have higher proportions of low-income residents and people of color. In other publications based on a city, county, or state, the results are similar.[228,229]

Two recent studies provide strong evidence that reducing emissions from heavy-duty vehicles is extremely likely to reduce the disparity in exposures to traffic-related air pollutants, both using NO_2 observations from the recently launched TROPospheric Ozone Monitoring Instrument (TROPOMI) satellite sensor as a measure of air quality, which provides the highest-resolution observations heretofore unavailable from any satellite.[230]

One study evaluated satellite-based NO_2 concentrations during the COVID-19 lockdowns in 2020 and compared them to NO_2 concentrations from the same dates in 2019.[231] That study found that average NO_2 concentrations were highest in areas with the lowest percentage of white populations, and that the areas with the greatest percentages of non-White or Hispanic populations experienced the greatest declines in NO_2 concentrations during the lockdown. These NO_2 reductions were associated with the density of highways in the local area.

In the second study, satellite-based NO_2 measured from 2018-2020 was averaged by racial groups and income levels in 52 large U.S. cities.[232] Using census tract-level NO_2, the study reported average population-weighted NO_2 levels to be 28% higher in low-income non-White people compared with high-income White people. The study also used weekday-weekend differences and bottom-up emission estimates to estimate that diesel traffic is the dominant source of NO_2 disparities in the studied cities. Overall, there is substantial evidence that people who live or attend school near major roadways are more likely to be of a non-White race, Hispanic, and/or have a low SES. Although proximity to an emissions source is an indicator of potential exposure, it is important to note that the impacts of emissions from tailpipe sources are not limited to communities in close proximity to these sources. For example, the effects of potential decreases in emissions from sources affected by this final rule might also be felt many miles away, including in communities with EJ concerns. The spatial extent of these impacts depends on a range of interacting and complex factors including the amount of pollutant emitted, atmospheric lifetime of the pollutant, terrain, atmospheric chemistry and meteorology. However, recent studies using satellite-based NO_2 measurements provide evidence that reducing emission from heavy-duty vehicles is likely to reduce disparities in exposure to traffic-related pollution.

In Chapter 6.4.9 of this RIA, we also present an analysis of how the air quality impacts from this rule are distributed among different populations, specifically focusing on $PM_{2.5}$ and ozone concentrations in the contiguous U.S. Using air quality modeling results from the proposal, we assessed whether areas with the worst projected baseline air quality in 2045 have larger numbers of people of color living in them, and if those with the worst projected air quality would benefit more from the final rule. We found that in the 2045 baseline, nearly double the number of people of color live within areas with the worst air quality, compared to non-Hispanic Whites (NH-Whites). We also found that the largest improvements in both ozone and $PM_{2.5}$ are estimated to occur in these areas with the worst baseline air quality. When we consider the national implications of this rule on specific race and ethnic groups, non-Hispanic Blacks will benefit the most from $PM_{2.5}$ and ozone air quality improvements in both absolute and relative (percent change from baseline) terms. Although the spatial resolution of the air quality modeling is not sufficient to capture very local heterogeneity of human exposures, particularly the pollution concentration gradients near roads, the analysis does allow estimates of demographic trends at a national scale. See Chapter 6.4.9 of the RIA for additional information on the demographic analysis.

In summary, there is substantial evidence that people who live or attend school near major roadways are more likely to be people of color, Hispanic ethnicity, and/or low socioeconomic status. This final rule will reduce emissions that contribute to NO_2 and other near-roadway pollution, improving air quality for the 72 million people who live near major truck routes and are already overburdened by air pollution from diesel emissions. Heavy-duty vehicles also contribute to regional concentrations of ozone and $PM_{2.5}$. The emission reductions from this final rule will result in widespread reductions in such pollution. The largest predicted improvements in both ozone and $PM_{2.5}$ are estimated to occur in areas with the worst baseline air quality, and a larger number of people of color are projected to reside in these areas.

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Chapter 5 Emissions Inventory

5.1 Introduction

This chapter presents our analysis of the national emissions impacts of the final standards for calendar years 2027 through 2045. In addition, this chapter describes the methods used to estimate the spatially and temporally-resolved emission inventories in 2016 and 2045 that supported the air quality modeling analysis documented in Chapter Chapter 6.

As described in detail in Chapter 5.2, the onroad national inventories were estimated using the public version of EPA's Motor Vehicle Emission Simulator (MOVES) model, MOVES3. The onroad national emission inventories were developed using a single national modeling domain, referred to as "national-scale" in MOVES. Inputs developed to model the national emission inventories for the final standards are discussed in Chapter 5.2.2. The national emissions inventory impacts for calendar years 2030, 2040, and 2045 for the final standards are presented in Chapter 5.3. In addition, the national emissions results for calendar years 2027 through 2045 are presented in Chapter 5.5.4.

As described in Chapter 5.4, MOVES was also used to estimate the emission inventories for air quality modeling. However, as described in Section VII of the preamble to this rule and also in Chapter Chapter 6, we did not perform new air quality modeling for the final rule; as described in Chapter 5.4.2., the emission reductions modeled in the air quality analysis compare well with those estimated for the final standards.

5.2 Model and Data Updates

To quantify the emissions impacts of the final standards, EPA used the public version of MOVES available at the time of the FRM analysis, MOVES3.[1] MOVES3 includes all the model updates previously made for "MOVES CTI NPRM" (the MOVES model used for the NPRM analysis), as well as other more recent updates. The additional updates included in MOVES3 are described in Table 5-1. Detailed descriptions of the underlying data and analyses that informed the model updates are documented in peer-reviewed technical reports referenced in Table 5-1. In addition, the peer review materials are available on the EPA's science inventory webpage.[2] Finally, the MOVES3 version used to quantify the emissions impacts of the final standards can be found in the docket.[3]

Table 5-1: Updates to MOVES3 from MOVES CTI NPRM

MOVES updates	Description
Heavy-duty running exhaust emission rates [4]	Further update heavy-duty diesel running exhaust emission rates for 2010 and later model year vehicles using the latest data from the Heavy-Duty In-Use Testing (HDIUT) Program. In this update, the MY2010+ vehicles are further divided into two groups – "model year 2010-2013 vehicles" and "model year 2014 and later vehicles" –to account for differences in emission performance of more recent engines and aftertreatment systems
Heavy-duty crank case emission rates **Error! Bookmark not defined.**	Update heavy-duty diesel crank case emission rates for model year 2007-2009 vehicles (based on certification test data) and for model year 2010 and later vehicles (using US EPA's National Vehicle and Fuel Emissions Laboratory (NFVEL) testing data)
Heavy-duty population and activity information [6]	Update information on heavy-duty vehicle populations based on vehicle registration data, FHWA vehicle miles traveled data, and updated projections from the Annual Energy Outlook (AEO) 2019 [5]
Glider trucks [6]	Update projected glider vehicle sales estimates for model year 2018 and later
Light-duty vehicle and other changes [7]	Updated gasoline fuel properties, light-duty vehicle emissions rates, light-duty activity, and other changes

5.2.1 Methodology Overview

We used MOVES3 to estimate the emissions impacts of the HD2027 final standards. First, we estimated emissions for a baseline scenario in which there are no new heavy-duty engine emission standards. We then estimated emissions for the control scenario, described in Chapter 1. The emissions impacts of the final standards were estimated by calculating the difference between the emissions estimated in the baseline and the control scenario. All of the model inputs, MOVES runspec files, and the scripts used for the analysis, as well as the version of MOVES used to generate the emissions inventories, can be found in the docket.[3]

In modeling the baseline scenario, we used the MOVES3 default heavy-duty emission rates updated based on the latest data, as described in Table 5-1. In particular, the updates to the heavy-duty exhaust and crankcase emission rates resulted in a lower national-scale baseline inventory for NO_X and $PM_{2.5}$, compared to the baseline national-scale emissions inventory estimated for the proposal. [A,]Note that because the national emissions inventories used a single national modeling domain, the baseline scenario did not account for different emission standards in California or other states that have adopted the California emission standards.[B]

[A] Note that as described in preamble Section III, we also made a programmatic adjustment to the proposed requirement to close crankcases; the combination of this programmatic change and the update to crankcase emission rates influence the national-scale $PM_{2.5}$ emissions inventory results for the final rule (see RIA Chapter 5.2.2.4 for additional detail).

[B] EPA is reviewing a waiver request under CAA section 209(b) from California for the Omnibus rule; until EPA grants the waiver, the HD Omnibus program is not enforceable. For more information on the California Air Resources Board Omnibus rule see, "Heavy-Duty Engine and Vehicle Omnibus Regulation and Associated Amendments," December 22, 2021. https://ww2.arb.ca.gov/rulemaking/2020/hdomnibuslownox. Last accessed September 21, 2022. *See also* "California State Motor Vehicle Pollution Control Standards and Nonroad Engine Pollution Control Standards; The ''Omnibus'' Low NOx Regulation; Request for Waivers of Preemption; Opportunity for Public Hearing and Public Comment" at 87 FR 35765 (June 13, 2022).

The vehicle activity (e.g., fleet age distributions, vehicle miles traveled by vehicle type and road type, vehicle speeds, off-network idling, hotelling hours, and start activity) and fuel inputs were kept the same for both baseline and control scenarios, using the default values in MOVES3.[6,7] For example, as shown in Table 5-1 above, future year projections of vehicle populations and vehicle miles travelled were updated[C] to reflect the estimates from the Department of Energy's Annual Energy Outlook 2019.[5] Note that the vehicle activity data for the emissions inventory used for the air quality analyses in the proposal included local activity data (as documented in Chapter 5.4) which captures more detailed information than the national default values.

We used the default data in MOVES for fuel usage and fuel property for the national inventory runs for calendar years from 2027 through 2045.[7] The national emissions inventory runs aggregate the gasoline fuel properties into a single region representing the United States, whereas the county-level MOVES runs for the air quality modeling analysis account for county and regional differences in gasoline fuel properties. Diesel and CNG fuel properties are also aggregated into one set of nationally-representative fuel properties for each calendar year and fuel type.

The emission rate inputs developed for modeling the final standards and the proposed Option 2 are discussed in detail in the following section.

5.2.2 MOVES Emission Rates for Control Scenarios

We developed separate MOVES emission rates for the final standards and the proposed Option 2 (collectively referred to as the control scenarios).[D] The methodologies used to develop the emission rates reflect the effects of this rulemaking's program elements (duty-cycle standards, off-cycle standards, closed crankcase requirements, refueling standards, regulatory useful life, and emissions warranty) on vehicles subject to the final rule as described below. We did not estimate the emission impacts of certain compliance provisions that target long-term compliance assurance. As we describe in Section IV of the preamble to this rule, we expect the improved serviceability and updated approach to inducements that we are finalizing will discourage owners from tampering their engines or emission control systems; however, we had insufficient data to estimate the impact of these program elements in the final rule.

MOVES has separate emission rates for heavy-duty vehicles according to three different fuel types—diesel, gasoline, and natural gas (NG) —and six different regulatory vehicle classes, shown in Table 5-2. The final rule includes new duty-cycle standards for all heavy-duty vehicles in the LHD45, MHD, HHD, and Urban Bus regulatory classes for all fuel types. The Urban Bus

[C] The version of MOVES used for the NPRM analysis, MOVES_CTI_NPRM, had projections based on Annual Energy Outlook 2018. Most of the changes in the baseline inventory come from the differences between the two AEO projections.

[D] The control scenario analyzed for the final standards differs slightly from the final program; we note these minor differences in relevant tables throughout this chapter.

regulatory class was modeled using the same zero-mile emission rates as HHD for the control scenarios.[E]

Light heavy-duty Class 2b and 3 trucks (LHD2b3) are composed of vehicles that are both chassis- and engine-certified. The final standards will apply to all engine-certified LHD2b3 vehicles, which are estimated to be a small fraction of the diesel LHD2b3 vehicles. All Class 2b and 3 gasoline-fueled vehicles are chassis-certified and will not be affected by the final rule.

A glider vehicle is a new motor vehicle produced with a used or remanufactured engine, and typically does not include the aftertreatment systems needed to meet the 2007 or 2010 heavy-duty emission standards.[F] In MOVES, glider vehicles are modeled as heavy heavy-duty diesel vehicles (Class 8 Trucks) that are assumed to emit at a level equivalent to the model year 2000 HHD vehicles.[6] Annual glider sales are fixed at 4,000 units per year for 2018 and later based on the 2018 and later sales volume cap for glider vehicles set forth in the Heavy-duty Greenhouse Gas Phase 2 Rulemaking[8], as well as the number of glider manufacturers and their historic production levels, as documented in the MOVES population and activity report.[6] For the final rule analysis, we assumed there is no change to the sales of glider vehicles and no change to glider vehicle emission rates from the baseline scenario.[G]

Table 5-2: MOVES Heavy-duty Regulatory Classes and Relevant MOVES Fuel Types

MOVES Regulatory Class Description	regClassName	regClassID	Gross Vehicle Weight Rating (GVWR) [lb.]	MOVES Fuel Types
Light Heavy-Duty Class 2b and 3 trucks	LHD2b3	41	8,501 – 14,000	Gasoline, Diesel
Light Heavy-Duty Class 4 and 5 Trucks	LHD45	42	14,001 – 19,500	Gasoline, Diesel
Medium Heavy-Duty (Class 6 and 7 Trucks)	MHD	46	19,501 – 33,000	Gasoline, Diesel
Heavy Heavy-Duty (Class 8 Trucks)	HHD	47	> 33,000	Diesel, NG
Urban Bus	Urban Bus	48	> 33,000	Diesel, NG
Gliders (Class 8 Trucks)	Glider Vehicles	49	> 33,000	Diesel

For heavy-duty diesel vehicles, we developed nitrogen oxide (NO_X) emission rates for running, start, and extended idle processes, as discussed in Chapters 5.2.2.1, 5.2.2.2, and 5.2.2.3, respectively, in response to the final and proposed Option 2 duty-cycle and off-cycle standards.

[E] Urban Bus vehicles are also considered HHD vehicles. They are modeled as a separate regulatory class in MOVES because they had stricter PM emission standards for 1994 through 2006 model years. For the control scenarios, the Urban Bus emission rates were assumed to be equivalent to HHD except for the age effects discussed in Section 5.2.2.1.2.

[F] See the definition of "glider vehicle" in 40 CFR 1037.801 and our discussion in the preamble of the heavy-duty GHG Phase 2 rulemaking (81 FR 73941, October 25, 2016).

[G] It may be possible that future sales of glider vehicles are lower than assumed in MOVES3 or glider vehicles emit at a lower level due to the final standards, resulting in lower emissions from gliders. However, due to the uncertainty associated with these assumptions, the emissions from gliders were kept the same between the baseline and control scenarios in the current analysis.

The lower NO_X emissions are anticipated to be achieved using the technologies discussed in Chapter 3, including cylinder deactivation (CDA) and dual selective catalytic reduction exhaust aftertreatment system. We also revised heavy-duty diesel emission running rates for NO_X, total hydrocarbons (THC), carbon monoxide (CO), and particulate matter below 2.5 microns ($PM_{2.5}$), due to the changes to the regulatory useful life and warranty, by modifying MOVES' tampering and mal-maintenance (T&M) calculations, as discussed in Chapter 5.2.2.1.2. We also evaluated the impacts on the $PM_{2.5}$ crankcase emissions from heavy-duty diesel vehicles due to the closed crankcase design option for heavy-duty diesel engines[H] in the final and proposed Option 2 standards as discussed in Section 5.2.2.4.

For heavy-duty gasoline vehicles, we developed revised NO_X, THC, CO, and $PM_{2.5}$ emission rates for running exhaust in response to the final and proposed Option 2 duty-cycle standards, as discussed in Chapter 5.2.2.6. We also revised THC refueling emission rates in response to the final and proposed Option 2 refueling standard discussed in Chapter 5.2.2.7. We did not revise the start emission rates for heavy-duty gasoline vehicles to account for the final and proposed Option 2 standards due to lack of sufficient data to model the impact.

For heavy-duty NG vehicles, we did not estimate reductions in NO_X or other pollutants due to the final and proposed Option 2 heavy-duty spark-ignition duty-cycle standards (discussed in Section III of the Preamble). As shown in the MOVES heavy-duty exhaust report[I], the average FTP emissions for MY 2010-2017 CNG engine families is close to 0.1 g/hp-hr. We expect small reductions in NO_X emissions from heavy-duty NG engines with the final and proposed Option 2 scenarios. However, in part due to the small contribution of NG emissions to total emissions, we did not estimate the reductions in NG engines in this analysis.

Similarly, we did not estimate emission reductions from the final and proposed Option 2 useful life and warranty requirements for gasoline and NG vehicles. This is because, in MOVES, we do not tie emission estimates to heavy-duty gasoline or NG vehicle warranty and useful life periods, but rather estimate age effects from emissions data, or adapt light-duty gasoline or heavy-duty diesel age effects. Gasoline and NG-fueled vehicles' contributions to the heavy-duty vehicle NO_X emissions inventory are small (see Figure 5-15), and thus, we expect minimal impacts on our emission inventory estimates from our conservative approach of estimating reductions solely from running emissions for these vehicles.[J]

While we estimated different emission rates for the control scenarios, we did not estimate differences in the speciation of the total organic gases and particulate matter emissions between the baseline and control scenarios. Emissions speciation refers to the calculation of individual compounds or classes of compounds within a broader pollutant. For example, MOVES conducts speciation to estimate benzene from volatile organic compound (VOC) emissions and particle-

[H] As described in Preamble Section III.B, EPA is finalizing a requirement for manufacturers to use one of two options for controlling crankcase emissions, either: 1) as proposed, closing the crankcase, or 2) an updated version of the current requirements for an open crankcase that includes additional requirements for measuring and accounting for crankcase emissions.

[I] See Table 4-2 in Exhaust Emission Rates for Heavy-Duty Onroad Vehicles in MOVES3

[J] According to MOVES, the contributions from NG vehicles to HD NO_X inventory in calendar year 2045 are: 0.6% (baseline), 1.1% (final standards), and 1.0% (proposed Option 2). These contributions are dependent on the current projections of NG in the future heavy-duty fleet and the emission rates of NG vehicles, many of which are already meeting the 0.02 g/hp-hr NO_X standard.

phase naphthalene from particulate matter (PM) emissions. Speciation is important to estimate individual toxics and necessary for air quality modeling. However, we do not have sufficient data to support differences in the emissions speciation between the control case and baseline scenarios. Heavy-duty diesel vehicles in the control case scenarios apply the same speciation values as the baseline scenario for heavy-duty diesel vehicles model year 2010 and later. Similarly, emissions from heavy-duty gasoline vehicles in the control scenarios use the same speciation as heavy-duty gasoline vehicles in the baseline scenario. Changes in emissions for individual compounds presented in Section 5.3 (acetaldehyde, benzene, formaldehyde, and naphthalene) are due to changes in the hydrocarbon and particulate matter emission rates for the control scenarios presented in this section.

5.2.2.1 Heavy-Duty Diesel Running Emission Rates

This section documents the approach used to develop the heavy-duty diesel emission running rates for the control scenarios. We first estimated new zero-mile NO_X emission rates in response to the final and proposed Option 2 duty-cycle and off-cycle standards as discussed in Chapter 5.2.2.1.1. The zero-mile emission rate in MOVES is defined as the emission rate of a new vehicle without any tampering and mal-maintenance effects. We then estimated the effects of lengthened regulatory useful life and warranty periods and applied them to the aged NO_X, THC, CO and $PM_{2.5}$ emission rates for the final and proposed Option 2 standards in Chapter 5.2.2.1.2

Emission rates in MOVES are defined by the operating mode, regulatory class, model year, and fuel type. The running operating modes are defined in Table 5-3.

Table 5-3: MOVES Running Operating Mode Definitions

OpModeID	Operating Mode	Vehicle Speed (v, mph)	Scaled Tractive Power (STP, skW)
0	Deceleration/Braking	All speeds	
1	Idle	v < 1.0	
11	Coast	1 ≤ v < 25	STP< 0
12	Cruise/Acceleration		0 ≤ STP< 3
13	Cruise/Acceleration		3 ≤ STP< 6
14	Cruise/Acceleration		6 ≤ STP< 9
15	Cruise/Acceleration		9 ≤ STP< 12
16	Cruise/Acceleration		12 ≤ STP
21	Coast	25 ≤ v < 50	STP< 0
22	Cruise/Acceleration		0 ≤ STP< 3
23	Cruise/Acceleration		3 ≤ STP< 6
24	Cruise/Acceleration		6 ≤ STP< 9
25	Cruise/Acceleration		9 ≤ STP< 12
27	Cruise/Acceleration		12 ≤ STP< 18
28	Cruise/Acceleration		18 ≤ STP< 24
29	Cruise/Acceleration		24 ≤ STP< 30
30	Cruise/Acceleration		30 ≤ STP
33	Cruise/Acceleration	50 ≤ v	STP< 6
35	Cruise/Acceleration		6 ≤ STP< 12
37	Cruise/Acceleration		12 ≤ STP<18
38	Cruise/Acceleration		18 ≤ STP< 24
39	Cruise/Acceleration		24 ≤ STP< 30
40	Cruise/Acceleration		30 ≤ STP

Note: The braking mode is defined by deceleration events. The other running operating modes are defined by vehicle speed (v) and scaled tractive power (STP), which is the vehicle power scaled by a constant f_{scale} factor (in units of metric tons). Further details on the heavy-duty operating modes and definitions are included in the MOVES3 technical report.**Error! Bookmark not defined.**

5.2.2.1.1 *Emission Rates Based on Duty-Cycle and Off-Cycle Standards*

We developed zero-mile heavy-duty diesel NO_X emission rates that reflect the final and proposed Option 2 duty-cycle standards and the off-cycle standards. To do so, we first estimated the effects of the duty-cycle standards and the off-cycle standards separately, as discussed in Chapters 5.2.2.1.1.1 and 5.2.2.1.1.2, respectively. Then, we estimated the combined effect of both the duty-cycle standards and off-cycle standards on the zero-mile emission rates used in MOVES (see Chapter 5.2.2.1.1.3).

5.2.2.1.1.1 *Emission Rates Based on Duty-Cycle Standards*

In this section, we document the methods used to develop revised NO_X running emission rates for heavy-duty diesel vehicles based on the final and proposed Option 2 duty-cycle standards. The baseline, final and proposed Option 2 NO_X heavy-duty compression ignition duty-cycle exhaust emission standards are shown in Table 5-4. The duty-cycle standards include three separate tests: low load cycle (LLC), Federal Test Procedure (FTP), and Supplemental Emission Test - Ramped Modal Cycle (SET-RMC). Both the final and proposed Option 2 standards apply starting in MY 2027.

Table 5-4: Heavy-duty Compression Ignition Duty-Cycle NO_X Standards for the Final and Proposed Option 2 Scenarios

Scenario	Applicable Model Years	Regulatory Classes	LLC (g/hp-hr)	FTP (g/hp-hr)	SET-RMC (g/hp-hr)
Baseline	Model Year 2010+	LHD, MHD, HHD	-	0.2	0.2
Final Standards	Model Year 2027+	LHD, MHD, HHD	0.05 [0.065][A]	0.035 [0.05][A]	0.035 [0.05][A]
Proposed Option 2	Model Year 2027+	LHD, MHD, HHD	0.1	0.05	0.05

[A] Values in brackets denote standards that were only applied to HHD engines in the modeling of the final control scenario; in the final program, these values will apply to both MHD and HHD, see preamble Section III.B for details.

The final and proposed Option 2 scenarios also include revised standards for PM emissions as discussed in Section III of the preamble. We did not model the revised PM standards for heavy-duty diesel vehicles in MOVES, because the revised PM standards are intended to prevent backsliding of PM reductions already achieved with current heavy-duty diesel emission control systems. We estimate reductions in heavy-duty diesel PM emissions will occur due to the lengthened warranty and useful life periods, discussed in Chapter 5.2.2.1.2, and due to the crankcase emissions control, as discussed in Chapter 5.2.2.4.

We used the final and proposed Option 2 standards for the FTP and (SET-RMC) to estimate the effect of the duty-cycle standards on MOVES NO_X emission rates. Because we do not have sufficient (LLC) test data on existing heavy-duty diesel vehicles to develop the modeling inputs specific for the LLC standard in MOVES, we used the FTP standard to model the impact of the standards on low-power operation.

Equation 5-1 through Equation 5-6 were used to incorporate the effects of more stringent FTP and SET-RMC engine duty-cycle emission standards on MOVES running exhaust NO_X emission rates for model years subject to the final and proposed Option 2 standards. The term R_{duty} is the ratio between the final or proposed Option 2 emission standards and the current FTP and SET-RMC duty-cycle standards (0.2 g/hp-hr).

Equation 5-1

$$\mathrm{R_{duty}} = \frac{\text{Final or proposed Option 2 FTP or SET RMC standard}}{\text{Current standard}}$$

R_{duty} ranges between 17 and 25 percent for the control scenarios considered, as shown in Table 5-5.

Table 5-5: R_{duty} Ratios Calculated for Each Scenario

Scenario	Applicable Model Years	Regulatory Classes	FTP and SET standard (g/hp-hr)	R_{duty}
Final Standards	2027+	LHD, MHD	0.035	17.5%
		HHD	0.05[A]	25%
Proposed Option 2	2027+	LHD, MHD, HHD	0.05	25%

[A] Values in this row denote standards that were only applied to HHD engines in the modeling scenario for the final rule analysis; in the final program, these values will apply to both MHD and HHD, see preamble Section III.B for details

To estimate the effect of final and proposed Option 2 engine dynamometer duty-cycle standards on in-use emissions, we used the relationship between reductions in the most recent duty-cycle standard compared to reductions in in-use emissions. The 2010 0.2 g/hp-hr NO_X emission standard[9] is the most recent heavy-duty NO_X emission standard. To evaluate the in-use effectiveness of the 2010 standard, we compared the in-use NO_X emission rates from vehicles that were certified to the previous heavy-duty NO_X standard and the 2010 standard. Equation 5-2 defines R_{in_use} as the ratio between the percent change observed in-use from vehicles compliant with the 2010 NO_X standard relative to vehicles compliant with the previous standard, and the percent change in the 2010 standard FTP standards relative to the previous standard.[K] In other words, R_{in_use} is the ratio between the relative effectiveness of reducing in-use emissions compared to the relative reduction in the FTP duty-cycle emission standard.

Equation 5-2

$$\mathrm{R_{in\_use}} = \frac{\text{\% Change in the inuse emission rates from 2010 compliant vehicles}}{\text{\% Change in the 2010 FTP standard}}$$

The percent change in in-use emission rates from vehicles certified to the 2010 standard (the numerator in Equation 5-2) was estimated using Equation 5-3. The MOVES emission rates for HHD vehicles certified to the 2010 0.2 g/hp-hr standard (the numerator) are calculated from 93 MY 2010-2015 HHD vehicles with engine family emission limit (FEL) certified below the 0.2 g/hp-hr NO_X emissions level and tested as part of the Heavy-Duty In-Use Testing program.**Error! Bookmark not defined.** The MOVES emission rates for HHD vehicles certified to the 2004-2006 standard (the denominator) are based on 91 MY 2003-2006 trucks from two in-use datasets: ROVER data collected by the US EPA and the Heavy-Duty Diesel Consent Decree data collected by West Virginia University. The in-use datasets and analysis used to derive the emission rates in MOVES3 are documented in the MOVES3 heavy-duty exhaust technical report.**Error! Bookmark not defined.**

[K] In 2004-2006, the NMHC+NO_X emission standard was 2.4 g/hp-hr; the 0.2 g/hp-hr NO_X standard began to be phased-in starting in 2007, with a full-phase in 2010.

Equation 5-3

$$\%\ \text{Change in the in\_use emission rates from 2010 compliant vehicles} = \frac{\text{Emission rate from HHD 2010 compliant vehicles}}{\text{HHD MY 2006 MOVES emission rate}} - 1$$

The percent change from Equation 5-3 was applied to each MOVES operating mode to evaluate the effectiveness of the 2010 standard across different ranges of in-use operating conditions, as shown in Figure 5-1 and Table 5-6. The emission reduction is larger for the higher speed and higher load MOVES operating modes, with the largest decrease observed for speeds above 50 mph (operating modes 33 through 35). The lowest effectiveness of the standards is observed for low speed and several low power operating modes (operating mode 1, 11, and 21), with an exception of the deceleration bin (operating mode 0).

Equation 5-4 was used to estimate the percent reduction between the 2010 standard and the 2004-2006 emission standard. This term is also the denominator of Equation 5-2. The percent reduction in the NO_X emission standard was estimated assuming that NO_X emissions consist of 70 percent of the combined NMHC (Non-Methane Hydrocarbons) + NO_X 2004-2006 emission standard (2.4 g/hp-hr), consistent with the assumption used in MOVES3.**Error! Bookmark not defined.** This value is also plotted as a line in Figure 5-1 to compare to the in-use emission rate reductions.

Equation 5-4

$$\%\ \text{Change in the 2010 FTP standard} = \frac{0.2}{(70\% \times 2.4)} - 1 = -88.1\%$$

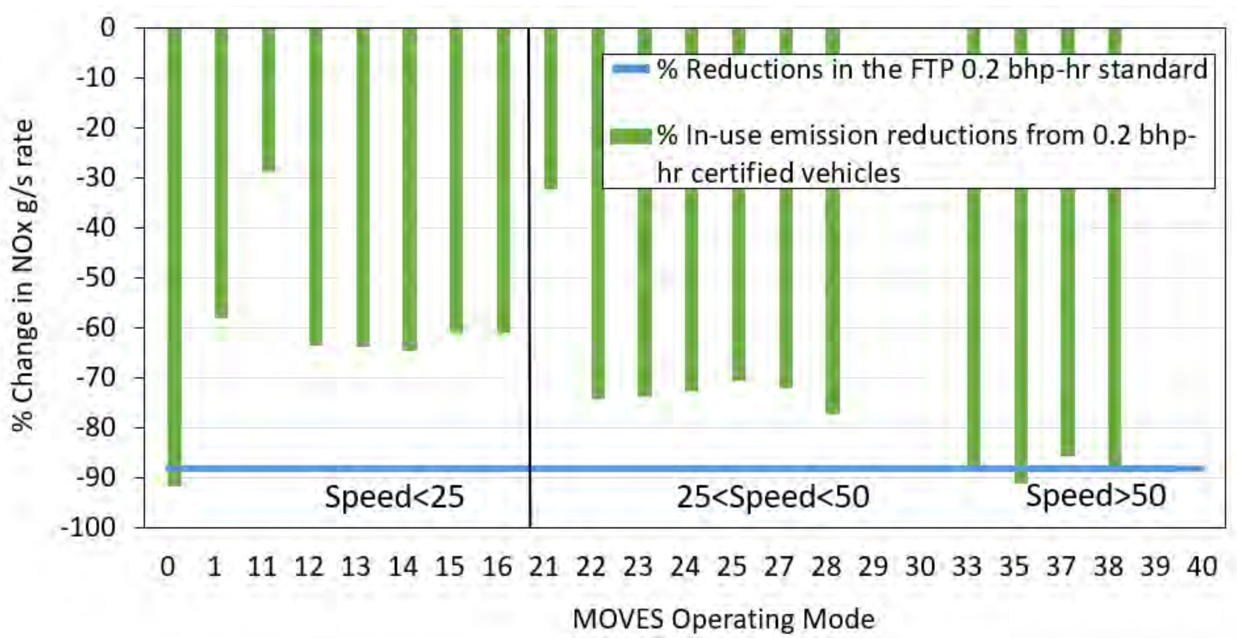


Figure 5-1: Percent change in in-use emission rates for 2010 standard (0.2 g/hp-hr) compliant HHD vehicles, compared to the percent change in the 2010 duty-cycle standard across MOVES operating modes

Table 5-6 displays the R_{in_use} as calculated by Equation 5-2. For operating modes with R_{in_use} values greater than one, the observed in-use emission reductions are greater than would be expected due to the change in FTP emission standard. R_{in_use} values less than one suggest that in-use emissions are less impacted than the change in the FTP emissions standards.

Table 5-6: Calculation of R_{in_use} by MOVES Operation Mode

MOVES OpMode	HHD MOVES MY 2006 NO_X emission rates (g/hr)	NO_X Emission rate from 2010 compliant HHD vehicles (g/hr)[a]	Percent change in in-use NO_X emission rates from 2010 compliant vehicles (%)	R_{in_use}
0	0.038	0.0031	-91.8	1.04
1	0.015	0.0063	-57.9	0.66
11	0.015	0.0106	-28.8	0.33
12	0.058	0.0210	-63.5	0.72
13	0.093	0.0339	-63.7	0.72
14	0.127	0.0453	-64.4	0.73
15	0.145	0.0565	-60.9	0.69
16	0.188	0.0734	-60.9	0.69
21	0.010	0.0066	-32.1	0.36
22	0.064	0.0163	-74.4	0.84
23	0.093	0.0243	-73.9	0.84
24	0.132	0.0359	-72.7	0.83
25	0.165	0.0485	-70.6	0.80
27	0.225	0.0633	-71.8	0.82
28	0.244	0.0558	-77.2	0.88
29	0.314			0.88[b]
30	0.384			0.88
33	0.051	0.0062	-87.8	1.00
35	0.148	0.0130	-91.2	1.04
37	0.226	0.0323	-85.7	0.97
38	0.268	0.0306	-88.6	1.01
39	0.345			1.01
40	0.422			1.01

Notes:

[a] The HHD rates in this table are based on f_{scale} of 17.1 metric tons to be consistent with the f_{scale} of the MY 2006 HHD emission rates in MOVES3. Note that the f_{scale} for model year 2010 and later in MOVES3 is 10 for HHD, 7 for MHD, and 5 for LHD45 and LHD2b3.**Error! Bookmark not defined.**

[b] For operating modes lacking data, we used the same R_{in_use} for the closest operating mode.

Equation 5-5 is used to estimate the percentage reduction to NO_X running emissions from the change in the duty-cycle standard for each operating mode.[L]

[L] We assumed that the R_{in_use} values calculated by MOVES operating mode can be applied to the MOVES rates that are derived using a different f_{scale}. The change in f_{scale} does not change the definition of operating modes that are not defined by Scaled Tractive Power, STP (deceleration operating mode 0, and idle operating mode 1), or operating modes with negative STP values (operating mode 11 and 21), defined in Table 5-3. Changing the f_{scale} values does change the definition of vehicle operation in the other operating modes. However, the R_{in_use} values are relatively constant for the positive power operating modes within each speed range as observed in Table 5-6. In Figure 5-1, we deemed it was not necessary to attempt to account for the f_{scale} differences when applying the R_{in_use} values.

Equation 5-5

$$\mathrm{R_{duty\_in\_use}} = \left(1 - \mathrm{R_{duty}}\right) \times \mathrm{R_{in\_use}}$$

Where:
$R_{duty\_in\_use}$= the percent emission reductions in the in-use running NO_X emissions estimated from changing the FTP duty-cycle standard.

Equation 5-6 is used to estimate the heavy heavy-duty diesel NO_X running emission rates from the changes in the duty-cycle standard. The same calculations were applied to estimate the other heavy-duty diesel regulatory classes.[M]

Equation 5-6

$$\mathrm{ER_{duty\_in\_use}} = \left(1 - \mathrm{R_{duty\_in\_use}}\right) \times \mathrm{ER_{MOVES\_baseline}}$$

Where:
$ER_{duty\_in\_use}$= the MOVES running NO_X emission rates for the control scenarios based on reduction in the duty-cycle standard
$R_{duty\_in\_use}$= the percent emission reductions in running NO_X emissions estimated from changing to FTP duty-cycle standard
$ER_{MOVES\_baseline}$= the MOVES baseline running NO_X emission rates for each regulatory class

The estimated HHD MOVES running NO_X emission rates for the control scenarios, estimated based on the duty-cycle standards in Table 5-4, are shown in Figure 5-2.

[M] We applied the $R_{duty\_in\_use}$ developed on HHD data to both the LHD45 and MHD regulatory classes, which have different f_{scale} values in MOVE3 for the 2010 and later model years. We believe this approximation is defensible for the same reason provided in footnote L.

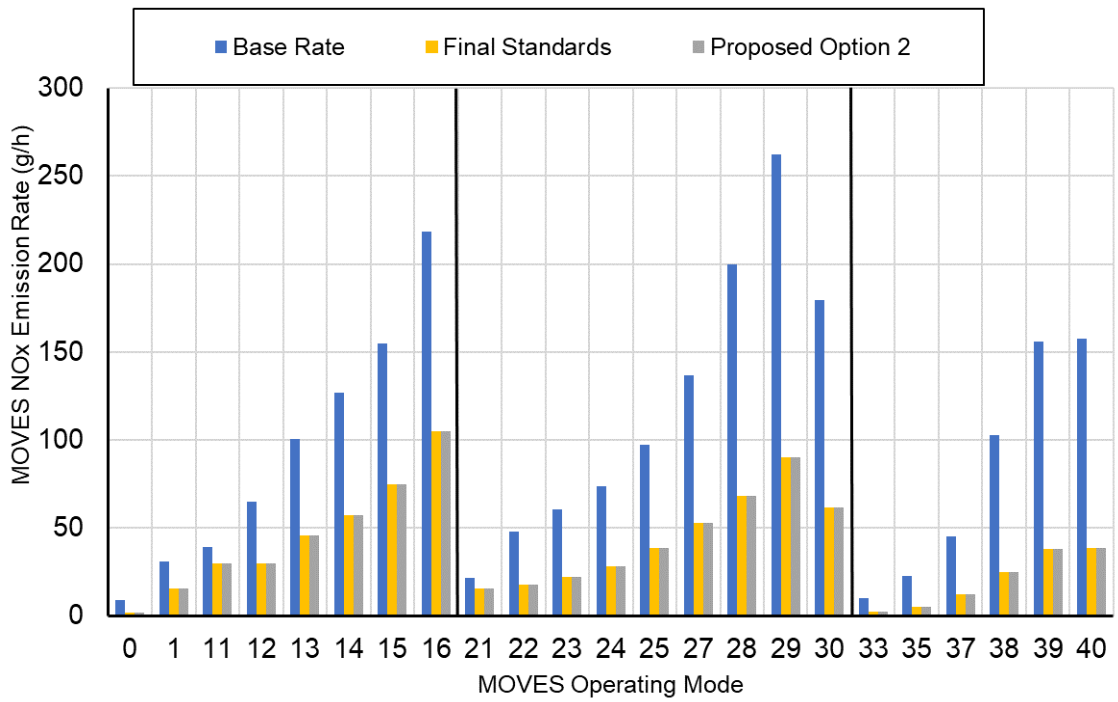


Figure 5-2: Duty-cycle-based running NO_X emissions, $ER_{duty\_in\_use}$, for HHD diesel for the control scenarios

5.2.2.1.1.2 *Emission Rates Based on Off-cycle Standards*

In this section, we document the methods used to estimate MOVES NO_X running emission rates based on the final and proposed Option 2 off-cycle standards for heavy-duty diesel vehicles. Table 5-7 presents the calculated off-cycle standards used to develop MOVES inputs for the control scenarios. The final and proposed Option 2 off-cycle standards all include requirements for operating conditions in three bins: idle, low-load and medium-to-high load.[N] The off-cycle operation in the control scenarios is defined as follows: idle is less than 6 percent of maximum power; low-load is 6 to 20 percent of maximum power; and medium-to-high load is above 20 percent of maximum power.[O] In developing inputs to MOVES, we did not apply a scaling factor to the off-cycle idling operation and assumed manufacturers will comply with the voluntary idle standard in all off-cycle idle operation. We then developed the off-cycle standards for the control scenarios using the procedures as described below.

[N] At the time of analyzing the final standards, we used the 3-bin approach as proposed in the NPRM (while setting the same numeric standards for the low-load and the medium-high load bins). In the final rule, the low-load and the medium-high load bins are consolidated into a single "non-idle operation" bin, see preamble Section III.C.
[O] See preamble Section III.C for more discussion on defining off-cycle operations in the final program.

Table 5-7. Calculated Off-Cycle NO_X Standards used for the Control Scenarios

Scenario	Model Year	Regulatory Class	Engine Cycle	Reference Off-cycle NO_X Standard	Off-cycle Bin	Off-Cycle NO_X Standards (g/hr for idling, g/hp-hr for low-load and medium to high-load)
Final Standards	2027+	LHD, MHD	Idle (g/hr)[A]	5	Idle, < 6% power	5
			LLC (g/hp-hr)	0.05	Low-load, 6-20% power	0.058
			FTP & SET (g/hp-hr)	0.035	Medium to High Load, >20% power	0.058
		HHD [B]	Idle (g/hr)[A]	5	Idle, < 6% power	5
			LLC (g/hp-hr)	0.075	Low-load, 6-20% power	0.088
			FTP & SET (g/hp-hr)	0.05	Medium to High Load, >20% power	0.088
Proposed Option 2	2027+	LHD, MHD, HHD	Idle (g/hr)[A]	10	Idle, < 6% power	10
			LLC (g/hp-hr)	0.1	Low-load, 6-20% power	0.15
			FTP & SET (g/hp-hr)	0.05	Medium to High Load, >20% power	0.075

[A] Note that the voluntary idle standard in the final control scenario that we modeled is different than the voluntary idle standard in the final program, see preamble Sections III.B and III.C for details on the voluntary idle standard in the final program and the off-cycle standard for idle emissions, respectively.

[B] Note that, in both control case scenarios, we modeled all engine categories complying with off-cycle standards during in-use operations. For the final control scenario, the off-cycle standards for HHD include an in-use compliance margin of 30 mg/hp-hr. As discussed in preamble Section III.C, the compliance margin for MHDE and HHDE in the final rule is 15 mg/hp-hr for both off-cycle and duty-cycle emissions.

We calculated the voluntary idle NO_X g/hr standard in units of NO_X g/CO_2 kg using Equation 5-7, and the resulting values are displayed in Table 5-8.

Equation 5-7

$$\text{Voluntary Idle } \frac{NO_X}{CO_2} \text{ standard } \left(\frac{g}{kg}\right) = \left[\text{Voluntary Idle } NO_X \text{ standard } \left(\frac{g}{hr}\right)\right] \Big/ \left[\text{Idle } CO_2 \left(\frac{kg}{hr}\right)\right]$$

Where $Idle\ CO_2 \left(\frac{kg}{hr}\right)$= the MOVES average CO_2 (kg/hr) emission rate for HHD diesel vehicles for MOVES idle (operating mode 1 defined in Table 5-3).

Table 5-8: Calculation of Voluntary Idle NO_X/CO_2 Standard (g/kg)

Scenario	Applicable Model Years	Voluntary Idle NO_X standard (g/hr)	Average HHD Idle (Operating Mode=1) CO_2 emission rate (kg/hr)	Voluntary Idle NO_X/CO_2 standard (g/kg)
Final Standards	2027+	5	7.68	0.65
Proposed Option 2	2027+	10	7.68	1.30

[A] Note that the voluntary idle standard in the final control scenario that we modeled is different than the voluntary idle standard in the final program, see preamble Sections III.B and III.C for details on the voluntary idle standard in the final program and the off-cycle standard for idle emissions, respectively.

Next, we converted the reference off-cycle NO_X standards into units of gram per hour (g/hr) for each MOVES operating mode. We refer to g/hr rates as the off-cycle standard compliant emission rates, which are shown in Table 5-9 in Columns (F) for HHD vehicles for the final standards.

In Table 5-9, Column (B) contains the MOVES CO_2 emission rate for Model Year 2027 HHD diesel vehicles for each MOVES operating mode. Column (C) includes the mean power for each operating mode bin calculated from the Heavy-Duty In-Use Testing data, which is the same data set that was used to derive the MOVES CO_2 emission rates.[10] The percent load, Column (D), is calculated for each operating mode bin using Equation 5-8

Equation 5-8

$$\text{Percent Load}_{\text{OpMode=i}} = \frac{\text{Mean Power}_{\text{OpMode= i}}}{\text{Mean Power}_{\text{OpMode=40}}}$$

Where i= one of the 23 MOVES running exhaust operating modes from 0 to 40
$Mean\ Power_{OpMode=\ i}$= the mean power for each of the MOVES operating modes shown in Column C.
$Mean\ Power_{OpMode=40}$= assumed maximum power bin = 470 hp for HHD diesel vehicles.

We then assigned each MOVES operating mode into a power classification (Column (E)) based on the percent load (Column (D)), where percent load less than 6 percent of maximum power is idle, 6 to 20 percent of maximum power is low-load, and above 20 percent of maximum power is medium-to-high load.

Table 5-9: Calculation of the Off-cycle NO_X Standard Compliant Emission Rate for HHD Diesel Vehicles for the Final Control Scenario

A	B	C	D	E	F
MOVES operating mode	MOVES MY 2027 HHD CO_2 emission rate (kg/hr)	Mean power (hp)	Percent load	Power classification	MY 2027+ off-cycle compliant emission rate (g/hr)
0	4.92	6.04	1.3%	Idle	3.20
1	7.68	8.10	1.7%	Idle	5.00
11	13.42	1.04	0.2%	Idle	8.73
12	21.69	28.90	6.2%	Low Load	2.54
13	37.31	75.16	16.1%	Low Load	6.61
14	52.20	121.18	26.0%	Medium to High Load	10.66
15	68.68	166.98	35.8%	Medium to High Load	14.69
16	110.42	282.24	60.5%	Medium to High Load	24.84
21	13.92	-1.61	-0.3%	Idle	9.06
22	32.99	34.43	7.4%	Low Load	3.03
23	44.71	77.71	16.7%	Low Load	6.84
24	59.82	121.62	26.1%	Medium to High Load	10.70
25	77.03	167.82	36.0%	Medium to High Load	14.77
27	102.53	230.56	49.4%	Medium to High Load	20.29
28	142.09	327.41	70.2%	Medium to High Load	28.81
29	181.90	403.76	86.5%	Medium to High Load	35.53
30	212.63	470.01	100.7%	Medium to High Load	41.36
33	28.36	34.76	7.4%	Low Load	3.06
35	71.87	145.03	31.1%	Medium to High Load	12.76
37	106.93	227.23	48.7%	Medium to High Load	20.00
38	148.35	323.23	69.3%	Medium to High Load	28.44
39	183.17	396.00	84.9%	Medium to High Load	34.85
40	196.42	466.62	100.0%	Medium to High Load	41.06

The off-cycle NO_X standard compliant emission rate in Column (F) is then calculated based on the power classification and the stringency of the off-cycle standard. For operating modes classified as idle, we multiplied the MOVES CO_2 emission rate in Column (B) with the NO_X/CO_2 off-cycle idle standard calculated in Table 5-8 for the corresponding control scenario using Equation 5-9.

Equation 5-9

$$\text{Idle Emission Rate} = \text{MOVES MY 2027 HHD CO}_2\text{ Emission Rate}\left(\frac{\text{g}}{\text{hr}}\right) \times \text{Voluntary Idle}\ \frac{NO_X}{\text{CO}_2}\ \text{in\_use standard}\left(\frac{\text{g}}{\text{kg}}\right)$$

For the operating modes classified as low-load and medium- to high-load, we multiplied the off-cycle (g/hp-hr) standard in Table 5-7 for the corresponding control scenario and power classification by the mean power (Column C), as shown in Equation 5-10.

Equation 5-10

$$\text{Low Load and Medium to High Load Emission Rate} = \text{Mean Power(hp)} \times \text{In\_use standard}\left(\frac{\text{g}}{\text{hp}\cdot\text{hr}}\right)$$

The estimated off-cycle NO_X standard compliant emission rates for heavy heavy-duty diesel vehicles are shown in Figure 5-3. Similarly, we applied Equation 5-8 through Equation 5-10 to estimate the off-cycle standard compliant emission rates for the other MOVES regulatory classes using corresponding CO_2 rates and the mean power for those vehicles.

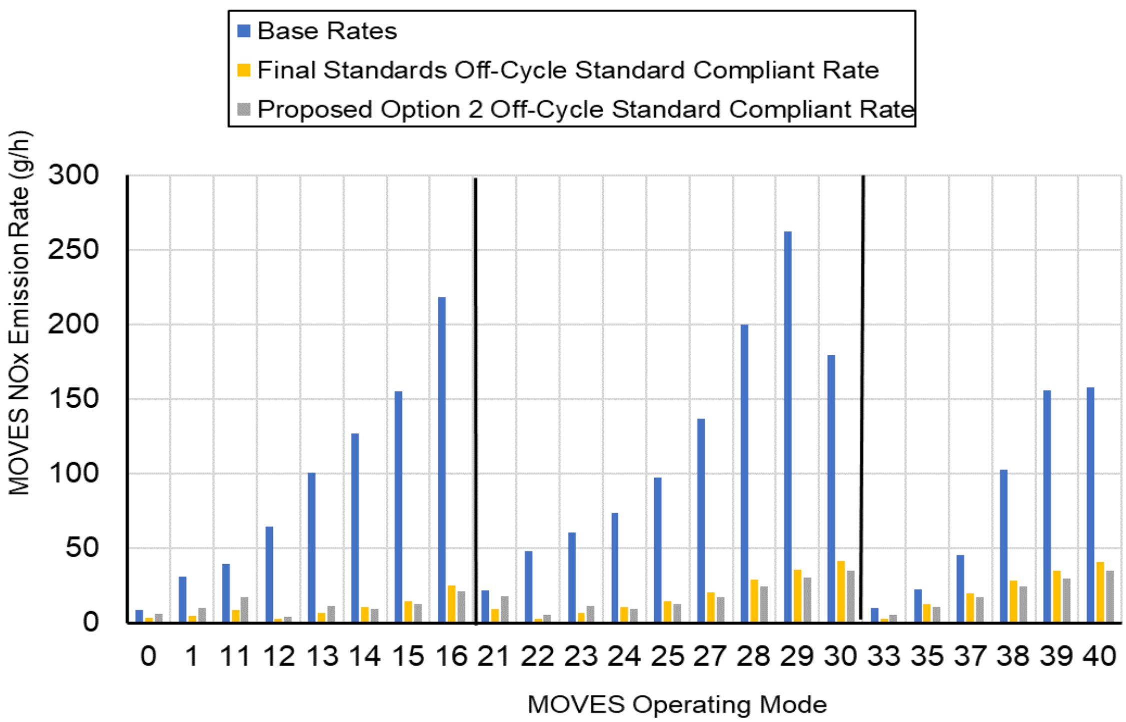


Figure 5-3: Base NO_X rates and off-cycle NO_X standard compliant emission rates for HHD diesel

5.2.2.1.1.3 *Emission Rates Based on Combination of Duty-Cycle and Off-Cycle Standards*

In this section, we document the methods used to develop MOVES NO_X emission rates for heavy-duty diesel vehicles that reflect the effects of both duty-cycle standards and off-cycle standards. As an example, Figure 5-4 shows that the final HHD duty-cycle and off-cycle standards for MYs 2027 and later affect running emission rates differently across MOVES operating modes. The duty-cycle standard is estimated to have a larger impact than the off-cycle standard in five operating modes (operating modes 0, 33, 35, 37, 38, and 40), while the off-cycle standard is estimated to have a larger impact in the remaining operating modes.

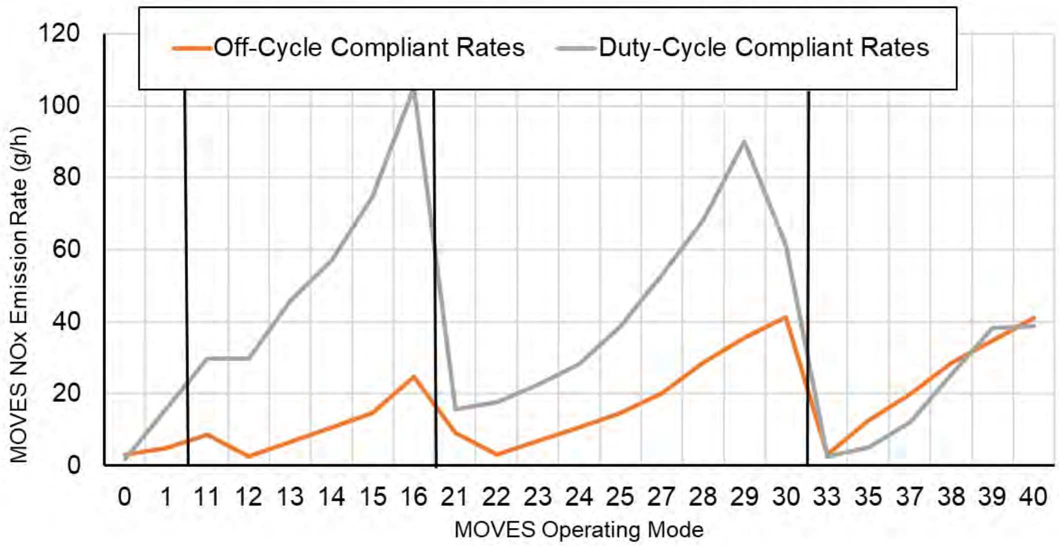


Figure 5-4: Comparison of Running NO_X emission rates for diesel-fueled HHD compliant with the MY2027+ final duty-cycle and off-cycle standards

Because manufacturers will need to comply with both the duty-cycle and off-cycle standards, we estimated the final MOVES NO_X emission rate for each operating mode as the lower of the two rates generated from the duty-cycle and the off-cycle standards (e.g., the emission rate based on the final off-cycle standards is selected for operating mode 12, but the emission rate based on the final duty-cycle standards is selected for operating mode 35). Figure 5-5 presents the estimated emission rates for HHD diesel vehicles that meet both the final duty-cycle and off-cycle standards. The same approach was used to estimate the emission rates for proposed Option 2 scenario. The final standards emission rates for MHD, LHD45 regulatory classes are shown in Appendix 5.5.1.

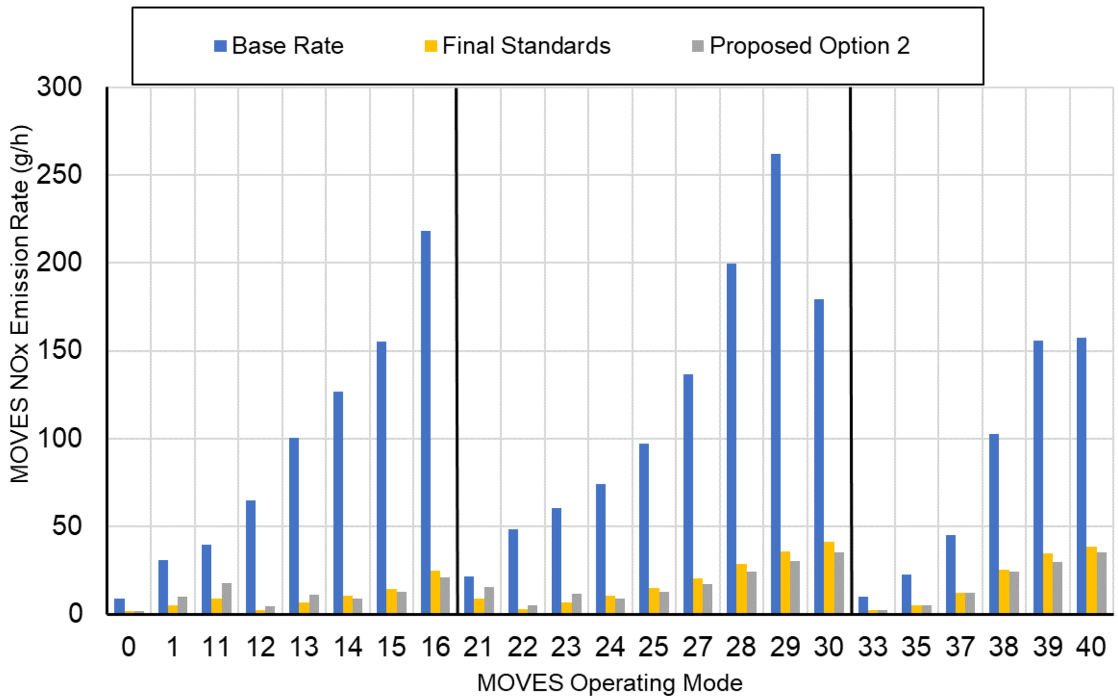


Figure 5-5: Estimated zero-mile NO_X emission rates for HHD diesel vehicles due to the final and proposed Option 2 duty-cycle and off-cycle standards

5.2.2.1.2 *Emission Rates Based on Final Changes in Warranty and Useful Life*

The MOVES NO_X, THC, CO, and $PM_{2.5}$ emission rates for heavy-duty diesel engines in the control scenarios are adjusted to reflect the useful life and warranty periods for the final standards and proposed Option 2 scenario shown in Table 5-10. The emission rate adjustments due to updated useful life and warranty periods are collectively considered adjustments to account for "age effects."

Table 5-10: Useful Life and Warranty Periods for Heavy-duty Diesel[A] Engines and Aftertreatment Systems in the Control Scenarios

Scenario	Applicable Model Years	Warranty			Useful Life		
		LHD	MHD	HHD	LHD	MHD	HHD
Baseline	Model Year 2010+	5yr/ 50k [B]	5yr/ 100k	5yr/ 100k	10yr/ 110k	10yr/ 185k	10yr/ 435k
Final Standards	Model Year 2027+	10yr/ 210k	10yr/ 280k	7yr [C] / 450k	15yr/ 270k	12yr/ 350k	11yr/ 650k
Proposed Option 2	Model Year 2027+	5yr/ 110k	5yr/ 150k	5yr/ 350k	10yr/ 250k	10yr/ 325k	10yr/ 650k

[A] The age effects for heavy-duty gasoline vehicles in MOVES are estimated directly from emissions data or adapted from light-duty gasoline or heavy-duty diesel vehicles and are not tied to warranty and useful life periods; thus, the heavy-duty gasoline or NG engine emission rates were not adjusted to account for the final and proposed Option 2 warranty and useful life periods.

[B] A warranty mileage of 100k instead of 50k was assumed in the MOVES baseline emission rates for LHD diesel, and thus, we underestimated the emissions impact of the longer warranty periods for LHD diesel vehicles in the final standards and proposed Option 2 scenarios.

[C] We analyzed a warranty years value of 7 years instead of 10 years in the final standards scenario for HHD diesel, and thus, underestimated the emissions impact of the longer warranty periods for Urban Buses in the final standards scenario.

We used the existing methodology[P] in MOVES to estimate the impact of lengthened useful life and warranty periods on heavy-duty diesel engine emissions for each of the two control scenarios (final and proposed Option 2 standards). In that approach, new vehicles/engines have zero-mile emission rates for each operating mode and maintain that rate until the age of the vehicle/engine matches the warranty period (Figure 5-6). Once the warranty period ends, the emission rate increases linearly until the vehicle/engine reaches its useful life age. At the end of the useful life, the emissions rates remain constant at a level calculated from the tampering and mal-maintenance (T&M) adjustment factor. In MOVES, we assume that tampering and mal-maintenance effects are the dominant source of emissions deterioration of fleet-wide heavy-duty diesel emissions. Although MOVES does not explicitly account for normal deterioration of heavy-duty diesel emissions, such as due to catalyst aging, tampering and mal-maintenance effects assume emission increases due to aging and deterioration.

[P] The existing methodology is documented in Appendix B "Tampering and Mal-maintenance" of the reference "Exhaust Emission Rates for Heavy-Duty Onroad Vehicles in MOVES3"**Error! Bookmark not defined.**

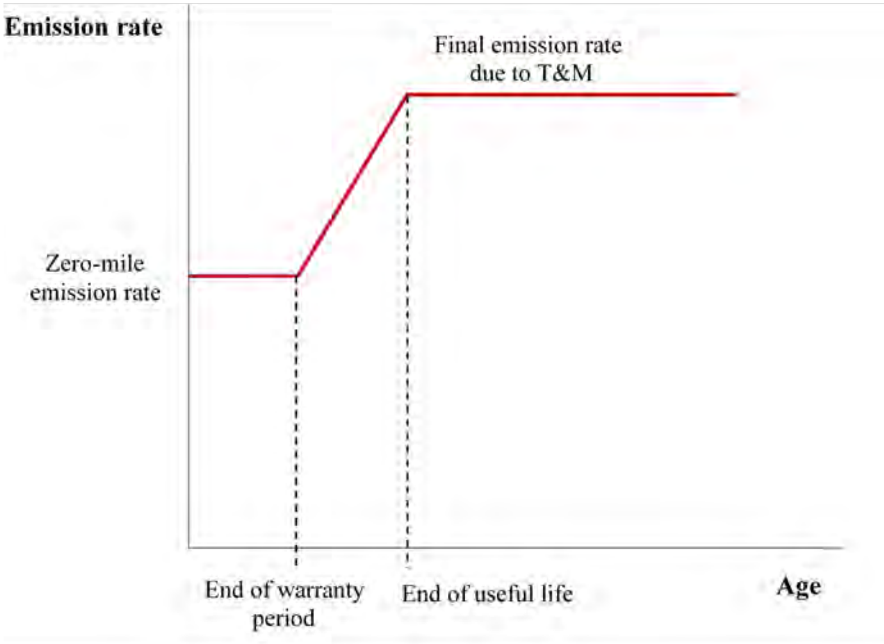


Figure 5-6: Methodology to model the effects of tampering and mal-maintenance (T&M) on emission rates according to warranty and useful life

For the baseline and control scenarios, we estimated the vehicle age at which heavy-duty diesel vehicles would reach the end of their warranty period and the end of their useful life period (Table 5-12). Table 5-11 shows example calculations for the final standards. Row (A) shows the age limit of the standards for warranty and useful life periods. Row (B) shows the mileage limit of the standards. Row (C) shows the typical miles driven per year[Q], which is used to calculate Row (D), the calculated age rounded to the nearest whole number when the mileage limit is reached. Row (E) is the smaller of the age at which the vehicle meets the end of its age limit, Row (A), or mileage limit, Row (D).

[Q] The typical miles per year used in Table 5-11 are the same values used to derive the vehicle age at the end of the warranty period and useful life in MOVES3.

Table 5-11: Estimated Vehicle Age at the End of the Warranty Period and the Useful Life for Each Heavy-duty Diesel Regulatory Class for the Final Control Scenario

Row		Warranty				Useful Life			
		LHD	MHD	HHD	Bus	LHD	MHD	HHD	Bus
(A)	Age limit	10	10	7[A]	7[A]	15	12	11	11
(B)	Mileage limit	210,000	280,000	450,000	450,000	270,000	350,000	650,000	650,000
(C)	Typical miles/year driven	26,000	41,000	105,000	44,000	26,000	41,000	105,000	44,000
(D)	Calculated age when the mileage limit is reached	8	7	4	10	10	9	6	15
(E)	Estimated age	8	7	4	7	10	9	6	11

[A] We analyzed a warranty years value of 7 years instead of 10 years in the final standards scenario for HHD diesel and Urban Bus, and thus, underestimated the emissions impact of the longer warranty periods for Urban Buses in the final standards scenario.

Similar calculations were performed for other regulatory classes for the baseline and control scenarios using the same estimated mileage per year; the resulting estimates of vehicle age at the end of the warranty and useful life periods are shown in Table 5-12.

Table 5-12: Estimated Vehicle Age at the End of the Warranty Period and the Useful Life for Each Heavy-duty Diesel Regulatory Class in the Baseline and Control Scenarios

	Warranty				Useful Life			
Vehicle age	LHD	MHD	HHD	Bus	LHD	MHD	HHD	Bus
Baseline	4	2	1	2	4	5	4	10
Final Standards Model Year 2027+	8	7	4	7	10	9	6	11
Proposed Option 2 Model Year 2027+	4	4	3	5	10	8	6	10

The T&M adjustment factor is calculated as the sum of the product of the T&M frequency for each failure *i*, and the corresponding T&M emission effect, as shown in Table 5-13.

Equation 5-11

$$f_{T\&M,p} = \sum_i \left(\text{T\&M frequency}_i \times \text{T\&M emission effect}_{p,i}\right)$$

Where:

$f_{T\&M}$ = the tampering and mal-maintenance adjustment factor for pollutant p

T&M frequency_i = estimated fleet average frequency of a tampering & mal-maintenance failure i.

T&M emission effect_i = estimated emission effect for pollutant p associated with tampering & mal-maintenance failure i.

The emission rate at the end of useful life is then calculated using Equation 5-12.

Equation 5-12

$$\mathrm{ER}_{\text{End of useful life,p,r,o}} = \mathrm{ER}_{\text{zero mile,p,r,o}} \times \left(1 + \mathrm{f}_{\text{T\&M,p}}\right)$$

Where:
$\mathrm{ER}_{\text{useful life,p,r,o}}$ = the heavy-duty diesel emission rate at the end of warranty for each pollutant p, regulatory class, r, and operating mode, o
$\mathrm{ER}_{\text{zero mile}}$ = the zero-mile heavy-duty diesel emission rate for each pollutant p, regulatory class, r, and operating mode, o
$\mathrm{f}_{\text{T\&M}}$ = the tampering and mal-maintenance adjustment factor for each pollutant p (Equation 5-11)

We used both the T&M frequency values and T&M emission effects for THC, CO, and $PM_{2.5}$ in MOVES3 for the baseline and control scenarios.**Error! Bookmark not defined.**

For the NO_X T&M emissions effects in the baseline scenario, we used the existing MOVES3 emission effects shown in Table 5-13, but for the control scenarios, we adjusted the emission effects to reflect the final numeric standards. As NO_X emissions become more tightly controlled with the application of advanced technologies to meet the final standards, we anticipate the NO_X T&M emission effects will increase (i.e., there will be a relatively larger impact of T&M because the emission control system is reducing a greater percentage of the NO_X produced by the engine). To estimate the NO_X T&M emission effects for the control scenarios, we first calculated the average zero-mile NO_X emission rate $\overline{ER}_{zero\ mile,NOx}$ based on the weighted average of the different operating modes o, and regulatory class r, using Equation 5-13.

Equation 5-13

$$\overline{\mathrm{ER}}_{\text{zero mile,NOX}} = \frac{\sum_{r,o}\left(ER_{zero\ mile,NOx,r,o} \times t_{r,o}\right)}{\sum_{r,o} t_{r,o}}$$

Where:
$\overline{ER}_{zero\ mile,NOx}$ = the average heavy-duty diesel NO_X emission rate
$\mathrm{ER}_{\text{zero mile,NOX,r,o}}$ = the zero-mile heavy-duty diesel NO_X emission rate for regulatory class, r, and operating mode, o
$t_{r,o}$= operation time by regulatory class and operating mode estimated by MOVES3 for calendar year 2045

Next, we estimated the NO_X emission rate of vehicles with a tampering and mal-maintenance failure i, using Equation 5-14, which was derived from Equation 5-12 using the fleet average emission rate from Equation 5-13 assuming the T&M frequency is 100 percent.

Equation 5-14

$$\overline{\mathrm{ER}}_{\text{T\&M i},NO_X} = \overline{\mathrm{ER}}_{\text{zero mile},NO_X} \times \left(1 + \text{T\&M emission effect}_{\text{i},NO_X}\right)$$

We then derived Equation 5-15, assuming that a NO_X aftertreatment equipment failure i, in the control scenario, would cause the average of the failed emission rates, $\overline{ER}_{T\&M\ i,NO_X}$, to be the same as a NO_X aftertreatment failure in the baseline case, Baseline $\overline{\mathrm{ER}}_{\text{T\&M i},NO_X}$.

Equation 5-15

$$\text{Baseline } \overline{\text{ER}}_{\text{T\&M i},NO_X} = \text{Control } \overline{\text{ER}}_{\text{T\&M i},NO_X}$$

$$\text{Baseline } \overline{\text{ER}}_{\text{zero mile},NO_X} \times \left(1 + \text{Baseline T\&M emission effect}_{\text{i},NO_X}\right) = \text{Control } \overline{\text{ER}}_{\text{zero mile},NO_X} \times \left(1 + \text{Control T\&M emission effect}_{\text{i},NO_X}\right)$$

By rearranging Equation 5-15, we derived Equation 5-16 to estimate the control scenario NO_X T&M emissions effects.

Equation 5-16

$$\text{Control T\&M emission effect}_{\text{i},NO_X} = \left[\frac{\text{Baseline } \overline{\text{ER}}_{\text{zero mile},NO_X} \times \left(1 + \text{Baseline T\&M emission effect}_{\text{i},NO_X}\right)}{\text{Control } \overline{\text{ER}}_{\text{zero mile},NO_X}}\right] - 1$$

Table 5-13 presents the T&M NO_X emission effects for the NO_X aftertreatment failures for the control scenarios calculated from Equation 5-16. The T&M NO_X emission effects for the NO_X aftertreatment failures are much larger than the baseline scenario, because the zero-mile NO_X emission rate in the control scenarios are lower than the baseline zero-mile NO_X emission rates. As shown in Table 5-13, the NO_X T&M emission effects for the other T&M failures (e.g., Timing Advanced and EGR Disabled/Low-Flow) in the control scenarios use the same NO_X T&M emissions effects as the baseline.

Table 5-13: NO_X Tampering & Mal-maintenance (T&M) Emission Effects for HHD

	Baseline	**Final Standards MY 2027+**	**Proposed Option 2 MY 2027+**
Timing Advanced	6%	6%	6%
Timing Retarded	-20%	-20%	-20%
Injector Problem (all)	-1%	-1%	-1%
Puff Limiter Mis-set	0%	0%	0%
Puff Limited Disabled	0%	0%	0%
Max Fuel High	0%	0%	0%
Clogged Air Filter	0%	0%	0%
Wrong/Worn Turbo	0%	0%	0%
Intercooler Clogged	3%	3%	3%
Other Air Problem	0%	0%	0%
Engine Mechanical Failure	-10%	-10%	-10%
Excessive Oil Consumption	0%	0%	0%
Electronics Failed	0%	0%	0%
Electronics Tampered	8%	8%	8%
EGR Stuck Open	-20%	-20%	-20%
EGR Disabled/Low-Flow	5%	5%	5%
NO_X Aftertreatment Sensor[A]	**200%**	**1505%**	**1407%**
Replacement NO_X Aftertreatment Sensor [A]	**200%**	**1505%**	**1407%**
NO_X Aftertreatment Malfunction [A]	**500%**	**3111%**	**2914%**
PM Filter Leak	0%	0%	0%
PM Filter Disabled	0%	0%	0%
Oxidation Catalyst Malfunction/Remove	0%	0%	0%

[A] For detailed descriptions of these failure modes, refer to Appendix B in Exhaust Emission Rates for Heavy-Duty On-road Vehicles in MOVES3.**Error! Bookmark not defined.**

Using the NO_X T&M emission effects in Table 5-13, we then calculated T&M adjustment factors $f_{T\&M,NOX}$ for each scenario using Equation 5-11 and the baseline T&M frequency values. For THC, CO, and $PM_{2.5}$, we used the existing T&M adjustment factors $f_{T\&M,p}$ in MOVES3. Then, we calculated the heavy-duty diesel emission rate for each pollutant p, age a, regulatory class r, and operating mode o, using Equation 5-17.

Equation 5-17

$$ER_{p,r,a,o} = ER_{zero\ mile,p,r,o} \times (1 + s_a \times f_{T\&M})$$

Where:
$ER_{p,r,o,a}$ = the heavy-duty diesel emission rate for each pollutant p, regulatory class r, age a, operating mode, o,
$ER_{zero\ mile}$ = the zero-mile heavy-duty diesel emission rate for each pollutant p, regulatory class r, operating mode, o
s_a = scaled age effect at age a
$f_{T\&M}$ = the tampering and mal-maintenance adjustment factor (Equation 5-11)

The scaled age effect, s_a, is calculated using the age of the vehicle in comparison to the warranty and useful life requirements, as shown in Table 5-14. When the vehicle age is between the end of the warranty and the useful life, s_a is interpolated between 0 and 1.

Table 5-14: Calculation of s_a

s_a	Where:
0	$age \leq$ end of warranty age
$\frac{(\text{age} - \text{end of warranty age})}{(\text{Useful life age} - \text{end of warranty age})}$	end of warranty age $<$ age $<$ useful life
1	age $\geq$ useful life

As the final step, the age-adjusted emission rates calculated in Equation 5-17 were averaged according to the age ranges shown in Table 5-15 that are used to define emission rates in MOVES for LHD45, MHD, HHD, and Urban Bus regulatory classes. The resulting age-adjusted running emissions have a relationship with vehicle age as shown in Figure 5-7 and Figure 5-8 for HHD NO_X emissions.[R]

Table 5-15: MOVES ageGroupID Which Are Used to Define Running and Start Emission Rates

ageGroupID	Lower bound (years)	Upper bound (years)
3	0	3
405	4	5
607	6	7
809	8	9
1014	10	14
1519	15	19
2099	20	30

5.2.2.1.3 *Summary of Diesel NO_X Running Emission Rates*

Figure 5-7 shows average running NO_X emission rates (g/mile) in MOVES3 for the model year 2027 fleet across vehicle age for the baseline and control scenarios. The MOVES running emission rates for the control scenarios reflect the adjustments to the duty-cycle and off-cycle standards, and the extended warranty and useful life as discussed above in Chapter 5.2.2.1.1 and Chapter 5.2.2.1.2, respectively. The gram per mile average running emissions are also a function of the default activity assumptions in MOVES3.[6]

Figure 5-7 shows that the average zero-mile NO_X emission rates for the control scenarios are significantly lower than the baseline scenario. The figure also demonstrates the larger T&M NO_X

[R] The average emission rate accounts for the frequency of different operating modes according to MOVES estimate of in-use vehicle activity. The trend in individual operating modes will be slightly different than the average trend shown in Figure 5 7. For example, the zero-mile idle operating mode is not reduced as much as the average emission rates in the control scenarios. Because the control case T&M emission effects were calculated using the average emission rates in Equation 5 16, individual emission rates in the control case, such as for idle, can be higher than the baseline scenario when fully aged. This is a feature of the method used to derive the aging effects, but the effect is averaged out when conducting emission inventory analysis.

emission effect for the control scenarios than for the baseline scenario as explained in Chapter 5.2.2.1.2. Although not shown, the emission rate is constant from age 15 through age 30.

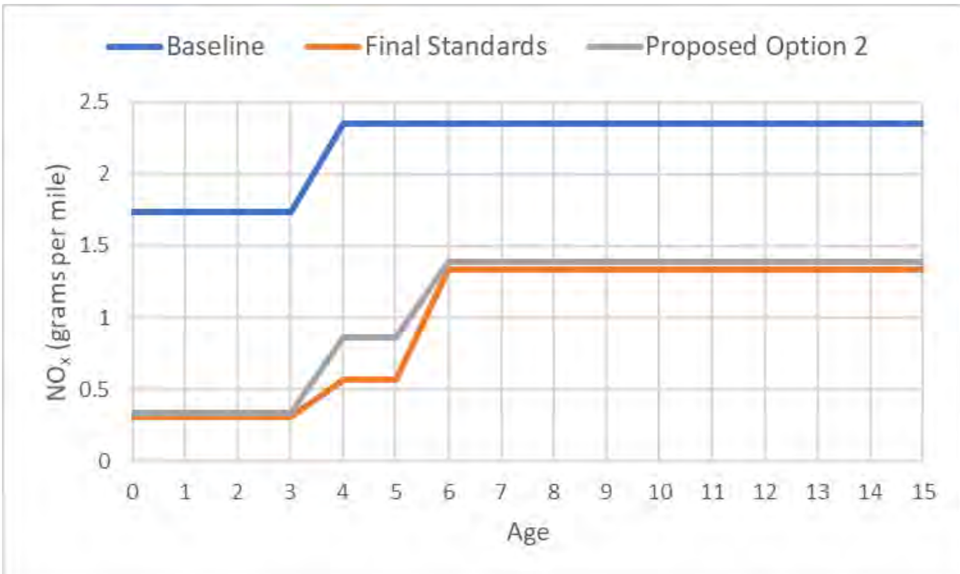


Figure 5-7: NO_X emission rates (g/mile) in MOVES for HHD diesel long-haul combination trucks for the model year 2027 fleet across vehicle age for the baseline and control scenarios

5.2.2.2 Heavy-Duty Diesel Start Emission Rates

In this section, we describe the methods used in MOVES3 to estimate lower start NO_X emission rates for the control scenarios due to the final and proposed Option 2 duty-cycle standards. We did not estimate the impact of the off-cycle standard on start emissions, in part because the baseline MY 2010 and later start emission rates in MOVES3 are not based on in-use data but are based on emissions data from the FTP duty-cycle.[Error! Bookmark not defined.] Additionally, because the baseline heavy-duty diesel start emission rates in MOVES3 do not vary with age due to insufficient data[Error! Bookmark not defined.], we did not estimate changes due to the changes in warranty and useful life in the control scenarios.

Start emission rates in MOVES are defined by regulatory class, fuel type, vehicle age and operating mode. Start operating modes in MOVES are also defined by different lengths of engine soak time (the time between the preceding engine off event and the engine start). The length of soak time for each MOVES operating mode bin is defined in Figure 5-9. Operating mode 108 represents a start with a soak longer than 720 minutes or 12 hours and is referred to as a 12-hour cold-start.

To estimate the start emissions under the control scenarios, we estimated the NO_X cold start emission rate (g/start) from the CARB Stage 1 HDD engine tested on the FTP duty-cycle cycle. Table 5-16 contains the NO_X Cold and Hot FTP measurements in Columns (B) and (C) for different aging lengths. Cold - Hot, Column (E), is calculated as the difference between the Columns (B) and (C). The cold start, Column (F), is then calculated by multiplying the difference in Column (E) by the work performed on the FTP cycle, Column (D), as shown in Equation 5-18.

Equation 5-18

$$NO_X\ \text{Cold Start}\ \left(\frac{g}{\text{start}}\right) = \left[\text{Cold}\ \left(\frac{g}{\text{hp}\cdot\text{hr}}\right) - \text{Hot}\left(\frac{g}{\text{hp}\cdot\text{hr}}\right)\right] \times \text{FTP work}\ (\text{hp}\cdot\text{hr})$$

Table 5-16: Calculation of NO_X 12-hour Cold Starts from the CARB Stage 1 HHD Engine from the Cold and Hot FTP Cycle

	(A)	(B)	(C)	(E)	(D)	(F)
Aged hours	**FTP composite (g/hp-hr)**	**Cold (g/hp-hr)**	**Hot (g/hp-hr)**	**Cold - Hot (g/hp-hr)**	**FTP Work (hp-hr)**	**Cold Start (g/start)**
0	0.008	0.025	0.005	0.02	31.4	0.63
333	0.012	0.042	0.006	0.036	31.4	1.13
656	0.018	0.061	0.009	0.052	31.4	1.64
1000	0.024	0.092	0.01	0.082	31.4	2.58
1000 hr Post Ash Clean	0.026	0.109	0.009	0.1	31.4	3.14

The Stage 1 HHD engine is deemed representative of an engine-certified to a 0.02 g/hp-hr NO_X standard based on the FTP composite measurements in Column (A). Table 5-16 demonstrates that there was larger cold start measured with increase in aged hours, and after the ash clean out at 1000 hours. We used the 1000 hr, Post Ash Clean cold start emission rate (3.14 g/start shown in Table 5-16) to represent the 12-hour cold-start (operating mode 108) emission rate.

To estimate the 12-hour cold-start NO_X emission rate for HHD diesel vehicles in the control scenarios, we interpolated the HHD 12-hour cold-start between the Stage 1 cold start (3.14 g/start) and the MOVES baseline 12-hour cold-start (8.4 g/start), and their respective FTP duty-cycle standards using Equation 5-19 as shown in Figure 5-8 and Table 5-17. For example, the interpolation yielded an estimated 12-hour cold start of 4.02 g/start for the 0.05 g/hp-hr FTP standard.

Equation 5-19

$$\text{Start ER}_{\text{FTPx,HHD,12 hour}} = \left(\frac{\text{MOVES start}_{\text{HHD,12 hour}} - \text{Stage1 start}}{\text{Baseline FTP} - \text{Stage1 FTP}}\right) \times (\text{FTP}_\text{x} - \text{Baseline FTP}) + \text{MOVES start}_{\text{HHD,12 hour}}$$

Where:
$\text{Start ER}_{\text{Duty-cycle standard x,HHD,12 hour}}$ = the estimated NO_X start emissions for an FTP duty-cycle standard, x, for heavy heavy-duty diesel emissions for a 12-hour cold-start (operating mode 108).
Stage1 start = 1000 Post Ash Clean start emission rate from the CARB Stage 1 HHD diesel engine = 3.14 g/start (Table 5-16)
Stage1 FTP = Composite FTP level of the CARB Stage 1 engine = 0.02 g/hp-hr
$\text{MOVES start}_{\text{HHD,12 hour}}$ = MOVES3 baseline start emission rate (= 8.4 g/start) for MY 2027 heavy heavy-duty diesel engine for a 12-hour soak (operating mode 108)
Baseline FTP = baseline FTP composite NO_X standard = 0.2 g/hp-hr
FTP_x = composite FTP standard in the control scenarios, either 0.035, 0.029, 0.05, or 0.02 g/hp-hr (Table 5-4)

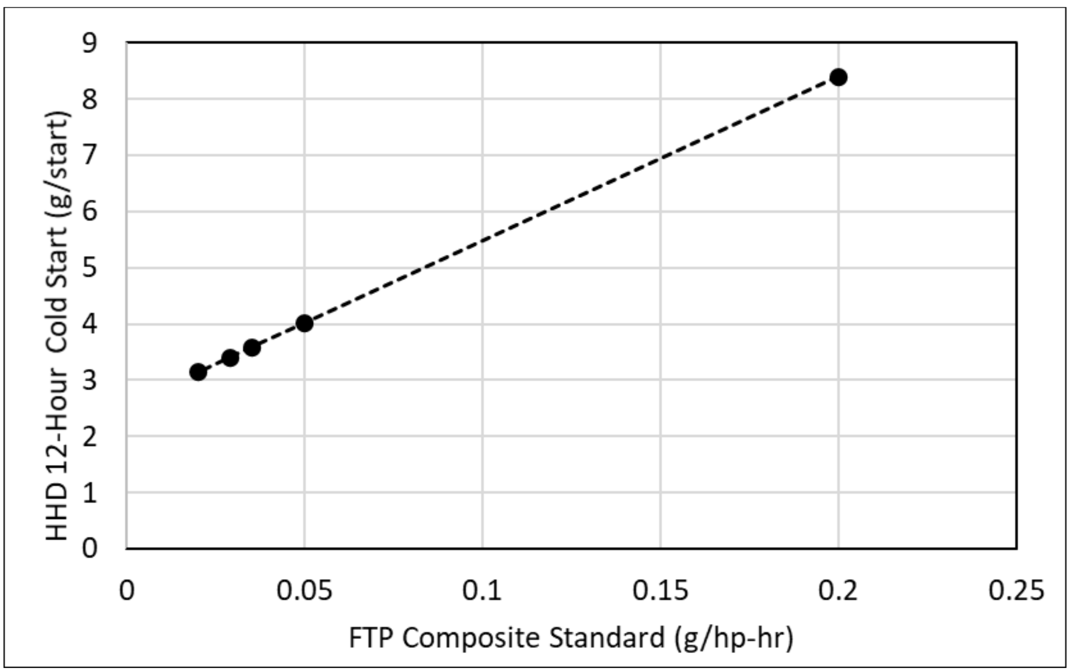


Figure 5-8: Estimated relationship between the HHD NO_X 12-hour cold-start and the composite FTP NO_X standards

Table 5-17: HHD Cold Start Emissions for Baseline and Control Scenarios

Scenario	**Applicable Model Years**	**Weighted Average FTP standard (g/hp-hr)**	**Cold Start emissions (g/start)**
Baseline	Model Year 2010+	0.2	8.40
Final Standards	Model Year 2027+	0.05	4.02
Proposed Option 2	Model Year 2027+	0.05	4.02

We then used Equation 5-20 to estimate the MOVES NO_X emission rates for each MOVES heavy-duty regulatory class (LHD45, MHD, and HHD) and for each MOVES start operating mode classified by different soak times (Figure 5-9). We assumed that the relative difference in emission rates by regulatory class and by operating mode is the same in the baseline and control scenarios.

Equation 5-20

$$\text{Start ER}_{\text{FTP=x ,reg class=y,soak=z}} = \text{Start ER}_{\text{Duty cycle standard x,HHD,12}} \times \left(\frac{\text{MOVES start}_{\text{reg class=y,soak=z}}}{\text{MOVES start}_{\text{HHD,12-hour}}} \right)$$

Where:
$\text{Start ER}_{\text{FTP}}$ = the start NO_X emission rates for the control scenarios with FTP x (0.035, 0.029, 0.05, or 0.02) for regulatory class y (LHD45, MHD, and HHD), and soak length z
$\text{Start ER}_{\text{Duty cycle standard x,HHD,12-hour}}$ = the estimated start emissions for an FTP duty-cycle standard, x, for heavy heavy-duty diesel emissions for a 12-hour soak (operating mode 108)
$\text{MOVES start}_{\text{reg class=y,soak=z}}$ = MOVES3 baseline start emission rate for MY 2027 for regulatory class y (LHD45, MHD, and HHD), and soak length z
$\text{MOVES start}_{\text{HHD,12-hour}}$ = MOVES3 baseline start emission rate for MY 2027 HHD diesel engine for a 12-hour soak (operating mode 108)

Figure 5-9 shows the estimated MOVES NO_X start emission rates for HHD diesel vehicles for the baseline scenario, as well as the final standards and proposed Option 2 scenarios.

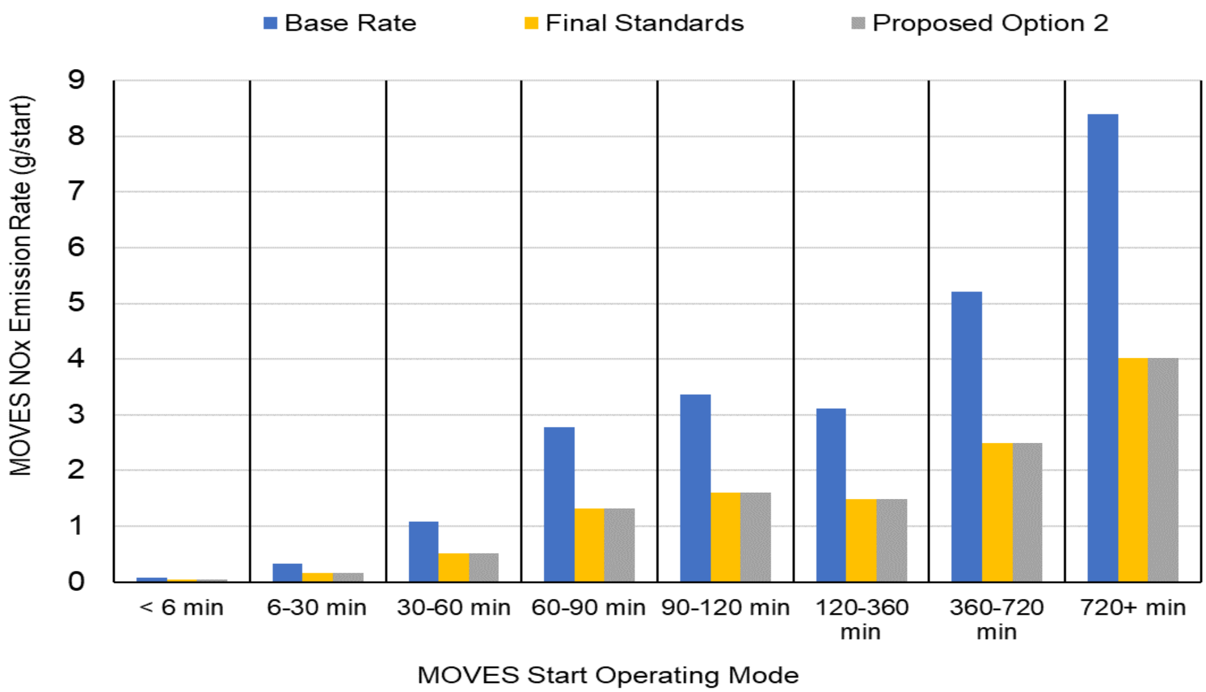


Figure 5-9: Duty-cycle-based NO_X start emissions for HHD Diesel for the baseline, final standards and proposed Option 2 scenarios

5.2.2.3 Heavy-Duty Extended Idle Emission Rates

In MOVES, extended idling is defined as idling for more than an hour, which occurs during hotelling activity when long-haul combination trucks idle during rest periods. MOVES has extended idle emission rates for long-haul combination trucks that include HHD, MHD[S] and glider vehicles[T]. All idling activity by other regulatory classes is modeled using the running idle emission rates (Table 5-3), which are different than extended idle emission rates. We anticipate that reductions in the HHD and MHD NO_X extended idle emissions rates will be driven by the idle standard, rather than the duty-cycle standards in the final rule. The duty-cycle standards do not contain high duration extended idling (> 1 hour) that is representative of truck hotelling activity. In addition, we did not estimate lower extended idle emission rates due to the lengthened warranty or useful life periods.[U]

First, we estimated extended idle emission rates that would comply with the off-cycle NO_X/CO_2 g/kg standard calculated in Table 5-8. We then used Equation 5-9 to calculate the extended idle off-cycle NO_X g/hr emission rate based on the MOVES extended idle CO_2 g/hr emission rate, as shown in Table 5-18.

[S] HHD and MHD have the same extended idle emission rates in MOVES.
[T] We assumed there are no changes to glider emission rates due to the final rule.
[U] Extended idle emission rates in MOVES are not differentiated by vehicle age.

Table 5-18: Calculation of HHD and MHD Extended Idle NO_X g/hr Emission Rates

Control Scenario	Model Years	2027 MY Baseline Rates NO_X (g/hr)	2027 MY Baseline Rates CO_2 (g/hr)	Idle Standard (g/hr)[A]	Idle Standard NO_X/CO_2 (g/kg)	Idle-standard compliant NO_X emission rate (g/hr)	% Change in NO_X emission rate
Final Standards	Model Year 2027+	42.6	7191	5	0.65	4.68	-89%
Proposed Option 2	Model Year 2027+	42.6	7191	10	1.30	9.36	-78%

[A] Note that the voluntary idle standard in the final control scenario we modeled is different than the voluntary idle standard in the final program, see preamble Section III.B for details on the voluntary idle standard in the final program.

5.2.2.4 Heavy-duty Diesel Crankcase Emissions

Since the proposal, we made improvements in MOVES to better model the emissions from crankcase for both open and closed systems, by including the fraction of the fleet with closed crankcase systems in the baseline as well as incorporating more recent data from MY2027 and later HD vehicles **Error! Bookmark not defined.**. With these updates, the inventory analysis done for the final rule estimates more accurately the emissions benefits of the closed crankcase for THC, CO, NO_X, and $PM_{2.5}$.

As described in Section III.B of the preamble, EPA is finalizing a requirement for manufacturers to use one of two options for controlling crankcase emissions, either: 1) as proposed, closing the crankcase, or 2) an updated version of the current requirements for an open crankcase that includes additional requirements for measuring and accounting for crankcase emissions. For the emissions impact analysis of the final standards and proposed Option 2 presented in Chapter 5.3 below, the emission reductions were estimated assuming that closing the crankcase would be the preferred option to meet the final standards.

In modeling the control scenarios, the $PM_{2.5}$ crankcase emissions from HHD, MHD, and LHD45 diesel vehicles were set to zero. For LHD2b3 diesel vehicles, we reduced the crankcase emissions by 94.9%, assuming that 5.1% of the LHD2b3 diesel vehicles are engine-certified (see Chapter 5.2.2.5).

5.2.2.5 Light Heavy-Duty Class 2b and 3 Diesel Emission Rates

We assumed that in 2027 and later model years, 5.1 percent of the diesel-fueled LHD2b3 vehicles will be engine-certified and therefore, will be impacted by the final rule.[V,W] To develop the emission rates for LHD2b3 vehicles, for the control scenarios, we assumed that 5.1 percent of the emissions from LHD2b3 are equivalent to the controlled emissions of LHD45 regulatory class vehicles. This is consistent with the analysis for model year 2010 and later diesel vehicles, where we used the same data from the HDIUT program to estimate emission rates for both LHD2b3 and LHD45 vehicles.**Error! Bookmark not defined.** In addition, we assumed that the final duty-cycle, off-cycle, and warranty and useful life requirements are the same for all engine-certified LHD vehicles.

We did not estimate the contribution of engine-certified vehicles on the emission rates for diesel-fueled LHD2b3 in the baseline scenario.[X] Because the LHD2b3 diesel emission rates certified to engine standards are higher than the emission rates certified to chassis standards, the control scenarios generally increase NO_X emissions compared to the baseline scenario for diesel LHD2b3 vehicles for most calendar years, even though we anticipate the final rule will reduce NO_X emissions for engine-certified LHD2b3 vehicles. We included the emission contribution from engine-certified diesel-fueled LHD2b3 vehicles in the control scenarios to better account for future NO_X emissions from these vehicles. We acknowledge that we are underestimating the benefits of controlling these vehicles due to their absence from the baseline scenario.

5.2.2.6 Heavy-Duty Gasoline Running Emission Rates

In this section, we describe the methods used to develop the running exhaust emission rates for NO_X, THC, CO, and $PM_{2.5}$ from heavy-duty gasoline vehicles in MOVES for the control scenarios. The final rule does not include off-cycle standards for vehicles fueled by gasoline or NG. Furthermore, even though we anticipate emission benefits from the lengthened warranty and useful life periods from gasoline and NG-fueled vehicles, they were not included in the analysis done for the final rule.

[V] Class 2b and 3 vehicles with GVWR between 8,500 and 14,000 pounds are primarily commercial pickup trucks and vans and are sometimes referred to as "medium-duty vehicles." The majority of Class 2b and 3 vehicles are certified as complete vehicles and which EPA intends to include in a future combined light-duty and medium-duty rulemaking action, consistent with Executive Order 14037, Section 2a.

[W] In Appendix 5.5.1 of the draft RIA, we presented an analysis suggesting that 4.2% of MY 2027 diesel-fueled LHD2b3 vehicles would be engine-certified. However, we used 5.1% in developing the MOVES rates for LHD2b3 vehicles and subsequent inventory analysis (including this final rulemaking analysis). Given the small contribution of engine-certified LHD2b3 to the total emissions inventory, we expect this would have only a negligible impact on the emission reductions estimated in the final rule. In addition, we deem that both estimates (4.2% and 5.1%) are within the range of feasible values for the fraction of engine-certified LHD2b3 vehicles in future years.

[X] In the baseline case created for MOVES3, we assumed, for simplicity, that 100% of diesel-fueled LHD2b3 vehicles are chassis-certified and are subject to the light-duty Tier 3 emission standard. Note that the estimated NO_x emission rates for engine-certified diesel LHD2b3 vehicles (subject to the final rule) are higher than chassis-certified diesel LHD2b3 vehicles (subject to the light-duty Tier 3 standard).

The final FTP duty-cycle standards shown in Table 5-4 apply to both heavy-duty compression-ignition engines and heavy-duty spark-ignition engines.[Y] For the control scenarios, we updated the NO_X exhaust emission rates for gasoline, assuming that emissions are reduced for all operating modes based on the reduction in the NO_X FTP standards from the current 0.2 g/hp-hr standard. Table 5-19 shows the estimated reduction in NO_X emission rates, which is consistent with the ratio of the current FTP emission standards and the final and proposed Option 2 FTP standards shown in Table 5-5.

In addition to modeling the final and proposed Option 2 standards for NO_X, we estimated emission rate reductions due to the final and proposed Option 2 standards for HC, CO and $PM_{2.5}$. As discussed in the preamble Section III.D, the final emissions standards for HC, CO, and $PM_{2.5}$ heavy-duty spark ignition are lower than the current MY 2010 standards. We estimated reduced THC and CO emission rates assuming that those emissions would be reduced due to improvements in the three-way catalyst emission controls. We used initial data from our production HD SI engines and from the heavy-duty gasoline technology demonstration program presented in Chapter 3.2 to estimate our modeled emissions levels. We assumed a 65 percent reduction in THC emissions would occur at a NO_X standard of 0.1 g/hp-hr.[Z] We assumed additional decreases in THC emissions to reflect tighter final NO_X standards in the control scenarios in MY 2027. We derived Equation 5-21 assuming a linear decrease in THC emissions between the estimated THC emissions emitted at the 0.1 g/hp-hr NO_X FTP level, and zero THC emissions at a hypothetical 0 g/hp-hr NO_X FTP level. We then used Equation 5-21 to estimate the reductions in THC emissions using the NO_X levels for the control scenarios (Table 5-19).

Equation 5-21

$$R_{gasoline,THC,NOx\,FTP} = 1 - \left(\frac{NOx\ FTP\ Standard}{0.1\,\frac{g}{bhp \cdot hr}}\right) \times \left(1 - R_{gasoline,THC,0.1\,NOx\,FTP}\right)$$

$$= \quad 1 - \left(\frac{NOx\ FTP\ Standard}{0.1\frac{g}{bhp \cdot hr}}\right) \times (1 - 65\%)$$

Where:
$R_{gasoline,THC,NO_X\,FTP}$ = percent emission reductions in heavy-duty gasoline THC emissions for NO_X FTP standards more stringent than the 0.1 NO_X FTP standard, calculated values shown in Table 5-19
NO_X *FTP Standard* = NO_X FTP standards in the control scenarios

We assumed a single CO standard for MY 2027 and later HD SI engines and we maintained a 60 percent reduction in CO for all scenarios (see Table 5-19). To meet the final PM standards, manufacturers are expected to improve fuel control and limit the need for catalyst protection. Therefore, we assumed a 50 percent reduction in $PM_{2.5}$, consistent with the 50 percent lower PM standard, for all scenarios. Table 5-19 contains the emission rate reductions, $R_{gasoline}$, applied in MOVES for the emission inventory analysis.

[Y] Our inventory analysis for HD SI engines only evaluated the impact of the final FTP duty-cycle standards. We did not analyze the impact of our final SET duty-cycle standards or idle provisions for HD SI engines.
[Z] The scenario analyzed for the air quality modeling assumed an FTP standard at 0.1 g/hp-hr.

Table 5-19: Running Emission Rate Reductions From Heavy-duty Gasoline Vehicles Due to Final and Proposed Option 2 Standards, $R_{gasoline}$, Across All Heavy-duty Gasoline Regulatory Classes and Operating Modes

Control Scenario	**Model Years**	**Regulatory Class[A]**	**FTP/SET NO_X standard (g/hp-hr)**	**NO_X**	**THC**	**CO**	**$PM_{2.5}$**
Final Standards	2027+	LHD, MHD, HHD	0.035	82.5%	87.8%	60%	50%
Proposed Option 2	2027+	LHD, MHD, HHD	0.035	82.5%	87.8%	60%	50%

[A] We applied the same standards for the final and proposed Option 2 scenarios to represent the SI engines modeled by the LHD, MHD, and HHD regulatory classes, unlike the final standards for compression-ignition engines (Preamble Section III.D)

We used Equation 5-22 to estimate the MOVES NO_X emission rates for the control scenarios using the $R_{gasoline}$ values for heavy-duty gasoline vehicles. Since the final rule does not require an in-use testing program for spark-ignition engines, we did not estimate operating mode-specific effectiveness of reductions of the in-use emissions compared to duty-cycle standard emissions, as was done for diesel running emissions. Instead, we assumed these reductions apply uniformly across all running exhaust operating modes. As such, we used Equation 5-22 to estimate the MOVES emission rates proportionally for all operating modes.

Equation 5-22

$$ER_{control} = \left(1 - R_{gasoline}\right) \times ER_{MOVES\_baseline}$$

Where:
$ER_{control}$ = MOVES running exhaust emission rates for the control scenarios based on the reduction in the FTP duty-cycle standard
$R_{gasoline}$ = percent emission reductions in heavy-duty gasoline emissions from Table 5-19
$ER_{MOVES\ baseline}$ = MOVES running exhaust emission rates for the baseline

The estimated heavy-duty gasoline MOVES running emission rates for the baseline, final standards, and proposed Option 2 scenarios are shown for NO_X and THC emissions in Figure 5-10 and Figure 5-11, respectively. CO and $PM_{2.5}$ were similarly estimated from the reductions shown in Table 5-19, but they have the same emission rates within each regulatory class for all the control scenarios and, therefore, are not plotted.

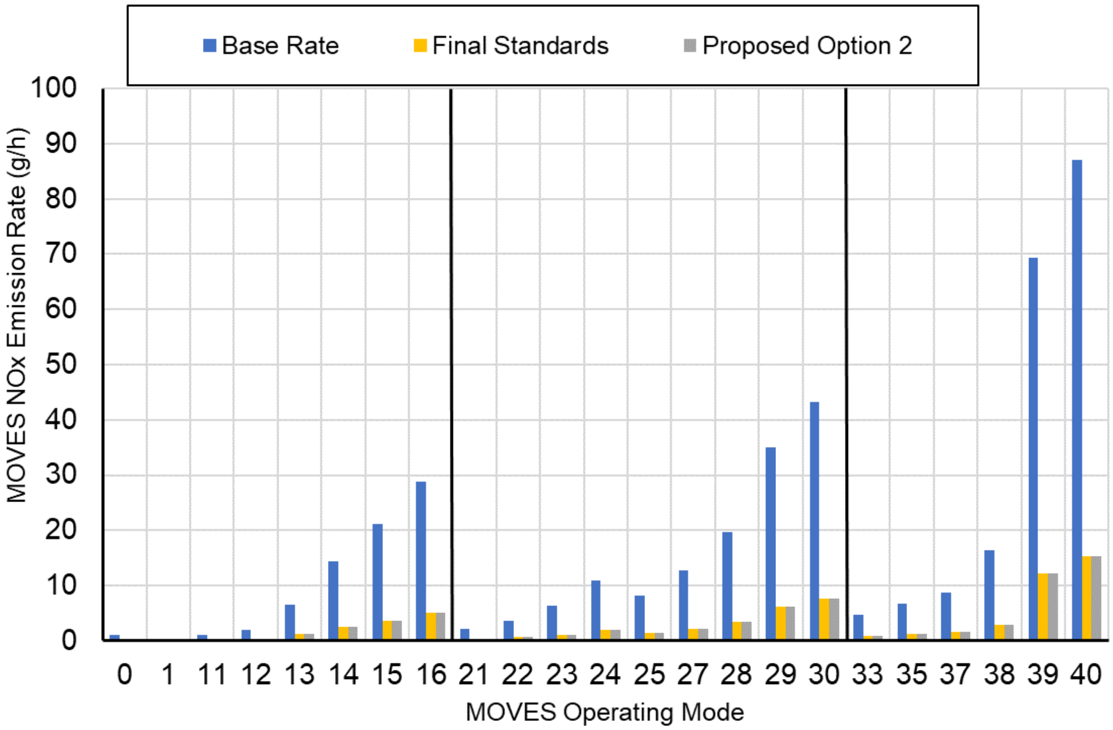


Figure 5-10: Duty-cycle-based running NO_X emission rates for LHD gasoline for the control scenarios

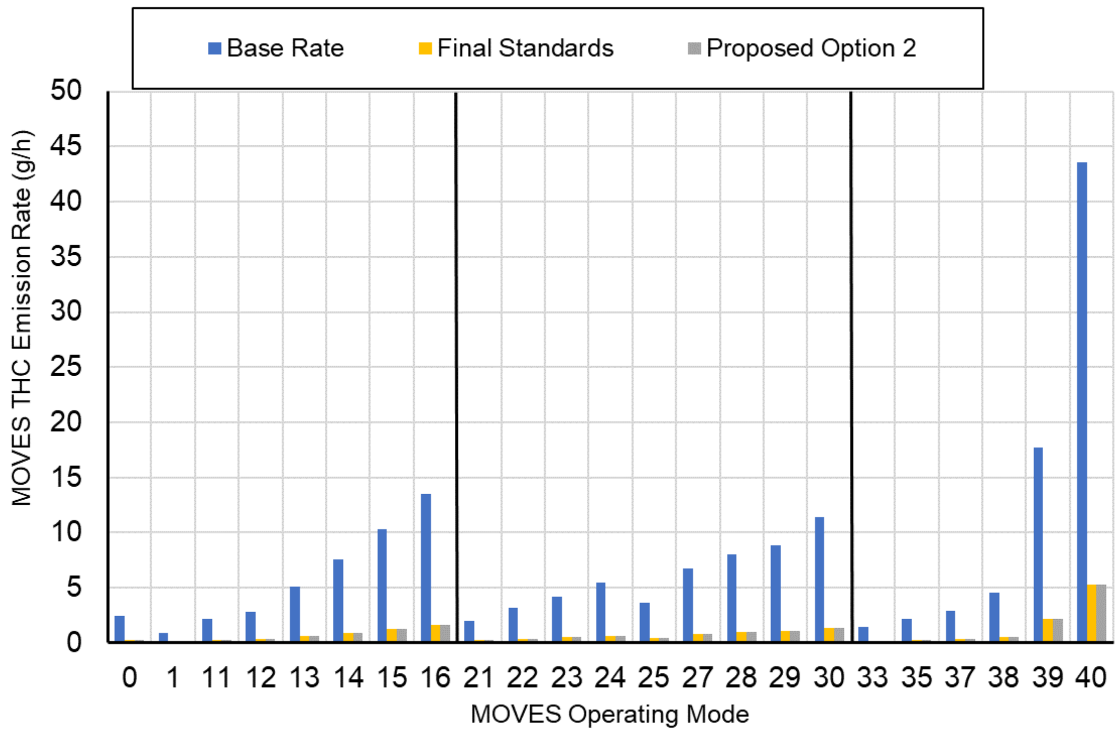


Figure 5-11: LHD gasoline Duty-cycle-based running THC emission rates for LHD gasoline for the control scenarios

5.2.2.7 Heavy-Duty Gasoline Refueling Emission Rates

In this section, we describe the methods used to estimate lower refueling emission rates in MOVES for the control scenarios due to the final Onboard Refueling Vapor Recovery (ORVR) requirements. Refueling emissions result when the pumped gasoline displaces the vapor in the vehicle tank. The THC emissions from refueling are a function of temperature and the gasoline Reid Vapor Pressure (RVP).[AA] The emissions control technology which collects the vapor from the refueling events is the ORVR system. ORVR requirements for light-duty vehicles started phasing in as part of EPA's Refueling Emission Regulations for Light-Duty Vehicles and Light-Duty Trucks Final Rule[11] in 1998. Under the EPA's Tier 2 vehicle program, all complete vehicles with a gross vehicle weight rating (GVWR) of 8,500 lbs. up to 14,000 lbs. (MOVES regulatory class LHD2b3) were required to meet the ORVR requirements between 2004 and 2006 model years.[12] With the Tier 3 rulemaking, all heavy-duty trucks up to 14,000 lbs. and all

[AA] See additional discussion of refueling updates in the Evaporative Emissions from Onroad Vehicles in MOVES3 [14] complete vehicles greater than 14,000 lbs. are required to meet a refueling standard of 0.2 grams of HC per gallon of gasoline dispensed by 2022.[13]

Table 5-20 shows the ORVR adoption rates applied to all heavy-duty gasoline trucks in MOVES. For the baseline scenario, we estimated that all heavy-duty gasoline trucks with GVWR up to 14,000 lbs. will have ORVR control by 2018 (as shown in Table 5-20). No heavy-duty gasoline vehicles over 14,000 lb GVWR are being certified today[BB] as complete vehicles, and our baseline scenario reflects a population of 100 percent incomplete vehicles that have not adopted ORVR technologies and are not expected to adopt ORVR without a regulatory action, due to the costs and added complexities.

As part of this final rulemaking, all heavy-duty gasoline vehicles, including those sold as incomplete vehicles, will be required to have an ORVR system and be certified to the same standard as light-duty by model year 2027. Therefore, for the control scenarios, we assumed manufacturers will fully implement ORVR technologies for HD vehicles over 14,000 GVWR starting in MY2027. For a 100 percent phase-in of ORVR, we estimated a 98 percent reduction in refueling emissions, because we assume that some ORVR systems would fail or may not be fully effective, similar to our assumptions made for current ORVR systems in light-duty vehicles.[14] The emissions inventory impact of the final ORVR control is summarized in Chapter 5.3.3.

Table 5-20: Phase-In of Onboard Refueling Vapor Recovery (ORVR) for Heavy-duty Trucks

Model Year	**Light Heavy-Duty Trucks 8,500-10,000 lbs GVWR (Class 2b)**	**Heavy-Duty Trucks 10,000-14,000 lbs GVWR (Class 3)**	**Heavy-Duty Trucks > 14,000 lbs GVWR (LHD45 and MHD) including incompletes**	
			Baseline	**Final Standard**
2003 and earlier	0%	0%	0%	0%
2004	40%	0%	0%	0%
2005	80%	0%	0%	0%
2006-2017	100%	0%	0%	0%
2018-2026	100%	100%	0%	0%
2027 and later	100%	100%	0%	100%

5.3 National Emissions Inventory Results

In the following sections, we present the estimated emissions impacts of the final control scenario and the proposed Option 2 in three select calendar years.[CC] The national (50 states and Washington DC, excluding Puerto Rico and the Virgin Islands) highway heavy-duty vehicle emission inventory was generated using the national-scale option in MOVES3 with the

[BB] We expect that only one complete vehicle model will exist in 2022 and it is not yet certified.

[CC] The final rule is expected to have minimal impacts on CO_2 emissions. We estimated a small fuel savings for heavy-duty gasoline vehicles in Section 7.2.2 due to ORVR control (Chapter 5.2.2.7). However, for MOVES emissions inventories, we estimated no differences in the CO_2 emission rates for the baseline and control scenarios. See RIA Chapter 1 for more discussion of the technologies evaluated to control NO_X emissions without impacting CO_2 emissions.

methodology and the model inputs as described in Chapter 5.2.[DD] For comparison, we also present the emission impacts of the Proposed Option 2 using the same methodology.

5.3.1 **Final Standards**

Table 5-21 summarizes the emission impacts of the final control scenario for three select calendar years. Chapter 5.5.4 shows NO_X, VOC, $PM_{2.5}$, and CO inventories for all calendar years between 2027 and 2045.

Table 5-21: National Emission Reductions from Heavy-duty Vehicles in Calendar Years 2030, 2040, and 2045 ─ Final Control Scenario Emissions Relative to Heavy-Duty Vehicle Emissions Baseline

Pollutant	2030		2040		2045	
	US Short Tons	% Reduction	US Short Tons	% Reduction	US Short Tons	% Reduction
NO_X	139,677	14.0%	398,864	43.5%	453,239	47.9%
VOC	5,018	4.9%	17,139	19.6%	20,758	22.6%
Primary Exhaust $PM_{2.5}$ - Total	115	0.9%	491	6.6%	566	7.7%
Carbon Monoxide (CO)	43,978	3.0%	208,935	15.5%	260,750	18.3%
Acetaldehyde	36	1.5%	124	6.0%	145	6.7%
Benzene	40	3.7%	177	23.3%	221	27.6%
Formaldehyde	29	1.1%	112	6.6%	134	7.5%
Naphthalene	2	1.0%	7	13.2%	9	15.7%

More details about the impacts of the final standards can be found in Chapter 5.5.2, where the emission reductions are categorized by vehicle fuel type with further splits by emission process and by heavy-duty regulatory class. Contributions to NO_X emissions from different engine operational processes in calendar year 2045 are also provided in Chapter 5.5.3 for both the final and proposed Option 2 control scenarios.

5.3.2 **Proposed Option 2**

Table 5-22 summarizes the emissions impacts in three selected calendar years for proposed Option 2.

[DD] Because of the differences in the control scenarios and the differences in the emission inventory methodology between the proposal and the final rule, no direct comparison should be made between the emission impacts of the final rule presented in Table 5-21 and the emission impacts estimated in the proposal.

Table 5-22: National Emission Reductions from Heavy-duty Vehicles in Calendar Years 2030, 2040, and 2045 — Proposed Option 2 Program Emissions Relative to Heavy-Duty Vehicle Emissions Baseline

Pollutant	2030		2040		2045	
	US Short Tons	% Reduction	US Short Tons	% Reduction	US Short Tons	% Reduction
NOx	133,699	13.4%	352,468	38.5%	400,024	42.3%
VOC	5,018	4.9%	17,067	19.5%	20,681	22.5%
Primary Exhaust $PM_{2.5}$ - Total	115	0.9%	443	6.0%	515	7.0%
Carbon Monoxide (CO)	43,978	3.0%	204,178	15.1%	255,653	17.9%
Acetaldehyde	36	1.5%	121	5.9%	142	6.6%
Benzene	40	3.7%	177	23.3%	221	27.6%
Formaldehyde	29	1.1%	110	6.5%	132	7.4%
Naphthalene	2	1.0%	7	13.1%	9	15.6%

5.3.3 Impacts of Heavy-Duty Gasoline Refueling Controls

Table 5-23 shows the estimated impact on refueling emissions from heavy-duty vehicles due to the final refueling emission standard. For heavy-duty vehicles, MOVES3 only estimates refueling emissions from gasoline-fueled vehicles. Thus, the reductions reflect the control of the refueling emissions from only the heavy-duty gasoline vehicles above 14,000 lbs. Because benzene is calculated as a fraction of VOC emissions, the percent reductions are the same for both pollutants as shown in Table 5-23.

Table 5-23: Emission Reductions Due to Adoption of ORVR for Heavy-Duty Vehicles Relative to Heavy-Duty Vehicle Emissions Baseline

Pollutant	Calendar Year	Reductions in US Short Tons	% Reduction
Benzene	2027	3	6.6%
	2030	13	27.8%
	2040	43	80.2%
	2045	52	88.7%
VOC	2027	890	6.6%
	2030	3,718	27.8%
	2040	11,867	80.2%
	2045	14,381	88.7%

5.4 Emissions Inventories for Air Quality Modeling

When feasible, we conduct full-scale photochemical air quality modeling to accurately project levels of criteria and air toxic pollutants, because the atmospheric chemistry related to ambient concentrations of $PM_{2.5}$, ozone, and air toxics is very complex. Air quality modeling was performed for the proposed rule and demonstrated improvements in concentrations of air pollutants. We did not perform new air quality modeling for this final rule. This section of the RIA describes the emissions inventories that were used in the air quality modeling and presents the differences between the air quality modeling emissions inventories and those developed for the final rule. As further described in this Chapter 5.4, despite the differences in the version of the MOVES model used and the control scenario modeled, the emission reductions used in the air quality modeling analysis for the proposed rule compare well with the emission reductions estimated for the final standards.

The air quality modeling analysis required emission inventories with greater geographical and temporal resolution than the national-scale inventories described in Chapter 5.3. The approach for estimating emission inventories for an air quality modeling analysis is extremely complex and time and resource intensive since it involves modeling of each 12 km grid cell, for each hour of the day, for the entire year.

The methodology for developing the onroad emission inventories for the air quality modeling, also referred to as SMOKE-MOVES emission inventories, is summarized here. Additional details, including information for sectors other than onroad vehicles, are available in the Air Quality Modeling Technical Support Document (AQM TSD).[15] Figure 5-12 illustrates the process involved in generating the onroad emissions inventories for use in air quality modeling. First, MOVES county-level onroad emission factors (EF) by temperature and speed bins are generated based on the output from the meteorological preprocessor, Met4Moves, which is used to develop the temperature ranges, relative humidity, and temperature profiles. As discussed below, the MOVES emission rates for the air quality modeling inventory differed from the emission rates discussed in Chapter 5.2.2. Additionally, other MOVES data inputs for the county-level runs used for the air quality modeling sometimes differ from the inputs used for the national inventory. For example, the county-level MOVES runs incorporate county-specific inputs including: fuel programs, inspection and maintenance programs, adoption of LEV standards, and the age distribution of the local vehicle fleet. The emission factors for each representative county were generated with multiple runs of the MOVES CTI NPRM version of the model.

The MOVES-generated onroad emission factors were then combined with activity data to produce emissions within the Sparse Matrix Operator Kernel Emissions (SMOKE) modeling system. The collection of tools that compute the onroad mobile source emissions (as one of the sectors included in the air quality modeling) are known as SMOKE-MOVES. SMOKE-MOVES uses a combination of vehicle activity data, emission factors from MOVES, meteorology data, and temporal allocation information to estimate hourly onroad emissions. Additional types of ancillary data are used for the emissions processing, such as spatial surrogates which spatially allocate emissions to the 12 km grid cells used for air quality modeling.

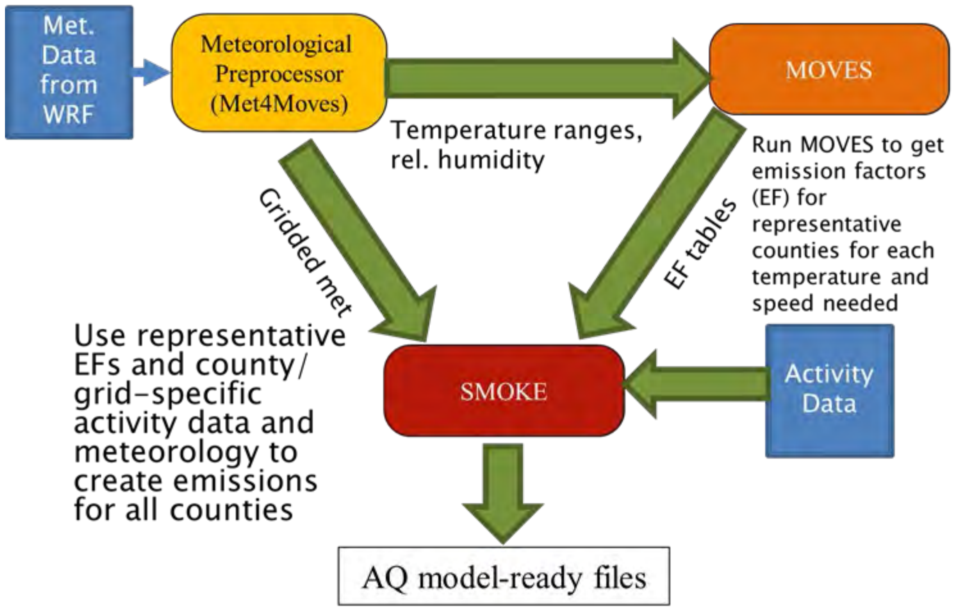


Figure 5-12: Modeling process of onroad emissions as part of the input for air quality modeling

5.4.1 Control Scenario Evaluated for the Air Quality Modeling Analysis

The control scenario evaluated for the air quality modeling analysis is different than the final standards that are represented in the national emissions inventories discussed in Chapter 5.3. Table 5-24 through Table 5-27 present the differences in duty-cycle NO_X standards, warranty, and useful life between the control scenario modeled for air quality modeling and the final standards.

Table 5-24 compares the differences in the duty-cycle standards used in developing the running, start, and extended idle emission rates in the final standards and the scenario used for the air quality modeling analysis.

Table 5-24: Duty-Cycle NO_X Standards for the Final Standards and the Control Scenario Analyzed for Air Quality Modeling

Model Year	Engine	Duty Cycle	Duty-Cycle NO_X Standards (mg/hp-hr)	
			Scenario Analyzed for Air Quality Modeling	Final Standards
2027	HHD, MHD, LHD	FTP	100	35 [50][B]
		SET	50	35 [50][B]
		LLC	200	50 [65][B]
		Idle[C]	18 g/hr	5 g/hr
	HD SI	FTP	100	35
		SET	50	35
2030[A]	HHD, MHD, LHD	FTP	50	35 [50][B]
		SET	20	35 [50][B]
		LLC	100	50 [65][B]
		Idle[C]	10 g/hr	5 g/hr
	HD SI	FTP	50	35
		SET	20	35

[A] The different duty-cycle NOx standards for MY2030 apply only to the air quality modeling control scenario. The final standards have the same duty-cycle NOx standards for MY2027 and later.

[B] Values in brackets denote the 15 mg/hp-hr compliance margin for MHDE and HHDE that applies after the engines are in-use in the final rule (see preamble Section III.B for details).

[C] In both scenarios, we assumed compliance with the Voluntary Idle standard which is more stringent than the off-cycle standard as discussed in Chapter 5.2.2.1.1.2. Note that the Voluntary Idle standard in the final control scenario that we modeled is different than the Voluntary Idle standard in the final program, see preamble Sections III.B for details on the Voluntary Idle standard in the final program.

In the control scenario analyzed for air quality modeling, the FTP and SET standards are different from one another. The R_{duty} values calculated from the FTP are applied to MOVES running operating modes for vehicle speeds below 50 miles per hour, which aligns with the transient behavior of the FTP cycle. The R_{duty} from the SET standard are applied to MOVES operating modes above 50 mph (operating mode 33 and above), which aligns with the high-speed activity that is targeted with the SET standard. However, the final FTP and SET standards are equivalent, so we used the same R_{duty} values to calculate the emission rates for all running operating modes, as discussed in Section 5.2.2.1.1.1.

Table 5-25: R_{duty} Ratios Calculated for the Control Scenario Analyzed for Air Quality Modeling

Scenario	Applicable Model Years	Emission standard (g/hp-hr)		R_{duty}	
		FTP	SET-RMC	FTP	SET-RMC
Air Quality Modeling	Model Year 2027-2029	0.1	0.05	50%	25%
	Model Year 2030+	0.05	0.02	25%	10%
Final Standards	Model Year 2027+	0.035 [0.05][A]	0.035 [0.05][A]	17.5% [25%][A]	17.5% [25%][A]

[A] Values in brackets denote the 15 mg/hp-hr compliance margin for MHDE and HHDE that applies after the engines are in-use in the final rule (see preamble Section III.B for details).

The differences in the warranty and useful life periods analyzed for air quality modeling are shown in Table 5-26 and Table 5-27.

Table 5-26: Warranty Mileages and Years in the Final Control Scenario and the Control Scenario Analyzed for Air Quality Modeling

Model Year	Engine	Warranty Mileage		Warranty Years	
		Air Quality Modeling Control Scenario	Final Standards	Air Quality Modeling Control Scenario	Final Standards
2027	HHD	350k	450k	5 years	7 years[B]
	MHD	150k	280k		10 years
	LHD	110k	210k		10 years
	HD SI	110k	160k	5 years	10 years
2030[A]	HHD	600k	Same as 2027	7 years	Same as 2027
	MHD	260k	Same as 2027		Same as 2027
	LHD	200k	Same as 2027		Same as 2027
	HD SI	160k	Same as 2027	7 years	Same as 2027

[A] The different warranty for MY2030 applies only to the air quality modeling control scenario. The warranty for final standards is the same for MY2027 and later.
[B] The FRM scenario we analyzed included a warranty years value of 7 years instead of 10 years in the final standards scenario for HHD diesel.

Table 5-27:Useful Life Mileages and Years in the Final Control Scenario and the Control Scenario Analyzed for Air Quality Modeling

Model Year	Engine	Useful Life Mileage		Useful Life Years	
		Air Quality Modeling Control Scenario	Final Standards	Air Quality Modeling Control Scenario	Final Standards
2027	HHD	650k	650k	10 years	11 years
	MHD	325k	350k		12 years
	LHD	250k	270k		15 years
	HD SI	200k	200k	10 years	15 years
2030[A]	HHD	850k	Same as 2027	10 years	Same as 2027
	MHD	450k	Same as 2027		Same as 2027
	LHD	350k	Same as 2027		Same as 2027
	HD SI	200k	Same as 2027	10 years	Same as 2027

[A] The different useful life for MY2030 applies only to the air quality modeling control scenario. The useful life for final standards is the same for MY2027 and later.

5.4.2 Estimated Differences in the Emission Reductions between the Final Control Scenario and the Control Scenario Analyzed for Air Quality Modeling

In addition to the differences between the final control scenario and the scenario modeled in the air quality analysis, we have used an updated version of MOVES to develop the emission inventories for the final rule, as described in Chapter 5.2. The combined net impact of the differences in the control scenarios and the differences in the emission inventory methodology are presented in this Chapter 5.4.2.

Overall, the estimated reductions from the final rule compare well with the reductions from the SMOKE-MOVES inventory used in the air quality modeling, despite the differences in modeling approaches (Chapter 5.2) and the control scenarios between the proposal and the final rule (Chapter 5.4.1). Table 5-28 shows that both scenarios estimate large reductions in NO_X, similar reductions in $PM_{2.5}$, and meaningful reductions in VOC, CO, and toxics. Based on this comparison and the findings from the air quality modeling done for the proposal, we conclude the final rule will lead to improvements in air quality.

Table 5-28: Comparison of the Onroad Vehicle Emission Reductions from the Air Quality Modeling Control Scenario vs. the Final Control Scenario

Pollutant	CY2045 Reduction in SMOKE-MOVES Inventory (50 states) Used in the Air Quality Modeling		CY2045 Reduction in MOVES National Inventory (50 states) from the Final Control Scenario	
	US Short Tons	% Reduction	US Short Tons	% Reduction
NO_X	449,408	48.0%	453,239	43.3%
VOC	7,854	1.7%	20,758	3.7%
$PM_{2.5}$ - Primary	548	1.4%	566	1.3%
Carbon Monoxide (CO)	167,241	3.6%	260,750	3.8%
Acetaldehyde	35	0.9%	145	3.3%
Benzene	113	1.6%	221	2.7%
Formaldehyde	46	1.6%	134	4.2%
Naphthalene	4	1.5%	9	2.7%

5.5 Chapter 5 Appendix

5.5.1 Zero-Mile Emission Rates for the Control Scenarios

The zero-mile NO_X emission rates for HHD diesel vehicles in the final standards and proposed Option 2 scenarios due to the duty-cycle and off-cycle standards are displayed in Figure 5-5.

Figure 5-13 and Figure 5-14 display the zero-mile NO_X emission rates for LHD45 and MHD diesel vehicles in the final standards and proposed Option 2 scenarios.

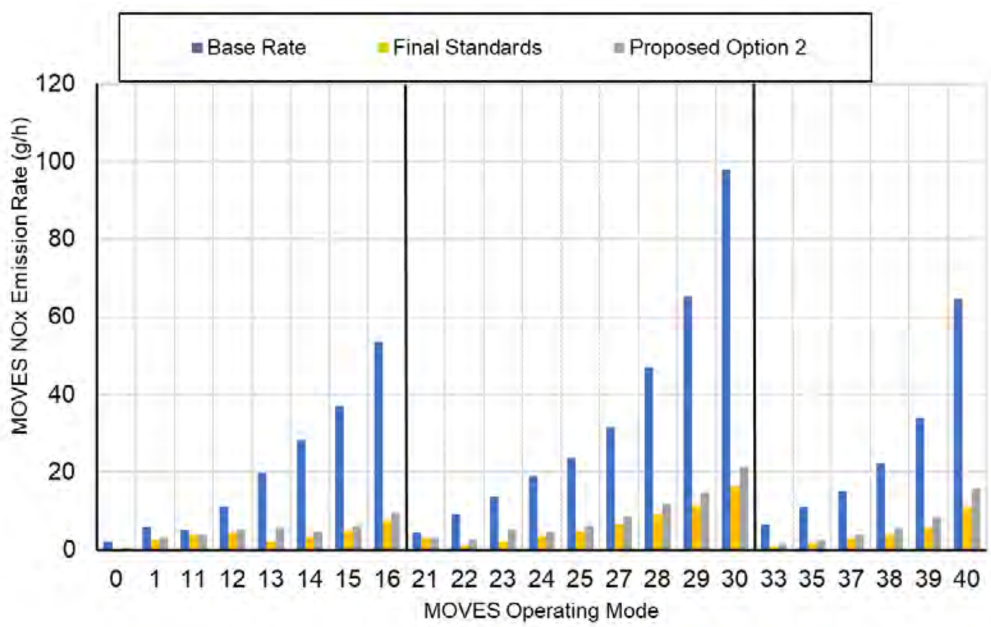


Figure 5-13: Estimated zero-mile emission rates for LHD45 diesel vehicles due to the final and proposed Option 2 duty-cycle and off-cycle standards

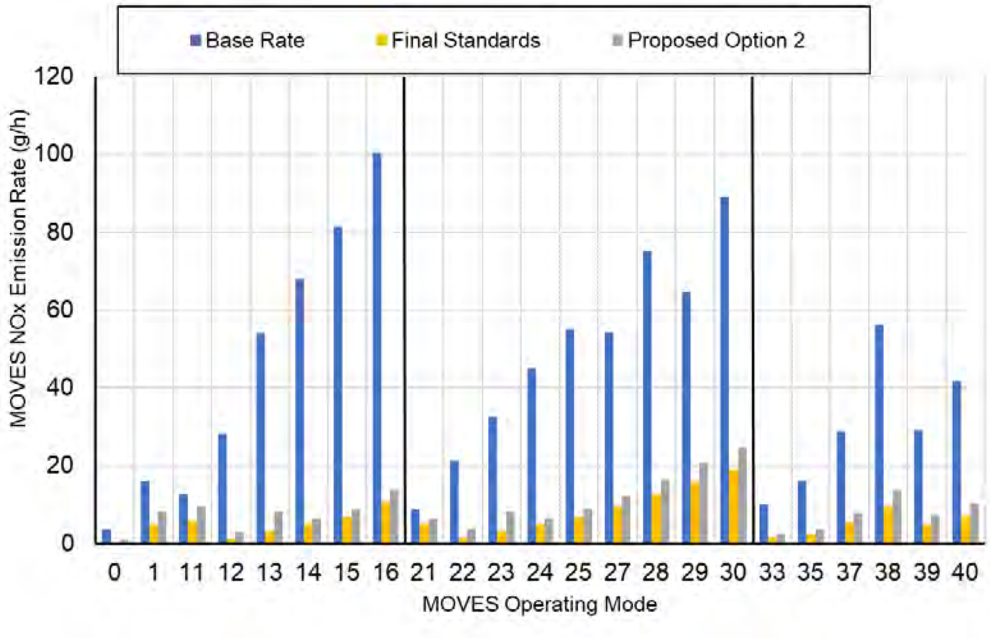


Figure 5-14: Estimated zero-mile emission rates for MHD diesel vehicles due to the final and proposed Option 2 duty-cycle and off-cycle standards

5.5.2 Details of the Emission Impacts of the Final Standards

In this section, we provide details of the national emission reductions from the heavy-duty vehicles due to the final standards (previously summarized in Chapter 5.3.1).

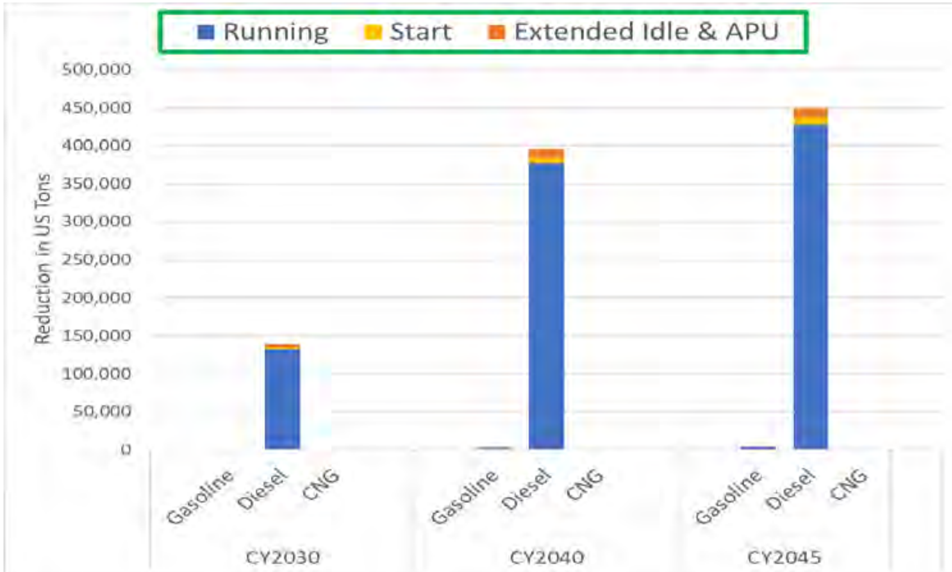


Figure 5-15: National NO_X Emission Reductions from Heavy-duty Vehicles in Calendar Years 2030, 2040, and 2045 ─ for Each Fuel Type Category by Emission Process

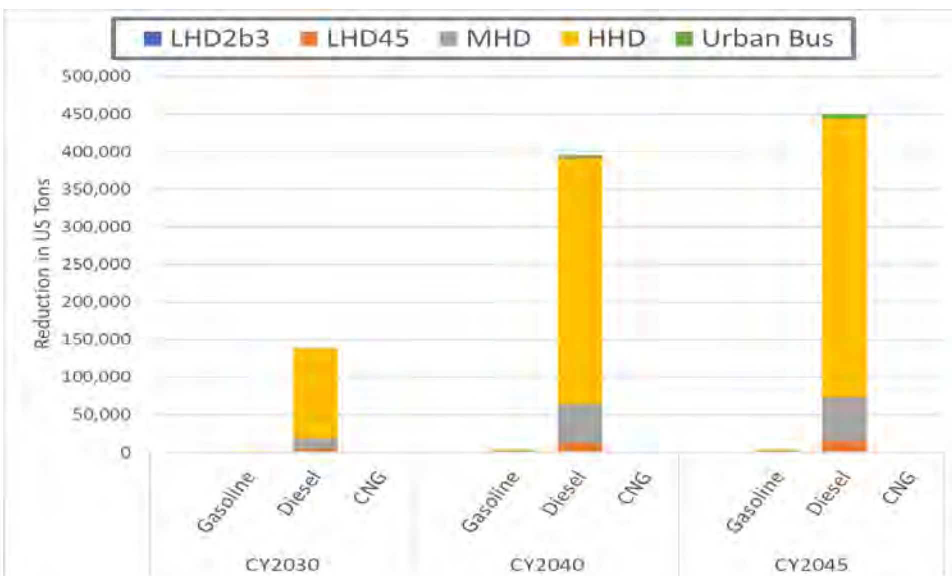


Figure 5-16: National NO_X Emission Reductions from Heavy-duty Vehicles in Calendar Years 2030, 2040, and 2045 ─ for Each Fuel Type Category by HD Regulatory Class

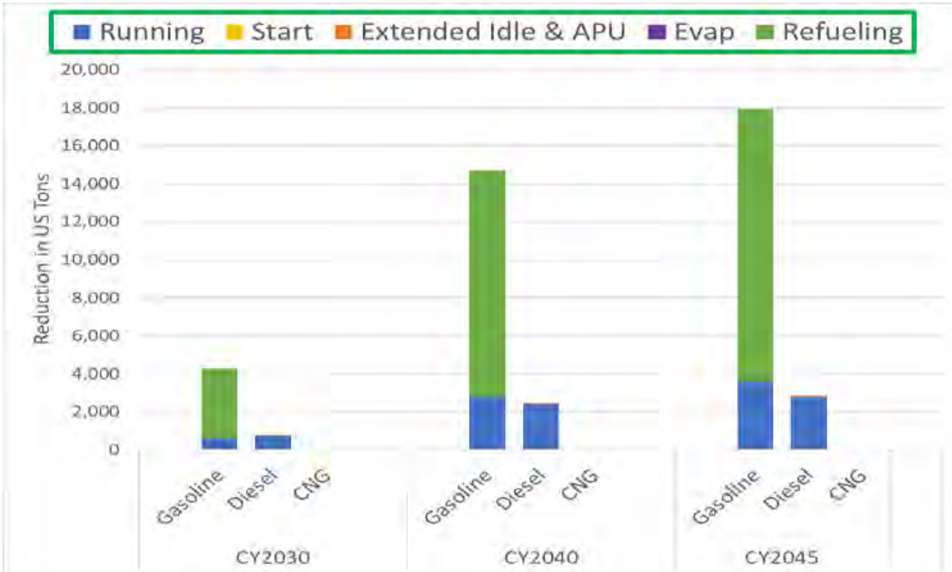


Figure 5-17: National VOC Emission Reductions from Heavy-duty Vehicles in Calendar Years 2030, 2040, and 2045 ─ for Each Fuel Type Category by Emission Process

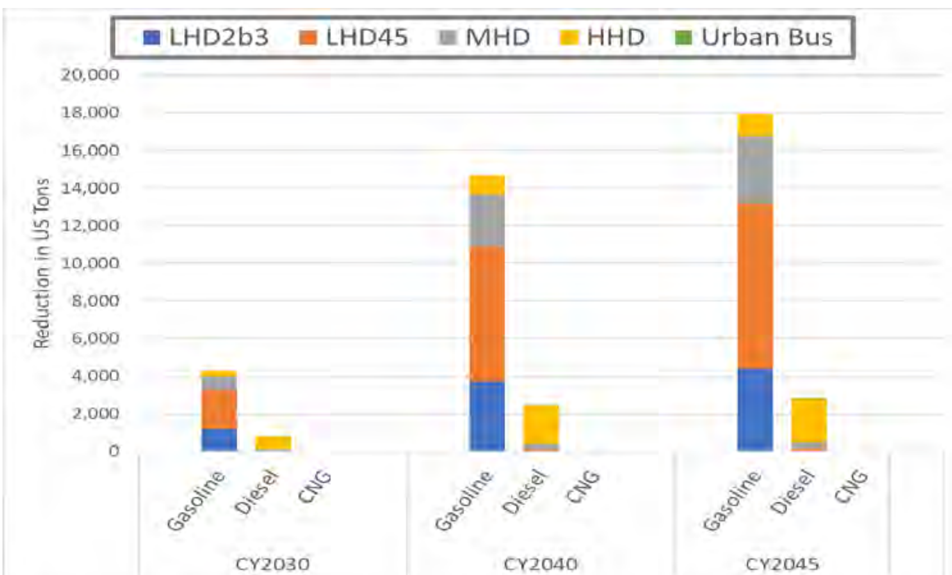


Figure 5-18: National VOC Emission Reductions from Heavy-duty Vehicles in Calendar Years 2030, 2040, and 2045 ─ for Each Fuel Type Category by HD Regulatory Class

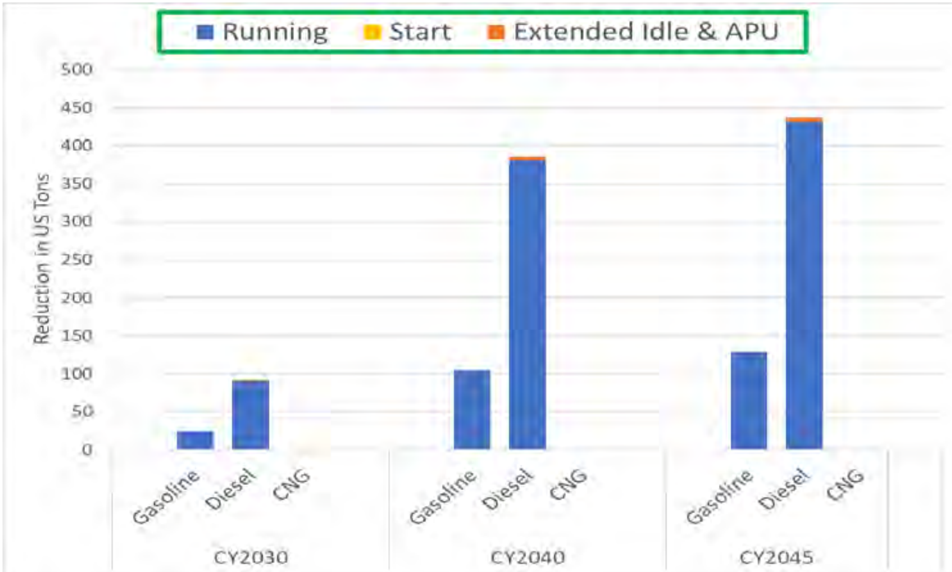


Figure 5-19: National Exhaust $PM_{2.5}$ Emission Reductions from Heavy-duty Vehicles in Calendar Years 2030, 2040, and 2045 – for Each Fuel Type Category by Emission Process

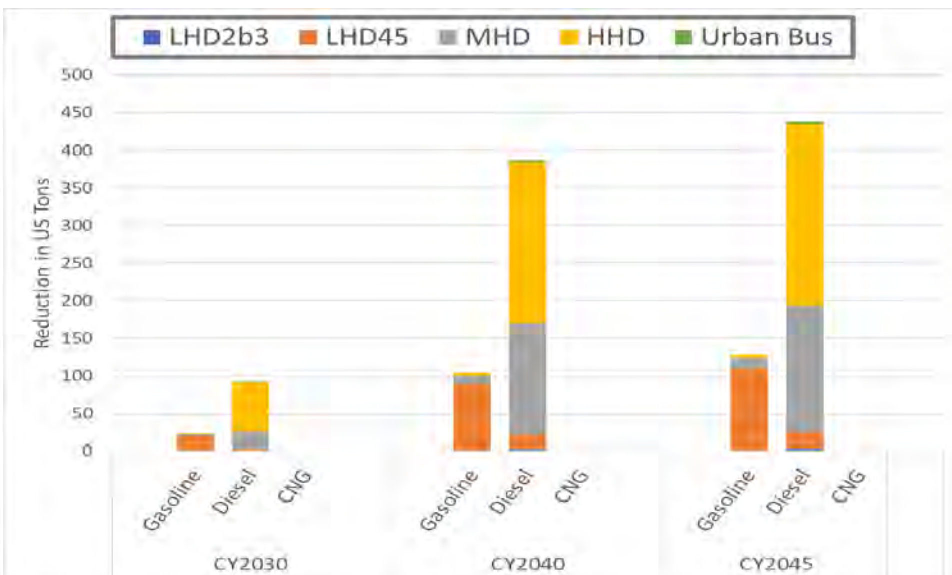


Figure 5-20: National Exhaust $PM_{2.5}$ Emission Reductions from Heavy-duty Vehicles in Calendar Years 2030, 2040, and 2045 – for Each Fuel Type Category by HD Regulatory Class

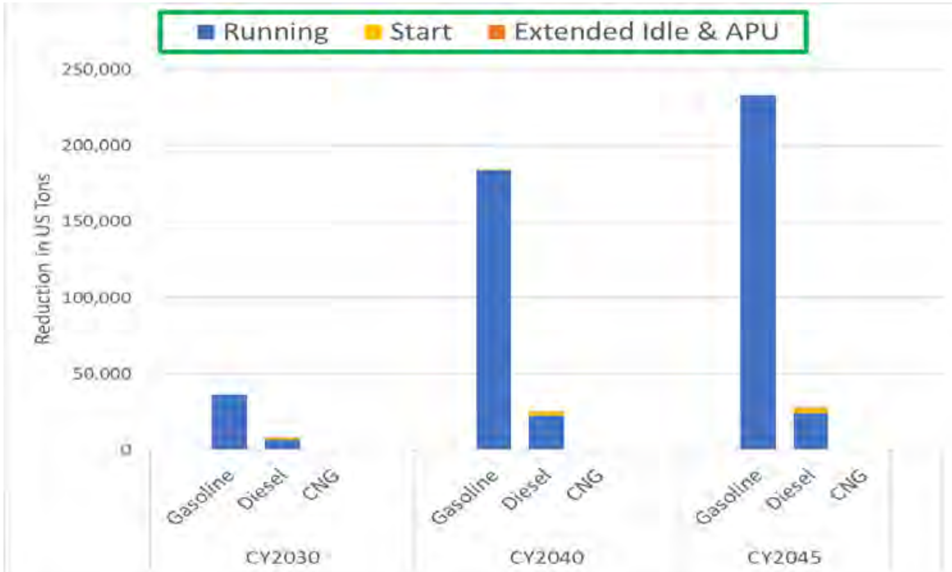


Figure 5-21: National CO Emission Reductions from Heavy-duty Vehicles in Calendar Years 2030, 2040, and 2045 ─ for Each Fuel Type Category by Emission Process

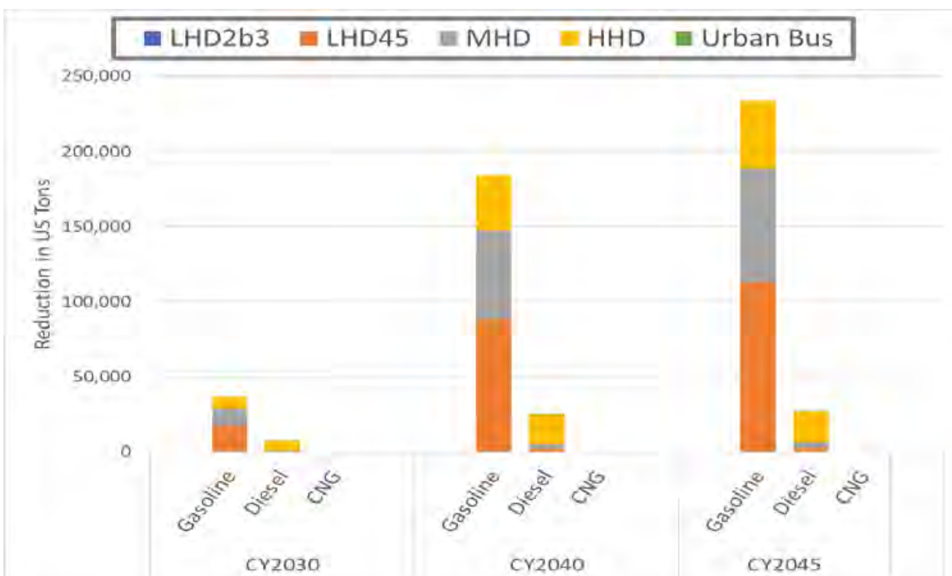


Figure 5-22: National CO Emission Reductions from Heavy-duty Vehicles in Calendar Years 2030, 2040, and 2045 ─ for Each Fuel Type Category by HD Regulatory Class

5.5.3 <u>Onroad Heavy-Duty NO_X Emissions by Engine Operational Process for the Baseline, Final and Proposed Option 2 Standards</u>

Figure 5-23 displays the estimated national onroad heavy-duty NO_X emissions in 2045 from the baseline, final, and proposed Option 2 standards by engine operation process for the MY 2027 and later fleet impacted by the rule. See Section VI of the preamble for more discussion on these comparisons.

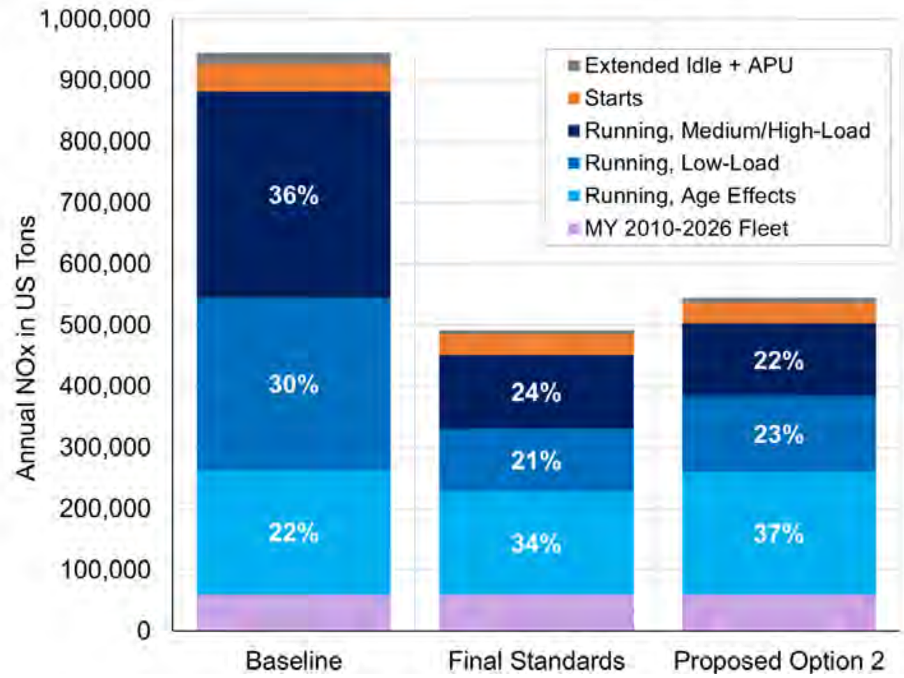


Figure 5-23: Comparison of Calendar Year 2045 Onroad Heavy-Duty NO_X Emissions from Different Engine Operational Process[EE] for the Baseline, Final Standards, and Proposed Option 2 Scenarios

5.5.4 Year-Over-Year Criteria Pollutant Emissions for Calendar Years Between 2027 and 2045

In this section, we present MOVES national inventory emissions (for selected criteria pollutants) across multiple calendar years (2027-2045) for baseline and control scenarios.

The national heavy-duty vehicle emissions inventories are summarized in Table 5-29 through Table 5-32 below for NO_X, VOC, $PM_{2.5}$ (exhaust), and CO, respectively, for the baseline, final standards and proposed Option 2 scenarios. The same results are also displayed graphically in Figure 5-24 through Figure 5-27.

[EE] In this graph, the "low-load running" emissions refer to the running exhaust emissions (for model year 2027 and later without age effects) from MOVES running operating mode 0, 1, 11-14, 21-24, 33, plus 50% of operating mode 35 (see Table 5-3 for MOVES operating mode definitions). The remainder are considered "medium/high-load running" emissions. The contribution of the "Running, Age Effects" was estimated by conducting a series of MOVES runs with and without running aging effects for the baseline, final standards, and proposed Option 2 scenarios. The MOVES inputs without the aging effects are available in the rulemaking docket [3].

Table 5-29: National Heavy-duty Vehicle NO_X Emissions (Annual US Tons) For Calendar Years Between 2027 and 2045

Calendar Year	Baseline	Final Standards	Proposed Option 2
2027	1,102,102	1,068,381	1,069,823
2028	1,059,062	990,842	993,761
2029	1,026,615	923,239	927,661
2030	996,202	856,525	862,503
2031	973,284	788,542	802,285
2032	954,582	726,649	747,755
2033	935,034	678,404	704,463
2034	925,914	641,732	672,513
2035	919,717	611,741	646,165
2036	917,205	587,210	625,054
2037	912,217	562,616	602,916
2038	914,585	546,719	589,342
2039	916,613	532,510	577,147
2040	916,684	517,820	564,216
2041	920,924	509,044	556,976
2042	925,534	502,047	551,409
2043	930,833	497,358	548,009
2044	937,395	494,049	545,975
2045	945,323	492,084	545,299

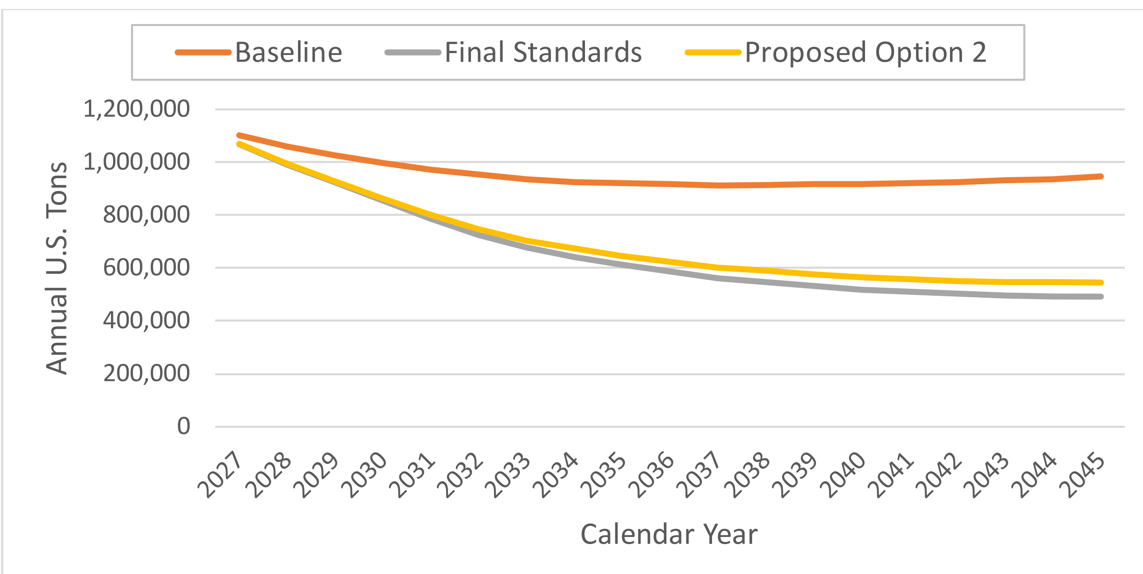


Figure 5-24: National Heavy-duty Vehicle NO_X Emissions (Annual US Tons) For Calendar Years Between 2027 and 2045

Table 5-30: National Heavy-Duty Vehicle VOC Emissions (Annual US Tons) For Calendar Years Between 2027 and 2045

Calendar Year	Baseline	Final Standards	Proposed Option 2
2027	119,297	118,094	118,094
2028	111,657	109,219	109,219
2029	108,273	104,569	104,569
2030	102,788	97,770	97,770
2031	99,012	92,632	92,652
2032	95,537	87,815	87,855
2033	92,868	83,728	83,777
2034	90,841	80,325	80,382
2035	88,878	77,076	77,138
2036	88,593	75,572	75,639
2037	86,610	72,446	72,515
2038	87,251	72,013	72,083
2039	87,826	71,607	71,678
2040	87,657	70,519	70,590
2041	88,448	70,481	70,554
2042	89,209	70,489	70,563
2043	89,964	70,559	70,633
2044	90,777	70,694	70,770
2045	91,810	71,053	71,130

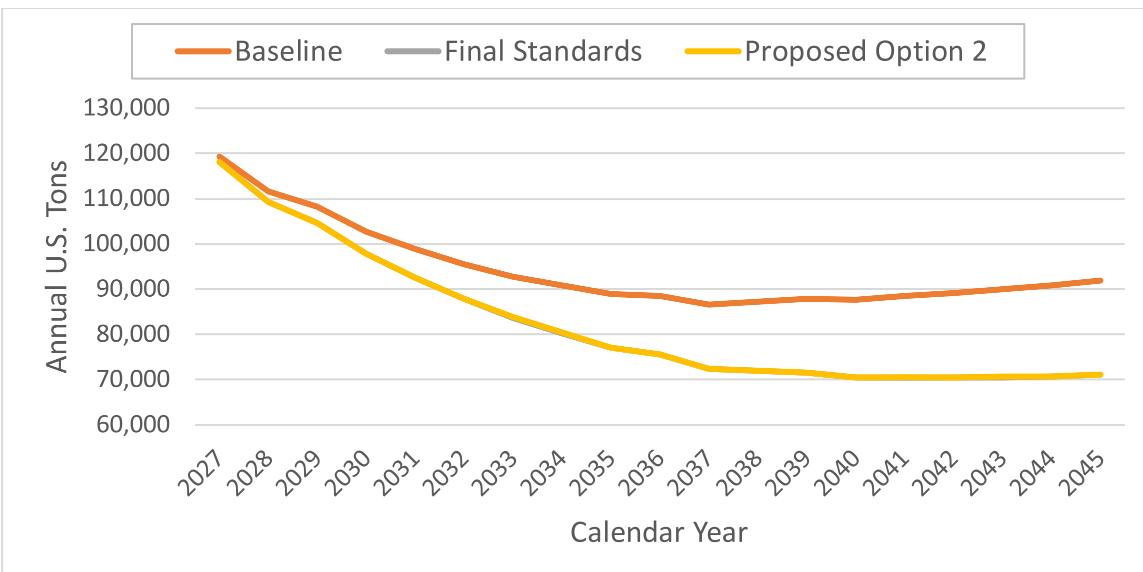


Figure 5-25: National Heavy-Duty Vehicle VOC Emissions (Annual US Tons) For Calendar Years Between 2027 and 2045

Table 5-31: National Heavy-duty Vehicle $PM_{2.5}$ (Exhaust Only) Emissions (Annual US Tons) For Calendar Years Between 2027 and 2045

Calendar Year	Baseline	Final Standards	Proposed Option 2
2027	17,790	17,762	17,762
2028	15,385	15,328	15,328
2029	14,101	14,015	14,015
2030	12,665	12,550	12,550
2031	11,627	11,446	11,454
2032	10,683	10,438	10,454
2033	9,889	9,597	9,624
2034	9,302	8,965	9,001
2035	8,795	8,426	8,466
2036	8,442	8,041	8,085
2037	7,544	7,118	7,163
2038	7,522	7,072	7,118
2039	7,495	7,024	7,070
2040	7,410	6,919	6,967
2041	7,390	6,882	6,930
2042	7,369	6,845	6,894
2043	7,356	6,818	6,868
2044	7,348	6,796	6,846
2045	7,357	6,791	6,842

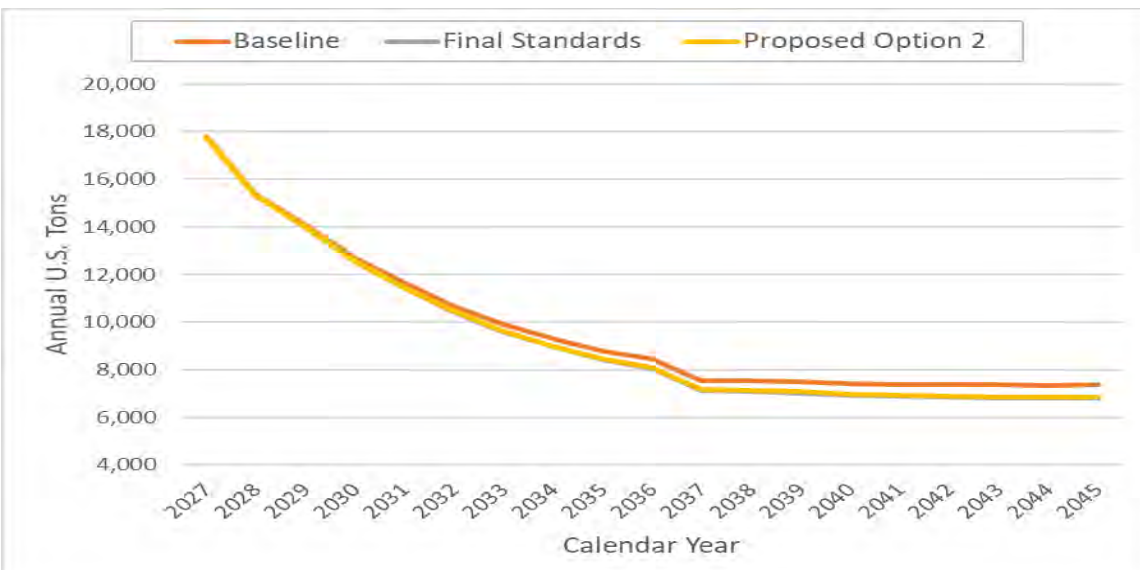


Figure 5-26: National Heavy-duty Vehicle $PM_{2.5}$ (Exhaust Only) Emissions (Annual US Tons) For Calendar Years Between 2027 and 2045

Table 5-32: National Heavy-Duty Vehicle CO Emissions (Annual US Tons) For Calendar Years Between 2027 and 2045

Calendar Year	Baseline	Final Standards	Proposed Option 2
2027	1,626,057	1,615,472	1,615,472
2028	1,538,135	1,516,707	1,516,707
2029	1,510,201	1,477,697	1,477,697
2030	1,469,592	1,425,614	1,425,614
2031	1,436,598	1,377,778	1,379,276
2032	1,403,726	1,330,388	1,333,299
2033	1,388,211	1,294,176	1,297,553
2034	1,370,037	1,255,992	1,259,824
2035	1,355,645	1,223,161	1,227,301
2036	1,356,898	1,206,858	1,211,295
2037	1,344,840	1,178,441	1,182,966
2038	1,346,192	1,164,406	1,169,030
2039	1,351,367	1,155,598	1,160,299
2040	1,347,716	1,138,782	1,143,539
2041	1,364,985	1,144,328	1,149,136
2042	1,378,756	1,147,320	1,152,196
2043	1,392,564	1,151,108	1,156,056
2044	1,407,837	1,156,631	1,161,649
2045	1,426,370	1,165,620	1,170,717

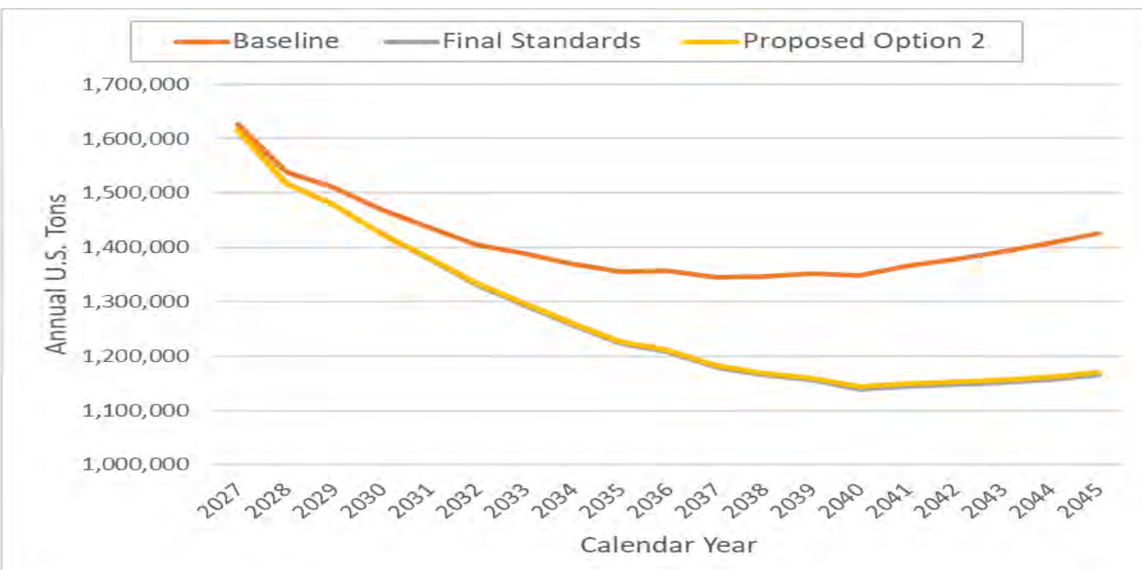


Figure 5-27: National Heavy-Duty Vehicle CO Emissions (Annual US Tons) For Calendar Years Between 2027 and 2045

5.5.5 Sensitivity Analysis of Emissions Impacts of 2026 Service Class Pull Ahead Credits Pathway

This section presents a sensitivity analysis of the estimated emission inventory impacts from one of the transitional credit pathways under the ABT program included in the final rule. As described in preamble Section IV.G.7, we are finalizing a transitional credit program that includes several pathways for manufacturers to generate transitional credits in MYs 2022 through 2026 that they can then use in MYs 2027 and later. We conducted the sensitivity analysis presented in this Appendix to evaluate the potential for additional emissions reductions, particularly in the early years of the program, from allowing manufacturers to generate transitional credits. We focused on the transitional credit pathway that provides the most flexibility to use credits in MYs 2027 and later in order to assess whether these additional flexibilities might impact the expected, additional early emissions reductions from the transitional credits.

The results of the sensitivity analysis presented in this Appendix show that, compared to the emissions reductions expected from the final rule, allowing manufacturers to generate transitional credits through the pathway selected for analysis (i.e., the "2026 Service Class Pull Ahead Credits Pathway") would result in additional emissions reductions in the early years of the program. In the later years of the program, the emissions reductions would be essentially the same with or without this transitional credits pathway. Below, we describe the methods used to analyze the three scenarios included in the sensitivity analysis, and then present detailed results of the comparison.

5.5.5.1 Modeling Scenario and MOVES Inputs

We estimated the emission impacts of the 2026 Service Class Pull Ahead Credits Pathway using the same version of MOVES (MOVES3) as the final rule. The MOVES inputs for the final rule are described in Chapter 5.2.2. The MOVES inputs for the 2026 Service Class Pull Ahead Credits Pathway are based on the duty-cycle test standards, warranty and useful life requirements described in preamble Section IV.G.7 and outlined immediately below.

For the purposes of this sensitivity analysis, we assumed that all heavy heavy-duty engines produced in MY 2026 participated in the 2026 Service Class Pull Ahead Credits Pathway. As such, we assumed that manufacturers certified all MY 2026 heavy-duty engines to FEL of 50 mg/hp-hr or less and met all other EPA requirements for MYs 2027 and later. Table 5-33 through Table 5-36 present our specific modeling inputs. We then assumed that manufacturers used the credits generated by MY2026 engines to produce both heavy-heavy and medium-heavy-duty engines at a FEL of 50 mg/hp-hr in MYs 2027 and later until the credits ran out; based on our analysis, credits generated by heavy heavy-duty engines in MY 2026 are estimated to run out approximately eight years later (i.e., in MY 2034).

Table 5-33: Duty-Cycle NO_X Standards for the 2026 Service Class Pull Ahead Credits Pathway and Final Program[A]

Model Year	Engine	Duty Cycle	2026 Service Class Pull Ahead Credits Pathway[E]	Final Program
2026	HHD, MHD	FTP	50 [65][D]	(200)
		SET	50 [65][D]	(200)
		LLC	71 [86][D]	-
		Idle[C]	7 g/hr	-
	HD SI	FTP	-	(200)
		SET	-	-
2027 and later[B]	HHD, MHD, LHD	FTP	50 [65][D]	35
		SET	50 [65][D]	35 [50][D]
		LLC	71 [86][D]	50 [65][D]
		Idle[C]	7 g/hr	5 g/hr
	HD SI	FTP	-	35
		SET	-	35

[A] (#) = final standards with no change in the noted model year
[B] The 2026 Service Class Pull Ahead Credits program allows credits generated in MY 2026 to be used through MY 2034; thus, standards in these rows apply through MY 2034 for the 2026 Service Class Pull Ahead Credits Pathway
[C] We assumed compliance with the voluntary idle standard; note that the voluntary idle standard that we modeled is different than the voluntary idle standard in the final program, see preamble Section III.B for details on the voluntary idle standard in the final program.
[D] [HHDE in-use compliance margin]. Note that in the final program, the in-use compliance margin applies to both HHDE and MHDE (see preamble Section III.B for details); for this sensitivity analysis, we modeled the compliance margin applying only to HHDE.
[E] For this sensitivity analysis, we modeled only HHDE certifying to the requirements in the 2026 Service Class Pull Ahead Credits Pathway in MY 2026; as discussed in preamble Section IV.G.7, only HHDE and MHDE can certify to requirements of the 2026 Service Class Pull Ahead Credits Pathway and generate credits through this pathway. Our analysis included both HHDE and MHDE using credits in MYs 2027 and later.

Table 5-34: Off-Cycle Standards in 2026 Service Class Pull Ahead Credits Pathway and Final Program[A,C]

Scenario	Model Year	Regulatory Class	Engine Cycle	Off-cycle Bin	Off-Cycle NO_X Standards (g/hr for idling, g/hp-hr for low-load and medium to high-load)
Final Standards	2027+	LHD, MHD	Idle (g/hr)[B]	Idle, < 6% power	5
			LLC (g/hp-hr)	Low-load, 6-20% power	0.058
			FTP & SET (g/hp-hr)	Medium to High Load, >20% power	0.058
		HHD	Idle (g/hr)[B]	Idle, < 6% power	5
			LLC (g/hp-hr)	Low-load, 6-20% power	0.088
			FTP & SET (g/hp-hr)	Medium to High Load, >20% power	0.088
2026 Service Class Pull Ahead Credits Pathway[A]	2026-2034	MHD, HHD	Idle (g/hr)[B]	Idle, < 6% power	7
			LLC (g/hp-hr)	Low-load, 6-20% power	0.083 (0.113 HHDE)
			FTP & SET (g/hp-hr)	Medium to High Load, >20% power	0.083 (0.113 HHDE)

[A] For this sensitivity analysis, we modeled only HHDE certifying to the requirements in the 2026 Service Class Pull Ahead Credits Pathway in MY2026; as discussed in preamble Section IV.G.7, only HHDE and MHDE can certify to requirements of the 2026 Service Class Pull Ahead Credits Pathway and generate credits through this pathway. Our analysis included both HHDE and MHDE using credits in MYs 2027 and later.

[B] Note that the voluntary idle standard that we modeled is different than the voluntary idle standard in the final program, see preamble Sections III.B and III.C for details on the voluntary idle standard in the final program and the off-cycle standard for idle emissions, respectively.
[C] Note that we modeled all engine categories complying with off-cycle standards during in-use operations, which for HHD includes an in-use compliance margin. In the final program, the in-use compliance margin applies to both HHDE and MHDE (see preamble Section III.B for details); for this sensitivity analysis, we modeled the compliance margin applying only to HHDE. The modeling for this sensitivity analysis also used a 30 mg/hp-hr compliance margin for HHD off-cycle emissions and a 15 mg/hp-hr compliance margin for duty-cycle emissions; as discussed in preamble Section III.B, the compliance margin for MHDE and HHDE in the final rule is 15 mg/hp-hr for both off-cycle and duty-cycle emissions.

Table 5-35: Warranty Mileages and Years in 2026 Service Class Pull Ahead Credits Pathway and Final Program[A]

<table>
<tr><th rowspan="2">Model Year</th><th rowspan="2">Engine</th><th colspan="2">Warranty Mileage</th><th colspan="2">Warranty Years</th></tr>
<tr><td>2026 Service Class Pull Ahead Credits Pathway[B]</td><td>Final Program</td><td>2026 Service Class Pull Ahead Credits Pathway</td><td>Final Program</td></tr>
<tr><td rowspan="4">2026</td><td>HHD</td><td>450k</td><td>(100k)</td><td>7 y</td><td rowspan="3">(5 y)</td></tr>
<tr><td>MHD</td><td>-</td><td>(100k)</td><td>-</td></tr>
<tr><td>LHD</td><td>-</td><td>(50k)</td><td>-</td></tr>
<tr><td>HD SI</td><td>-</td><td>(50k)</td><td>-</td><td>(5 y)</td></tr>
<tr><td rowspan="4">2027+</td><td>HHD</td><td>450k</td><td>450k</td><td rowspan="3">7 y</td><td rowspan="3">7 y</td></tr>
<tr><td>MHD</td><td>280k</td><td>280k</td></tr>
<tr><td>LHD</td><td>210k</td><td>210k</td></tr>
<tr><td>HD SI</td><td>160k</td><td>160k</td><td>7 y</td><td>7 y</td></tr>
</table>

[A] (#) = final standards with no change in the noted model year
[B] For this sensitivity analysis, we modeled only HHDE certifying to the requirements in the 2026 Service Class Pull Ahead Credits Pathway in MY2026; as discussed in preamble Section IV.G.7, only HHDE and MHDE can certify to requirements of the 2026 Service Class Pull Ahead Credits Pathway and generate credits through this pathway. Our analysis included both HHDE and MHDE using credits in MYs 2027 and later.

Table 5-36: Useful Life Mileages and Years in 2026 Service Class Pull Ahead Credits Pathway and Final Program [A]

<table>
<tr><th rowspan="2">Model Year</th><th rowspan="2">Engine</th><th colspan="2">Useful Life Mileage</th><th colspan="2">Useful Life Years</th></tr>
<tr><th>2026 Service Class Pull Ahead Credits Pathway[B]</th><th>Final Program</th><th>2026 Service Class Pull Ahead Credits Pathway</th><th>Final Program</th></tr>
<tr><td rowspan="4">2026</td><td>HHD</td><td>650k</td><td>(435k)</td><td>11</td><td rowspan="3">(10 y)</td></tr>
<tr><td>MHD</td><td>-</td><td>(185k)</td><td>-</td></tr>
<tr><td>LHD</td><td>-</td><td>(110k)</td><td>-</td></tr>
<tr><td>HD SI</td><td>-</td><td>(110k)</td><td>-</td><td>(10 y)</td></tr>
<tr><td rowspan="4">2027</td><td>HHD</td><td>650k</td><td>650k</td><td>11 y</td><td>11 y</td></tr>
<tr><td>MHD</td><td>350k</td><td>350k</td><td>12 y</td><td>12 y</td></tr>
<tr><td>LHD</td><td>270k</td><td>270k</td><td>15 y</td><td>15 y</td></tr>
<tr><td>HD SI</td><td>200k</td><td>200k</td><td>15 y</td><td>15 y</td></tr>
</table>

[A] (#) = final standards with no change in the noted model year
[B] For this sensitivity analysis, we modeled only HHDE certifying to the requirements in the 2026 Service Class Pull Ahead Credits Pathway in MY2026; as discussed in preamble Section IV.G.7, only HHDE and MHDE can certify to requirements of the 2026 Service Class Pull Ahead Credits Pathway and generate credits through this pathway. Our analysis included both HHDE and MHDE using credits in MYs 2027 and later.

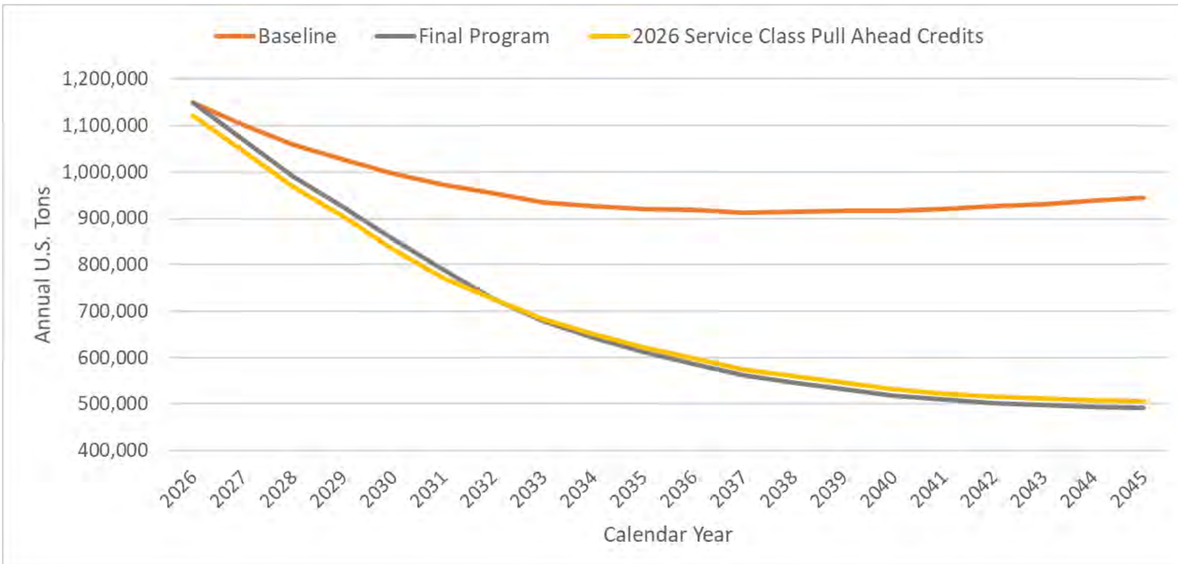
5.5.5.2 NO_X Emissions Inventory Impacts of 2026 Service Class Pull Ahead Credits

The results of our sensitivity analysis, with the assumptions described in 5.5.5.1, are shown in Figure 5-28. Our data show that including the 2026 Service Class Pull Ahead Credits pathway in the final rule provides approximately 2 percent greater emissions reductions in CYs 2026 through 2031 compared to the final rule without the 2026 Service Class Pull Ahead Credits pathway. In CYs 2032 through 2045, the emissions reductions from the final rule with the 2026 Service Class Pull Ahead Credits pathway in place are comparable to the final rule without the transitional credits.

As described in 5.5.5.1, we assumed all heavy heavy-duty engines certified in MY 2026 would participate in the 2026 Service Class Pull Ahead Credits pathway, which likely overestimates the volume of credits that would be generated and hence the magnitude of additional emissions reductions in the early years of the program. However, our modeling did not include the 10 percent discount that is part of the final pathway's requirements to move credits between heavy heavy-duty engines and medium heavy-duty engine averaging sets; the 10 percent discount ensures that there will be a reduction of the overall emission level from generating and using credits that are transferred between averaging sets. In addition, we assumed that all credits generated in MY 2026 would be used; however, as the heavy-duty fleet continues to transition to ZEVs, it is possible that manufacturers may not use all the credits generated, which would result in greater emissions reductions than shown in our analysis.

As noted in 5.5.5.1, our sensitivity analysis represents one of several transitional credit pathways in the final rule. However, since the transitional credit pathway we analyzed includes the most flexibilities for using credits in MYs 2027 and later, we believe the results are indicative of the types of additional, early emissions reductions that the other transitional credit pathways could provide.

Figure 5-28: Additional NO_X Emissions Inventory Reductions in Early Program Years from 2026 Service Class Pull Ahead Credits Program Compared to Final Program



Chapter 5 References

[1] USEPA (2020), MOVES3: Latest Version of Motor Vehicle Emission Simulator. https://www.epa.gov/moves/latest-version-motor-vehicle-emission-simulator-moves

[2] USEPA (2020). Science Inventory: Onroad Emission Rate Updates to MOVES3. https://cfpub.epa.gov/si/si_public_record_report.cfm?dirEntryId=347138&Lab=OTAQ&simplesearch=0&showcriteria=2&sortby=pubDate&searchall=moves&timstype=&datebeginpublishedpresented=07/19/2020

[3] Han, Jaehoon. Memorandum to the Docket EPA-HQ-OAR-2019-0055: "MOVES Modeling-Related Data Files (MOVES Code, Input Databases and Runspecs) for the Final Heavy-Duty 2027 Standards." December 2022.

[4] USEPA (2020). Exhaust Emission Rates for Heavy-Duty On-road Vehicles in MOVES3. EPA-420-R-20-018. Assessment and Standards Division. Office of Transportation and Air Quality. US Environmental Protection Agency. Ann Arbor, MI. November 2020. https://www.epa.gov/moves/moves-onroad-technical-reports.

[5] US Energy Information Administration (EIA), Annual Energy Outlook 2019, Washington, DC: January 2020, https://www.eia.gov/outlooks/archive/aeo19/

[6] USEPA (2021). Population and Activity of Onroad Vehicles in MOVES3. EPA-420-R-21-012. Assessment and Standards Division. Office of Transportation and Air Quality. US Environmental Protection Agency. Ann Arbor, MI. April 2021. https://www.epa.gov/moves/moves-onroad-technical-reports.

[7] USEPA (2020). Fuel Effects on Exhaust Emissions from Onroad Vehicles in MOVES3. EPA-420-R-20-016. Assessment and Standards Division. Office of Transportation and Air Quality. US Environmental Protection Agency. Ann Arbor, MI. November 2020. https://www.epa.gov/moves/moves-onroad-technical-reports.

[8] Greenhouse Gas Emissions and Fuel Efficiency Standards for Medium- and Heavy-Duty Engines and Vehicles— Phase 2. 81 FR 73941 (October 25, 2016)

[9] 2007/2010 Heavy-duty rulemaking. 66 FR 5002, January 18, 2001

[10] USEPA (2020). Greenhouse Gas and Energy Consumption Rates for Onroad Vehicles MOVES3. EPA-420-R-20-015. Assessment and Standards Division. Office of Transportation and Air Quality. US Environmental Protection Agency. Ann Arbor, MI. November 2020. https://www.epa.gov/moves/moves-onroad-technical-reports.

[11] 59 FR 16262, April 6, 1994

[12] 65 FR 6698, February 10, 2000.

[13] 79 FR 23414, April 28, 2014 and 80 FR 0978, February 19, 2015.

[14] USEPA (2020). Evaporative Emissions from Onroad Vehicles in MOVES3. EPA-420-R-20-012. Assessment and Standards Division. Office of Transportation and Air Quality. US Environmental Protection Agency. Ann Arbor, MI. November 2020. https://www.epa.gov/moves/moves-onroad-technical-reports.

[15] U.S. EPA (2021) Technical Support Document: Air Quality Modeling for the HD 2027 Proposal.

Chapter 6 Air Quality Impacts

This chapter presents information on air quality, including a discussion of current air quality in Chapter 6.1, a discussion of air quality impacts from the final standards in Chapter 6.2, details related to the methodology used for the proposal air quality modeling analysis in Chapter 6.3, and results from the proposal air quality modeling analysis which are summarized in Chapter 6.4.

6.1 Current Air Quality

In this section we present information related to current levels of air pollutants, visibility levels, and deposition amounts. This provides context for the need for this rule and a comparison for the modeled projections from the rule.

6.1.1 **Ozone**

As described in Chapter 4 of this RIA, ozone causes adverse health effects, and EPA has set national ambient air quality standards (NAAQS) to protect against those health effects. The primary NAAQS for ozone, established in 2015 and retained in 2020, is an 8-hour standard with a level of 0.07 ppm.[A] EPA recently announced that it will reconsider the decision to retain the ozone NAAQS.[B] EPA is also implementing the previous 8-hour ozone primary standard, set in 2008 at a level of 0.075 ppm. As of August 31, 2022, there were 34 ozone nonattainment areas for the 2008 primary ozone NAAQS, composed of 141 full or partial counties, with a population of more than 90 million (see Figure 6-1); there were 49 ozone nonattainment areas for the 2015 primary ozone NAAQS, composed of 212 full or partial counties, with a population of more than 125 million (see Figure 6-2). In total, there were, as of August 31, 2022, 57 ozone nonattainment areas with a population of more than 130 million people.[C]

[A] *https://www.epa.gov/ground-level-ozone-pollution/ozone-national-ambient-air-quality-standards-naaqs*.
[B] *https://www.epa.gov/ground-level-ozone-pollution/epa-reconsider-previous-administrations-decision-retain-2015-ozone*.
[C] The total population is calculated by summing, without double counting, the 2008 and 2015 ozone nonattainment populations contained in the Criteria Pollutant Nonattainment Summary report (https://www.epa.gov/green-book/green-book-data-download).

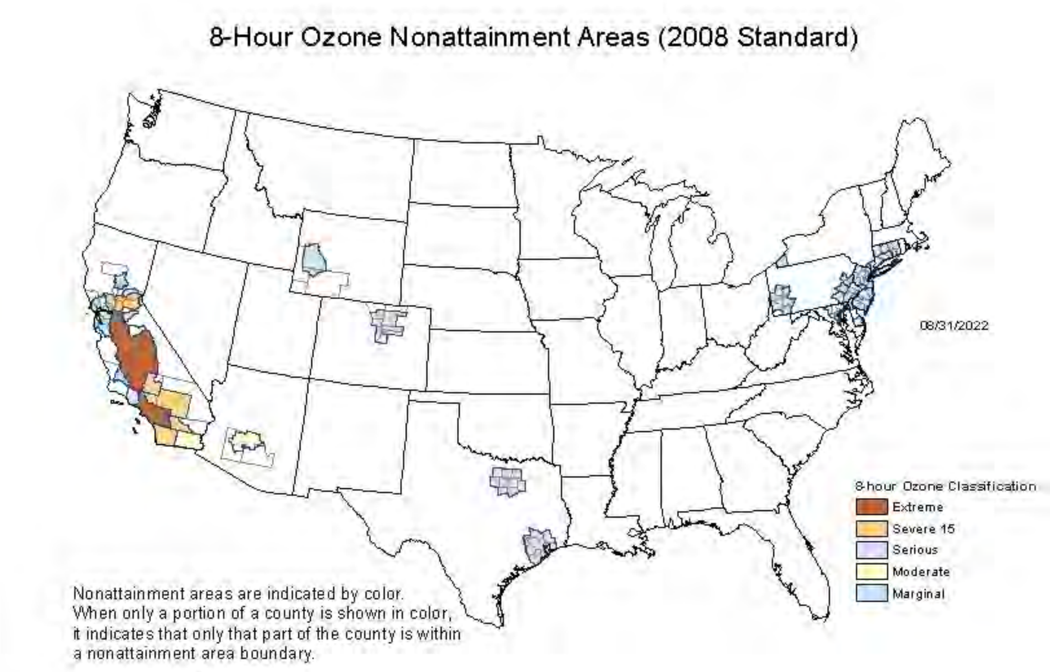


Figure 6-1: 8-Hour Ozone Nonattainment Areas (2008 Standard)

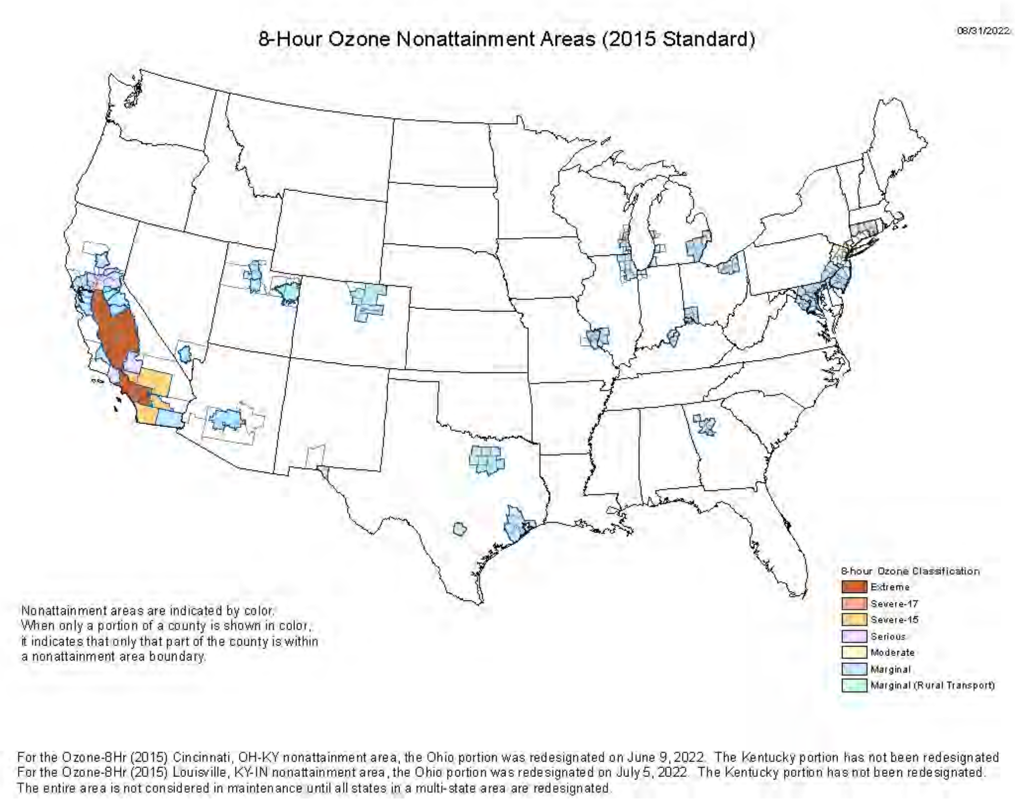


Figure 6-2: 8-Hour Ozone Nonattainment Areas (2015 Standard)

States with ozone nonattainment areas are required to take action to bring those areas into attainment. The attainment date assigned to an ozone nonattainment area is based on the area's classification. The attainment dates for areas designated nonattainment for the 2008 8-hour ozone NAAQS are in the 2015 to 2032 timeframe, depending on the severity of the problem in each area. Attainment dates for areas designated nonattainment for the 2015 ozone NAAQS are in the 2021 to 2038 timeframe, again depending on the severity of the problem in each area.[1] The final standards will begin to take effect in 2027 and will assist areas with attaining the NAAQS and may relieve areas with already stringent local regulations from some of the burden associated with adopting additional local controls.[D] The rule will also provide assistance to counties with ambient concentrations near the level of the NAAQS who are working to ensure long-term attainment or maintenance of the NAAQS.

6.1.2 **$PM_{2.5}$**

As described in Chapter 4 of this RIA, PM causes adverse health effects, and EPA has set NAAQS to protect against those health effects. There are two primary NAAQS for $PM_{2.5}$: an annual standard (12.0 micrograms per cubic meter (μg/m^3)) and a 24-hour standard (35 μg/m^3), and there are two secondary NAAQS for $PM_{2.5}$: an annual standard (15.0 μg/ m^3) and a 24-hour standard (35 μg/m^3). The initial $PM_{2.5}$ standards were set in 1997 and revisions to the standards were finalized in 2006 and in December 2012, and then retained in 2020. On June 10, 2021, EPA announced that it will reconsider the decision to retain the PM NAAQS.[2]

There are many areas of the country that are currently in nonattainment for the annual and 24-hour primary $PM_{2.5}$ NAAQS. As of August 31, 2022, more than 19 million people lived in the 4 areas that are designated as nonattainment for the 1997 annual $PM_{2.5}$ NAAQS. Also, as of August 31, 2022, more than 31 million people lived in the 14 areas that are designated as nonattainment for the 2006 24-hour $PM_{2.5}$ NAAQS, and more than 20 million people lived in the 5 areas designated as nonattainment for the 2012 annual $PM_{2.5}$ NAAQS. In total, there are currently 15 $PM_{2.5}$ nonattainment areas with a population of more than 32 million people.[E] Nonattainment areas for the $PM_{2.5}$ NAAQS are pictured in Figure 6-3.

[D] While not quantified in the air quality modeling analysis for this rule, elements of the Averaging, Banking, and Trading (ABT) program could encourage manufacturers to introduce new emission control technologies prior to the 2027 model year, which may help to accelerate some emission reductions of the final rule (See Preamble Section IV.G for more details on the ABT program in the final rule).

[E] The population total is calculated by summing, without double counting, the 1997, 2006 and 2012 $PM_{2.5}$ nonattainment populations contained in the Criteria Pollutant Nonattainment Summary report (https://www.epa.gov/green-book/green-book-data-download).

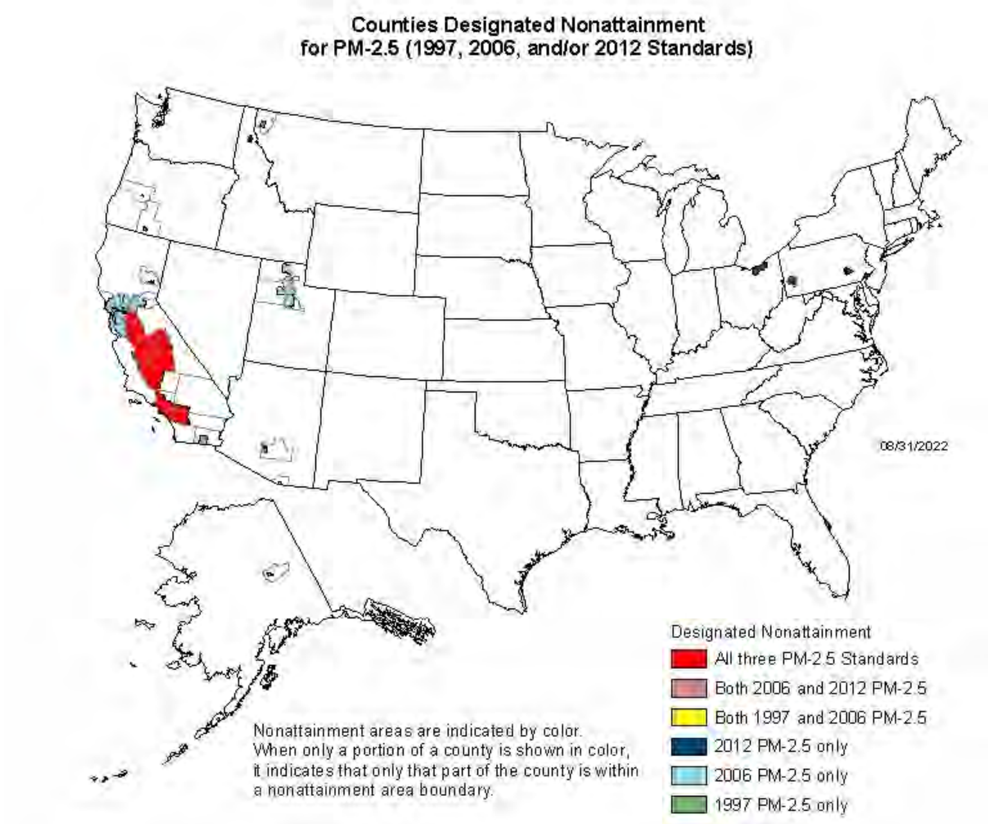


Figure 6-3: Counties Designated Nonattainment for $PM_{2.5}$ (1997, 2006, and/or 2012 standards)

The final standards will take effect in 2027 and will assist areas with attaining the NAAQS and may relieve areas with already stringent local regulations from some of the burden associated with adopting additional local controls.[F] The rule will also provide assistance to counties with ambient concentrations near the level of the NAAQS who are working to ensure long-term attainment or maintenance of the $PM_{2.5}$ NAAQS.

6.1.3 **NO_2**

There are two primary NAAQS for NO_2: an annual standard (53 ppb) and a 1-hour standard (100 ppb).[G] In 2010, EPA established requirements for monitoring NO_2 near roadways expected to have the highest concentrations of NO_2 within large cities. Monitoring within this near-roadway network began in 2014, with additional sites deployed in the following years. At present, there are no nonattainment areas for NO_2.

[F] While not quantified in the air quality modeling analysis for this rule, elements of the Averaging, Banking, and Trading (ABT) program could encourage manufacturers to introduce new emission control technologies prior to the 2027 model year, which may help to accelerate some emission reductions of the final rule (See Preamble Section IV.G for more details on the ABT program in the final rule).

[G] The statistical form of the 1-hour NAAQS for NO_2 is the 3-year average of the yearly distribution of 1-hour daily maximum concentrations.

6.1.4 **CO**

There are two primary NAAQS for CO: an 8-hour standard (9 ppm) and a 1-hour standard (35 ppm). There are currently no CO nonattainment areas; as of September 27, 2010, all CO nonattainment areas had been redesignated to attainment.

6.1.5 **Air Toxics**

The most recent available data indicate that millions of Americans live in areas where air toxics pose potential health concerns.[3] The levels of air toxics to which people are exposed vary depending on where people live and work and the kinds of activities in which they engage, as discussed in detail in EPA's 2007 Mobile Source Air Toxics Rule.[4] According to EPA's Air Toxics Screening Assessment (AirToxScreen) for 2018, mobile sources were responsible for 40 percent of outdoor anthropogenic toxic emissions and were the largest contributor to national average cancer and noncancer risk from directly emitted pollutants.[5,H] Mobile sources are also significant contributors to precursor emissions which react to form air toxics.[6] Formaldehyde is the largest contributor to cancer risk of all 71 pollutants quantitatively assessed in the 2018 AirToxScreen. Mobile sources were responsible for 26 percent of primary anthropogenic emissions of this pollutant in 2018 and are significant contributors to formaldehyde precursor emissions. Benzene is also a large contributor to cancer risk, and mobile sources account for about 60 percent of average exposure to ambient concentrations.

6.1.6 **Visibility**

As of August 31, 2022, over 32 million people live in areas that are designated nonattainment for the $PM_{2.5}$ NAAQS. Overall, the evidence is sufficient to conclude that a causal relationship exists between PM and visibility impairment.[7] Thus, the populations who live in nonattainment areas and travel to these areas will likely be experiencing visibility impairment. Additionally, while visibility trends have improved in Mandatory Class I Federal areas, these areas continue to suffer from visibility impairment.[8,9,I] In summary, visibility impairment is experienced throughout the U.S., in multi-state regions, urban areas, and remote Mandatory Class I Federal areas.

6.1.7 **Deposition**

Over the past two decades, the EPA has undertaken numerous efforts to reduce nitrogen deposition across the U.S. Analyses of monitoring data for the U.S. show that deposition of nitrogen compounds has decreased over the last 25 years. At 34 long-term monitoring sites in the eastern U.S., where data are most abundant, average total nitrogen deposition decreased by 43 percent between 1989-1991 and 2014-2016.[10,11] Although total nitrogen deposition has decreased over time, many areas continue to be negatively impacted by deposition.

[H] AirToxScreen also includes estimates of risk attributable to background concentrations, which includes contributions from long-range transport, persistent air toxics, and natural sources; as well as secondary concentrations, where toxics are formed via secondary formation. Mobile sources substantially contribute to long-range transport and secondarily formed air toxics.

[I] Mandatory Class I Federal areas are the 156 national parks and wilderness areas where state and federal agencies work to improve visibility, https://www.epa.gov/visibility/regional-haze-program.

6.2 Air Quality Impacts of the Final Rule

We expect the standards in the final rule to result in meaningful reductions in emissions of NO_X, VOC, CO and $PM_{2.5}$. When feasible, we conduct full-scale photochemical air quality modeling to accurately project levels of criteria and air toxic pollutants, because the atmospheric chemistry related to ambient concentrations of $PM_{2.5}$, ozone, and air toxics is very complex. Air quality modeling was performed for the proposed rule and demonstrated improvements in concentrations of air pollutants. We did not perform new air quality modeling for this final rule. Chapter 5.4 of the RIA provides additional detail on the emissions inventory used for the proposal's air quality modeling, including a comparison of the emission reductions modeled in the air quality analysis and those from this final rule. Generally, despite the differences in the version of the MOVES model used and the control scenario modeled, the emission reductions used in the air quality modeling analysis compare well to the emission reductions estimated for the final standards. Both scenarios result in reductions in emissions of VOC and $PM_{2.5}$ and large reductions in emissions of NO_X, and we conclude that given the similar structure of the proposed and final programs, the geographic distribution of emissions reductions and modeled improvements in air quality are consistent and demonstrate that the final rule will lead to substantial improvements in air quality.

We expect this rule will decrease ambient concentrations of air pollutants, including significant improvements in ozone concentrations in 2045 as demonstrated in the air quality modeling analysis. We also expect reductions in ambient $PM_{2.5}$, NO_2 and CO due to this rule. Although the spatial resolution of the air quality modeling is not sufficient to quantify it, this rule's emission reductions will also reduce air pollution in close proximity to major roadways, where concentrations of many air pollutants are elevated and where people of color and people with low income are disproportionately exposed. The emission reductions provided by the final standards will be important in helping areas attain the NAAQS and prevent future nonattainment. In addition, the final standards are expected to result in improvements in nitrogen deposition and visibility. Additional information and maps showing modeled changes in ambient concentrations of air pollutants in 2045 from the proposed standards are included in Chapter 6.4 of this RIA and in the Air Quality Modeling Technical Support Document from the proposed rule.[12]

6.3 Air Quality Modeling Methodology for Proposal Analysis

6.3.1 **Air Quality Model**

CMAQ is a non-proprietary computer model that simulates the formation and fate of photochemical oxidants, primary and secondary PM concentrations, acid deposition, and air toxics, over regional and urban spatial scales for given inputs of meteorological conditions and emissions. CMAQ includes numerous science modules that simulate the emission, production, decay, deposition and transport of organic and inorganic gas-phase and particle pollutants in the atmosphere. The CMAQ model is a well-known and well-respected tool and has been used in numerous national and international applications.[J]

[J] More information available at: https://www.epa.gov/cmaq

The air quality modeling analysis used the 2016v1 platform with the most recent multi-pollutant CMAQ code available at the time of air quality modeling (CMAQ version 5.3.1). The 2016 CMAQ runs utilized the CB6r3 chemical mechanism (Carbon Bond with linearized halogen chemistry) for gas-phase chemistry, and AERO7 (aerosol model with non-volatile primary organic aerosol) for aerosols. The CMAQ model is regularly peer reviewed, with the most recent review completed in 2019 on version 5.2 and 5.3beta.[13]

6.3.2 Model Domain and Configuration

The CMAQ modeling analyses used a domain covering the continental United States, as shown in Figure 6-4. This single domain covers the entire continental U.S. (CONUS) and large portions of Canada and Mexico using 12 km × 12 km horizontal grid spacing.[K] The 2016 simulation used a Lambert Conformal map projection centered at (-97, 40) with true latitudes at 33 and 45 degrees north. The model extends vertically from the surface to 50 millibars (approximately 17,600 meters) using a sigma-pressure coordinate system with 35 vertical layers.

[K] The 12 km grid resolution of the air quality modeling domain does not allow us to analyze the concentration gradients of NO_2 and other pollutants which are likely to occur within a few hundred meters near roads.

Figure 6-4: Map of the CMAQ 12 km modeling domain (noted by the purple box)

6.3.3 Model Inputs

The key inputs to the CMAQ model include emissions from anthropogenic and biogenic sectors, meteorological data, and initial and boundary conditions.

The onroad emissions inputs used for the 2045 reference and control scenarios are summarized in Chapter 5 of the DRIA, and emissions inputs for other sectors are described in the documentation for the 2016v1 modeling platform.[14] The reference scenario represents projected 2045 emissions without the proposed rule, and the control scenario represents projected 2045 emissions with the proposed rule. The AQM TSD also contains a detailed discussion of the emissions inventory inputs used in our air quality modeling.[22]

The CMAQ meteorological input files were derived from simulations of the Weather Research and Forecasting Model (WRF) version 3.8 for the entire 2016 year.[15,16] The WRF Model is a state-of-the-science mesoscale numerical weather prediction system developed for both operational forecasting and atmospheric research applications.[17] The meteorological outputs from WRF were processed to create 12 km model-ready inputs for CMAQ using the Meteorology-Chemistry Interface Processor (MCIP) version 4.3. These inputs included hourly varying horizontal wind components (i.e., speed and direction), temperature, moisture, vertical diffusion rates, and rainfall rates for each grid cell in each vertical layer.[18]

The boundary and initial species concentrations were provided by a northern hemispheric CMAQ modeling platform for the year 2016.[19,20] The hemispheric-scale platform uses a polar stereographic projection at 108 km resolution to completely and continuously cover the northern hemisphere for 2016. Meteorology is provided by WRF v3.8. Details on the emissions used for hemispheric CMAQ can be found in the 2016 hemispheric emissions modeling platform TSD.[21] The atmospheric processing (transformation and fate) was simulated by CMAQ (v5.2.1) using the CB6r3 and the aerosol model with non-volatile primary organic carbon (AE6nvPOA). The CMAQ model also included the on-line windblown dust emission sources (excluding agricultural land), which are not always included in the regional platform but are important for large-scale transport of dust.

6.3.4 **CMAQ Evaluation**

The CMAQ predictions for ozone, fine particulate matter, sulfate, nitrate, ammonium, organic carbon, elemental carbon, nitrogen deposition, and specific air toxics (formaldehyde, acetaldehyde, benzene and naphthalene) from the 2016 base scenario were compared to measured concentrations in order to evaluate the ability of the modeling platform to replicate observed concentrations. This evaluation was comprised of statistical and graphical comparisons of paired modeled and observed data. Details on the model performance evaluation, including a description of the methodology, the model performance statistics, and results, are provided in the Air Quality Modeling TSD for this proposed rulemaking (AQM TSD).[22]

6.3.5 **Model Simulation Scenarios**

As part of our analysis for this rulemaking, the hourly CMAQ outputs were used to calculate 8-hour ozone design value concentrations, daily and annual $PM_{2.5}$ design value concentrations, annual NO_2 concentrations, annual CO concentrations, annual and seasonal (summer and winter) air toxics concentrations, visibility levels and annual total nitrogen deposition for each of the following scenarios:

- 2016 base year
- 2045 reference
- 2045 control

Air quality modeling was done for the future year 2045 when the program will be fully implemented and when most of the regulated fleet will have turned over. We use the predictions from the air quality model in a relative sense by combining the 2016 base-year predictions with predictions from each future-year scenario and applying these modeled ratios to ambient air quality observations to estimate 8-hour ozone concentrations for the May 1 - Sept 30 ozone season, daily and annual $PM_{2.5}$ concentrations, and visibility impairment for each of the 2045 scenarios. The ambient air quality observations are average conditions, on a site-by-site basis, for a period centered around the model base year (i.e., 2014-2018).[23] Additional predictions from the CMAQ model are used in the demographic analysis (Chapter 6.4.9) and in the benefits analysis described in Chapter 8.3.1 of the RIA. The CO, NO_2, annual and seasonal formaldehyde, acetaldehyde, benzene, naphthalene, and annual nitrate deposition projections were not predicted in a relative sense due to the limited observational data available.

The projected daily and annual $PM_{2.5}$ design values were calculated using the Speciated Modeled Attainment Test (SMAT) approach. Details of the SMAT procedures can be found in the report "Procedures for Estimating Future $PM_{2.5}$ Values for the CAIR Final Rule by Application of the (Revised) Speciated Modeled Attainment Test (SMAT)."[24] Several updated datasets and techniques were used for this analysis. These changes are fully described within the technical support document for the Final Transport Rule Air Quality Modeling Technical Support Document.[25] The projected 8-hour ozone design values were calculated using the approach identified in EPA's guidance on air quality modeling attainment demonstrations.[26]

6.4 Air Quality Modeling Results of the Proposed Rule

This section describes the results of the air quality modeling analysis done for the proposed rule. The "reference" scenario represents projected 2045 air quality without the proposed rule and the "control" scenario represents projected 2045 air quality with the proposed rule. This section presents modeled changes in ambient concentrations of air pollutants when comparing the "reference" and "control" scenarios. Decreases in concentration mean that the "control" scenario decreases the pollutant concentration compared to the "reference" scenario.

Everything in the reference and control scenarios was held constant except the onroad inventories, which reflected the application of the proposed standards at the time we conducted the modeling. This includes the meteorological data (reflecting calendar year 2016 conditions) and the emissions for all other sources, including boundary conditions and initial conditions used in the air quality modeling methodology.

The reference and control scenarios include projections of existing control programs that EPA had already adopted for mobile source emissions, as well as other federal, state and local programs which are expected to reduce concentrations of pollutants in the ambient air in the future. These control programs include (but are not limited to) the Tier 3 Motor Vehicle Emission and Fuel Standards (79 FR 23414, April 28, 2014), the New Marine Compression-Ignition Engines at or Above 30 Liters per Cylinder Rule (75 FR 22895, April 30, 2010), the Locomotive and Marine Compression-Ignition Engine Rule (73 FR 25098, May 6, 2008), the Clean Air Nonroad Diesel (69 FR 38957, June 29, 2004), and the Heavy-Duty Engine and Vehicle Standards and Highway Diesel Fuel Sulfur Control Requirements (66 FR 5002, January 18, 2001).

Not included in the reference or control scenarios for the air quality modeling are additional federal or state programs that were not finalized at the time that the air quality modeling analysis was initiated. For example, the CARB Heavy-Duty Low NO_X Omnibus rule and the CA Advanced Clean Trucks (ACT) rule were not final, so the emission reductions associated with these rulemakings are not included in the air quality modeling analysis for the proposed rule.[L,M]

[L] Additional information on the CARB Omnibus program is available in Section I.D of the preamble for the proposed rule. Additional discussion on the CARB ACT program is available in Sections I.D, VI.D, and XI of the preamble for the proposed rule.

[M] The draft RIA Chapter 5 Appendix 6 presents a sensitivity analysis of the estimated emission inventory impacts from nationwide adoption of the Omnibus rule; the draft RIA was made available with the proposed rule and is available on the EPA website for this rulemaking: https://www.epa.gov/regulations-emissions-vehicles-and-engines/proposed-rule-and-related-materials-control-air-1.

Since we did not include these rules in either the reference or control scenarios, our modeling for this rule appropriately reflects the expected air quality improvements from this action.

6.4.1 **Ozone Design Value Impacts of Proposed Rulemaking**

This section summarizes the ozone air quality impacts of the proposed rule in 2045, based on our CMAQ modeling. Our modeling indicates that ozone design value concentrations will decrease dramatically in many areas of the country as a result of the proposed rule.

Figure 6-5 presents the changes in 8-hour ozone design value concentrations in 2045.[N]

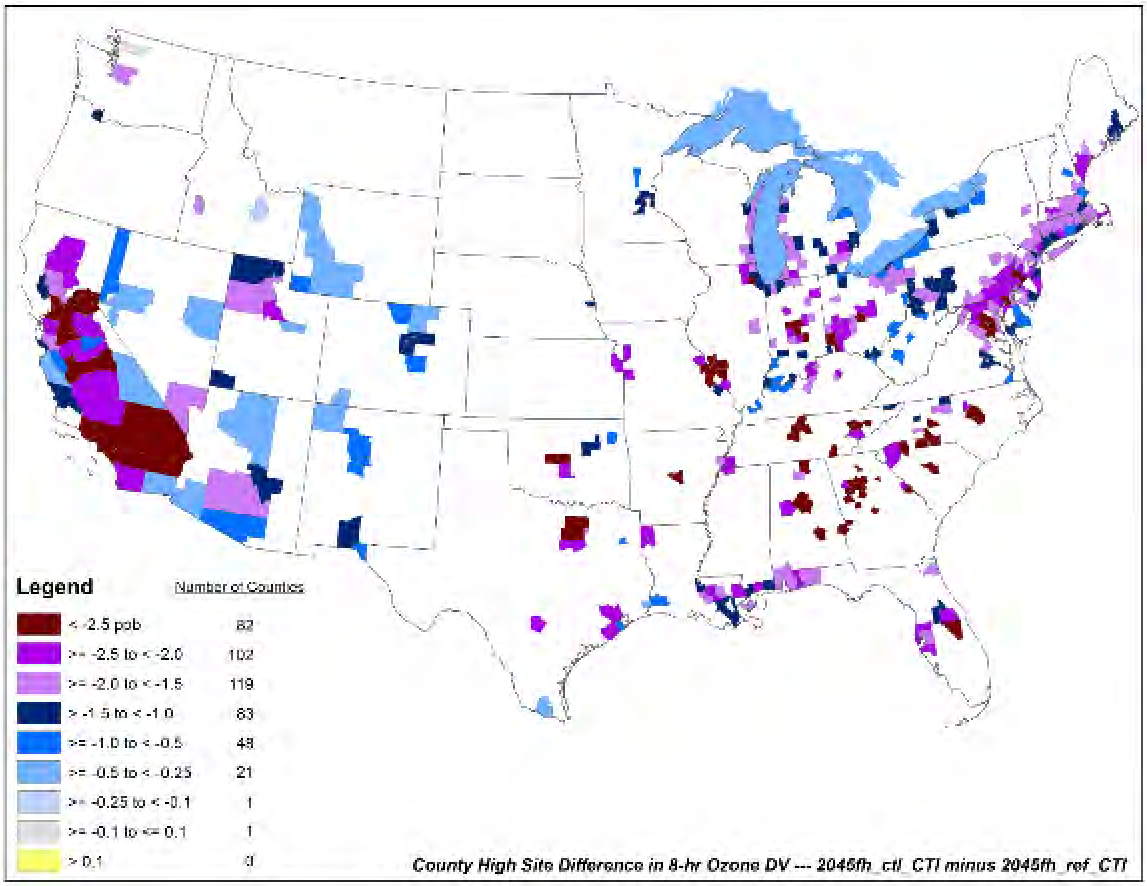


Figure 6-5: Projected Change in 8-hour Ozone Design Values in 2045 due to Proposed Rule

As shown in Figure 6-5, the majority of the design value decreases in 2045 are greater than 1.5 ppb. There are also 82 counties with projected 8-hour ozone design value decreases of more than 2.5 ppb; many of the counties with the largest design value decreases are in California, and in the Atlanta and St. Louis urban areas. The maximum projected decrease in an 8-hour ozone design value in 2045 is 5.1 ppb in Riverside County, California. Not all counties have monitor

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[N] An 8-hour ozone design value is the concentration that determines whether a monitoring site meets the NAAQS for ozone. The full details involved in calculating an 8-hour ozone design value are given in appendix I of 40 CFR part 50.

data that meets the requirements to calculate a design value concentration; counties without a calculated design value are left white.

Table 6-1 shows the average projected change, due to the proposed rule, in 2045 8-hour ozone design values for: (1) all modeled counties (with 2016 base case design values), (2) counties with 2016 base case design values that are above the level of the 2015 ozone NAAQS, (3) counties with 2016 base case design values that are equal to or within 10% below the level of the 2015 NAAQS, (4) counties with 2045 reference scenario design values that are above the level of the 2015 ozone NAAQS, (5) counties with 2045 reference scenario design values that are equal to or within 10% below the level of the 2015 ozone NAAQS, (6) counties with 2045 control scenario design values that are above the level of the 2015 ozone NAAQS, and (7) counties with 2045 control scenario design values that are equal to or within 10% below the level of the 2015 ozone NAAQS. Counties within 10 percent of the level of the NAAQS are intended to reflect counties that although not violating the standards, would also be impacted by changes in ambient levels of ozone as they work to ensure long-term attainment or maintenance of the ozone NAAQS. On a population-weighted basis, the average modeled future-year 8-hour ozone design value is projected to decrease by over 2 ppb in 2045 due to the proposed rule.

Table 6-1: Average Change in Projected 8-hour Ozone Design Values in 2045 due to Proposed Rule

Projected Design Value Category	**Number of Counties**	**2045 Population[a]**	**Average Change in 2045 Design Value (ppb)**	**Population-Weighted Average Change in Design Value (ppb)**
all modeled counties	457	246,949,949	-1.87	-2.23
counties with 2016 base year design values above the level of the 2015 8-hour ozone standard	118	125,319,158	-2.12	-2.43
counties with 2016 base year design values within 10% of the 2015 8-hour ozone standard	245	93,417,097	-1.83	-2.10
counties with 2045 reference design values above the level of the 2015 8-hour ozone standard	15	37,758,488	-2.26	-3.03
counties with 2045 reference design values within 10% of the 2015 8-hour ozone standard	56	39,302,665	-1.78	-2.02
counties with 2045 control design values above the level of the 2015 8-hour ozone standard	10	27,930,138	-2.36	-3.34
counties with 2045 control design values within 10% of the 2015 8-hour ozone standard	42	31,395,617	-1.69	-1.77

[a] Population numbers based on Woods & Poole data. Woods & Poole Economics, Inc. (2015). Complete Demographic Database. Washington, DC. http://www.woodsandpoole.com/index.php.

These modeling results project that there would be 15 counties with 8-hour ozone design values above the level of the 2015 ozone NAAQS in 2045 without the proposed rule or any other additional standards in place. Table 6-2 below presents the changes in design values for these counties.

Table 6-2: Change in 8-hour Ozone Design Values for Counties Projected to be Above the Level of the 2015 8-hour Ozone NAAQS in 2045

County Name, State	Population in 2045a	Change in 2045 projected 8-hour Ozone Design Value (DV) (ppb)	2045 Reference Ozone Design Value (ppb)	2045 Control Ozone Design Value (ppb)
San Bernardino, California	3,191,663	-4.6	98.0	93.4
Los Angeles, California	11,755,545	-3.3	92.3	89.0
Riverside, California	3,926,478	-5.1	83.3	78.2
Fairfield, Connecticut	1,050,293	-1.4	79.4	78.0
Imperial, California	296,070	-0.3	76.6	76.3
Kern, California	1,251,350	-2.2	76.5	74.3
San Diego, California	4,452,722	-2.5	75.2	72.7
Fresno, California	1,371,355	-2.8	74.9	72.1
Richmond, New York	614,033	-0.8	73.7	72.9
Mariposa, California	20,630	-0.6	72.0	71.4
Salt Lake, Utah	1,387,960	-1.9	71.9	70.0
Tulare, California	601,851	-2.4	71.5	69.1
Sheboygan, Wisconsin	124,284	-1.9	71.4	69.5
Davis, Utah	556,296	-1.9	71.3	69.4
Harris, Texas	7,157,959	-2.2	71.2	69.0
a Population numbers based on Woods & Poole data. Woods & Poole Economics, Inc. (2015). Complete Demographic Database. Washington, DC. http://www.woodsandpoole.com/index.php.				

Our modeling predicts that the proposed rule would reduce ozone design values in some counties from above the level of the standard to below it. While the number of counties with projected design values above the level of the NAAQS is less certain than the average projected changes in design values, our modeling projects that in 2045 ozone design values in five counties (Salt Lake and Davis Counties in Utah, Tulare County in California, Sheboygan County in Wisconsin, and Harris County in Texas) will change from being above the level of the standard in the reference scenario to being below the level of the standard in the control scenario. The projected population in these five counties in 2045 is almost 10 million people.

As described in Chapter 4 of this RIA, the science of ozone formation, transport, and accumulation is complex. The air quality modeling projects ozone design value decreases as a result of emissions changes from the proposed standards in the vast majority of counties. This change in ozone results from interactions between photochemistry, background concentrations of ozone, VOC and NO_X, local emissions and meteorology. However, there is one county in 2045 that is projected to have no change in modeled ozone design value concentration (Skagit County, Washington).

6.4.2 **Annual $PM_{2.5}$ Design Value Impacts of Proposed Rulemaking**

This section summarizes the annual average $PM_{2.5}$ air quality impacts of the proposed rule in 2045, based on our CMAQ modeling. Our modeling indicates that annual $PM_{2.5}$ design values will decrease due to the proposed rule. The decreases in annual $PM_{2.5}$ design values are due to the projected reductions in NO_X, primary $PM_{2.5}$, and VOC emissions. We expect this rule's reductions in directly emitted $PM_{2.5}$ will also contribute to reductions in $PM_{2.5}$ concentrations near roadways, although our air quality modeling is not of sufficient resolution to capture that impact.

Figure 6-6 presents the changes in annual $PM_{2.5}$ design values in 2045.[O]

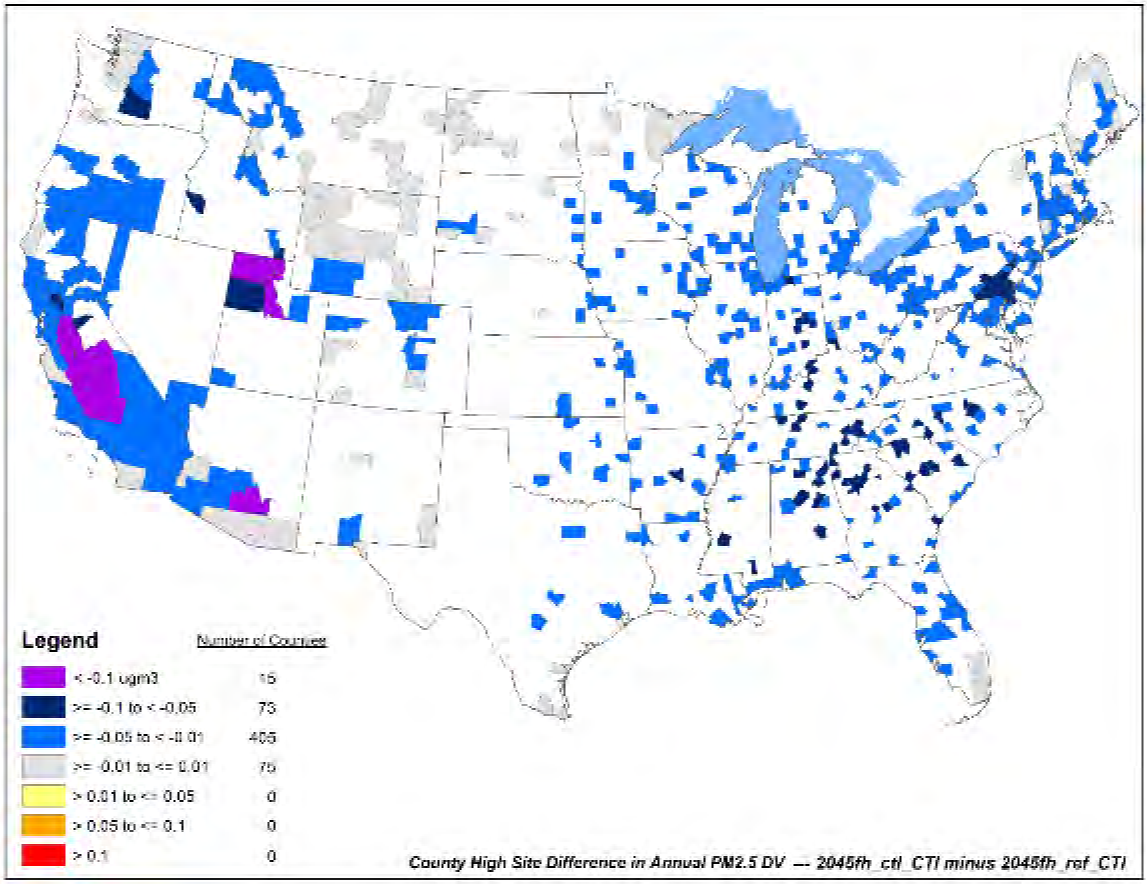

Figure 6-6: Projected Change in Annual $PM_{2.5}$ Design Values in 2045 due to Proposed Rule

As shown in Figure 6-6, we project that in 2045 most counties will have design value decreases of between 0.01 µg/m^3 and 0.05 µg/m^3. There are also 15 counties with projected annual $PM_{2.5}$ design value decreases of more than 0.1 µg/m^3; these counties are in California and Utah. The maximum projected decrease in a 2045 annual $PM_{2.5}$ design value is 0.21 µg/m^3 in Tulare County, California. Not all counties have monitor data that meets the requirements to calculate a design value concentration; counties without a calculated design value are left white.

Table 6-3 presents the average projected change, due to the proposed rule, in 2045 annual $PM_{2.5}$ design values for: (1) all modeled counties (with 2016 base case design values), (2) counties with 2016 base case design values that are above the level of the 2012 annual $PM_{2.5}$ standard, (3) counties with 2016 base case design values that are equal to or within 10% below the level of the 2012 standard, (4) counties with 2045 reference scenario design values that are above the level of the 2012 annual $PM_{2.5}$ standard, (5) counties with 2045 reference scenario design values that are equal to or within 10% below the level of the 2012 annual $PM_{2.5}$ standard,

[O] An annual $PM_{2.5}$ design value is the concentration that determines whether a monitoring site meets the annual NAAQS for $PM_{2.5}$. The full details involved in calculating an annual $PM_{2.5}$ design value are given in appendix N of 40 CFR part 50.

(6) counties with 2045 control scenario design values that are above the level of the 2012 annual $PM_{2.5}$ standard and (7) counties with 2045 control scenario design values that are equal to or within 10% below the level of the 2012 annual $PM_{2.5}$ standard. Counties within 10 percent of the level of the standard are intended to reflect counties that although not violating the standards, would also be impacted by changes in ambient levels of $PM_{2.5}$ as they work to ensure long-term attainment or maintenance of the 2012 annual $PM_{2.5}$ NAAQS. On a population-weighted basis, the average modeled future year annual $PM_{2.5}$ design value is projected to decrease by 0.04 $\mu g/m^3$ due to the proposed rule.

Table 6-3: Average Change in Projected Annual $PM_{2.5}$ Design Values in 2045 due to Proposed Rule

Projected Design Value Category	Number of Counties	2045 Population[a]	Average Change in 2045 Design Value (ug/m^3)	Population-Weighted Average Change in Design Value (ug/m^3)
all modeled counties	568	273,604,437	-0.04	-0.04
counties with 2016 base year design values above the level of the 2012 annual $PM_{2.5}$ standard	17	26,726,354	-0.09	-0.05
counties with 2016 base year design values within 10% of the 2012 annual $PM_{2.5}$ standard	5	4,009,527	-0.06	-0.06
counties with 2045 reference design values above the level of the 2012 annual $PM_{2.5}$ standard	12	25,015,974	-0.10	-0.05
counties with 2045 reference design values within 10% of the 2012 annual $PM_{2.5}$ standard	6	1,721,445	-0.06	-0.06
counties with 2045 control design values above the level of the 2012 annual $PM_{2.5}$ standard	10	23,320,070	-0.10	-0.05
counties with 2045 control design values within 10% of the 2012 annual $PM_{2.5}$ standard	8	3,417,349	-0.08	-0.09

[a] Population numbers based on Woods & Poole data. Woods & Poole Economics, Inc. (2015). Complete Demographic Database. Washington, DC. http://www.woodsandpoole.com/index.php.

There are 12 counties, mostly in California, that are projected to have annual $PM_{2.5}$ design values above the level of the NAAQS in 2045 without the proposed rule or any other additional standards in place. Table 6-4 below presents the changes in design values for these counties.

Table 6-4: Change in Annual $PM_{2.5}$ Design Values for Counties Projected to be Above the Level of the Annual $PM_{2.5}$ NAAQS in 2045

County Name, State	Population in 2045[a]	Change in 2045 Projected Annual $PM_{2.5}$ Design Value (DV) (μg/m^3)	2045 Reference Design Value (μg/m^3)	2045 Control Design Value (μg/m^3)
Kern, California	1,251,350	-0.15	15.78	15.63
Kings, California	185,866	-0.19	14.82	14.63
San Bernardino, California	3,191,663	-0.02	14.21	14.19
Plumas, California	21,297	-0.05	14.12	14.06
Tulare, California	601,851	-0.21	14.05	13.84
Riverside, California	3,926,478	-0.04	13.43	13.39
Imperial, California	296,070	-0.02	12.93	12.91
Fresno, California	1,371,355	-0.16	12.87	12.71
Los Angeles, California	11,755,545	-0.02	12.26	12.24
Pinal, Arizona	718,595	-0.11	12.24	12.13
Stanislaus, California	716,019	-0.16	12.17	12.02
San Joaquin, California	979,885	-0.11	12.07	11.96

[a] Population numbers based on Woods & Poole data. Woods & Poole Economics, Inc. (2015). Complete Demographic Database. Washington, DC. http://www.woodsandpoole.com/index.php.

Our modeling predicts that the proposed rule would reduce annual $PM_{2.5}$ design values in some counties from above the level of the standard to below it. While the number of counties with projected design values above the level of the NAAQS is less certain than the average changes in design values, annual $PM_{2.5}$ design values in two counties (Stanislaus County, California and San Joaquin County, California) are projected to change from being above the level of the standard in the reference scenario to being below the level of the standard in the control scenario. The projected population in these two counties in 2045 is over 1.5 million people.

6.4.3 24-hour $PM_{2.5}$ Design Value Impacts of Proposed Rulemaking

This section summarizes the 24-hour $PM_{2.5}$ air quality impacts of the proposed rule in 2045, based on our CMAQ modeling. Our modeling indicates that most 24-hour $PM_{2.5}$ design values would decrease due to the proposed rule. The decreases in 24-hour $PM_{2.5}$ design values are due to the projected reductions in NO_X, primary $PM_{2.5}$, and VOC emissions. We expect this rule's reductions in directly emitted $PM_{2.5}$ will also contribute to reductions in $PM_{2.5}$ concentrations near roadways, although our air quality modeling is not of sufficient resolution to capture that impact.

Figure 6-7 presents the changes in 24-hour $PM_{2.5}$ design values in 2045.[P]

[P] A 24-hour $PM_{2.5}$ design value is the concentration that determines whether a monitoring site meets the 24-hour NAAQS for $PM_{2.5}$. The full details involved in calculating a 24-hour $PM_{2.5}$ design value are given in appendix N of 40 CFR part 50.

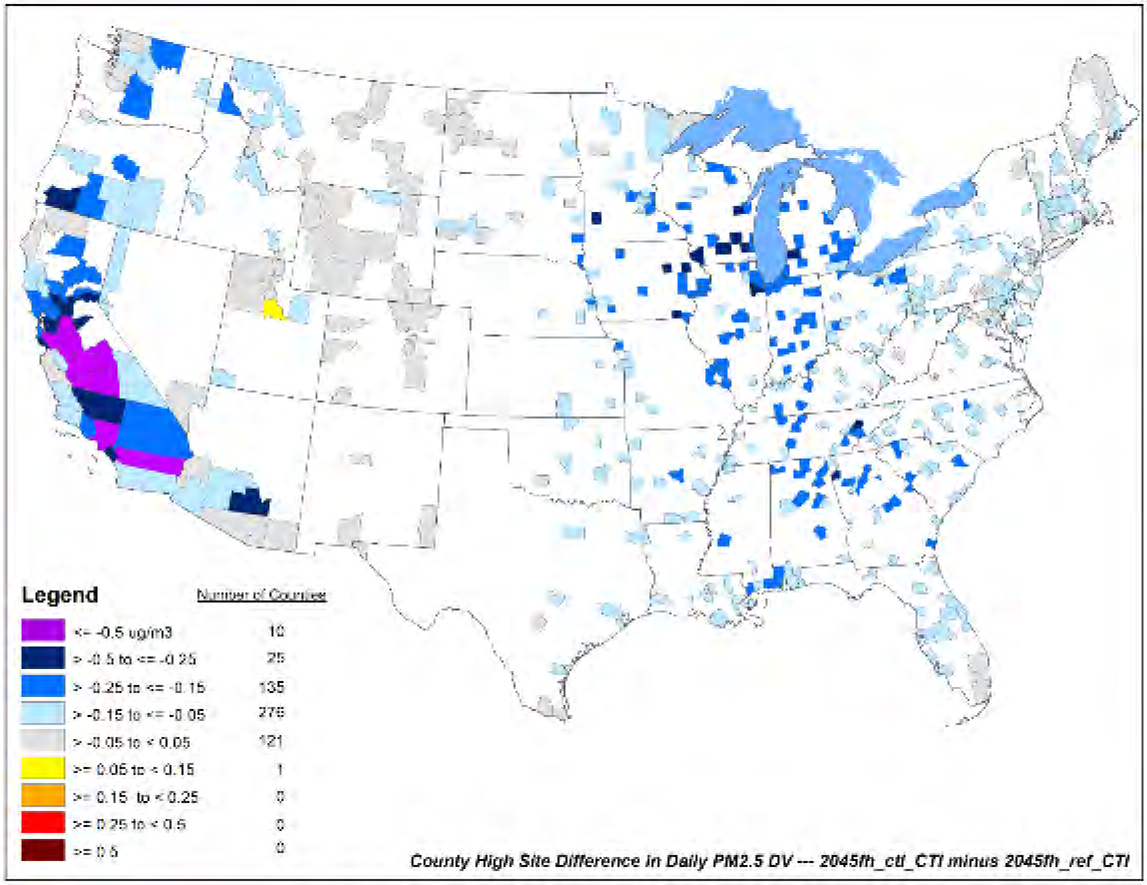

Figure 6-7: Projected Change in 24-hour $PM_{2.5}$ Design Values in 2045 due to Proposed Rule

As shown in Figure 6-7, in 2045 there are 170 counties with projected 24-hour $PM_{2.5}$ design value decreases greater than 0.15 µg/m^3. These counties are in mainly in the midwest, southeast and western United States. The maximum projected decrease in a 2045 24-hour $PM_{2.5}$ design value is 1.79 µg/m^3 in Tulare County, California. Not all counties have monitor data that meets the requirements to calculate a design value concentration; counties without a calculated design value are left white.

Table 6-5 shows the average projected change, due to the proposed rule, in 2045 24-hour $PM_{2.5}$ design values for: (1) all modeled counties (with 2016 base case design values), (2) counties with 2016 base case design values that are above the level of the 2006 24-hour $PM_{2.5}$ standard, (3) counties with 2016 base case design values that are equal to or within 10% below the level of the 2006 24-hour $PM_{2.5}$ standard, (4) counties with 2045 reference scenario design values that are above the level of the 2006 24-hour $PM_{2.5}$ standard, (5) counties with 2045 reference scenario design values that are equal to or within 10% below the level of the 2006 24-hour $PM_{2.5}$ standard, (6) counties with 2045 control scenario design values that are above the level of the 2006 24-hour $PM_{2.5}$ standard, and (7) counties with 2045 control scenario design values that are equal to or within 10% below the level of the 2006 24-hour $PM_{2.5}$ NAAQS. Counties within 10 percent of the level of the standard are intended to reflect counties that although not violating the standards, would also be impacted by changes in ambient levels of $PM_{2.5}$ as they work to ensure long-term attainment or maintenance of the 2006 24-hour $PM_{2.5}$

NAAQS. On a population-weighted basis, the average modeled future-year 24-hour $PM_{2.5}$ design value is projected to decrease by 0.17 μg/m^3 in 2045 due to the proposed rule.

Table 6-5: Average Change in Projected 24-hour $PM_{2.5}$ Design Values in 2045 due to Proposed Rule

Projected Design Value Category	Number of Counties	2045 Population[a]	Average Change in 2045 Design Value (ug/m^3)	Population-Weighted Average Change in Design Value (ug/m^3)
all modeled counties	568	272,852,777	-0.12	-0.17
counties with 2016 base year design values above the level of the 2006 daily PM2.5 standard	33	28,394,253	-0.40	-0.67
counties with 2016 base year design values within 10% of the 2006 daily PM2.5 standard	15	13,937,416	-0.18	-0.27
counties with 2045 reference design values above the level of the 2006 daily PM2.5 standard	29	14,447,443	-0.38	-0.55
counties with 2045 reference design values within 10% of the 2006 daily PM2.5 standard	12	22,900,297	-0.30	-0.59
counties with 2045 control design values above the level of the 2006 daily PM2.5 standard	29	14,447,443	-0.38	-0.55
counties with 2045 control design values within 10% of the 2006 daily PM2.5 standard	10	19,766,216	-0.26	-0.60

[a] Population numbers based on Woods & Poole data. Woods & Poole Economics, Inc. (2015). Complete Demographic Database. Washington, DC. http://www.woodsandpoole.com/index.php.

There are 29 counties that are projected to have 24-hour $PM_{2.5}$ design values above the level of the NAAQS in 2045 without the proposed rule or any other additional controls in place. Table 6-6 below presents the changes in design values for these counties.

Table 6-6: Change in 24-hour $PM_{2.5}$ Design Values for Counties Projected to be Above the 24-hour $PM_{2.5}$ NAAQS in 2045

County Name, State	Population in 2045[a]	Change in 24-hour $PM_{2.5}$ Design Value ($\mu g/m^3$)	2045 Reference Design Value ($\mu g/m^3$)	2045 Control Design Value ($\mu g/m^3$)
Okanogan, Washington	47,922	-0.24	57.02	56.79
Ravalli, Montana	52,336	-0.03	56.93	56.90
Kern, California	1,251,350	-0.49	55.78	55.29
Fresno, California	1,371,355	-0.68	49.86	49.17
Jackson, Oregon	281,974	-0.28	49.22	48.94
Kings, California	185,866	-1.42	47.23	45.81
Plumas, California	21,297	-0.16	46.30	46.14
Klamath, Oregon	71,950	-0.16	44.39	44.24
Siskiyou, California	46,491	-0.03	44.04	44.02
Lincoln, Montana	19,924	-0.14	43.05	42.91
Tulare, California	601,851	-1.79	42.63	40.84
Missoula, Montana	139,759	-0.13	42.49	42.36
Lemhi, Idaho	8,830	-0.08	42.46	42.39
Lewis and Clark, Montana	95,256	-0.06	41.17	41.11
Flathead, Montana	150,424	-0.12	40.75	40.64
Yakima, Washington	289,388	-0.22	40.64	40.42
Lake, Oregon	8,605	-0.10	40.43	40.33
Stanislaus, California	716,019	-1.26	39.54	38.28
Lane, Oregon	440,599	-0.13	39.53	39.39
Josephine, Oregon	106,207	-0.27	39.46	39.19
Alameda, California	1,936,700	-0.30	38.81	38.51
Madera, California	208,957	-0.62	38.49	37.87
San Joaquin, California	979,885	-1.22	38.15	36.93
Kittitas, Washington	53,927	-0.17	37.82	37.65
Riverside, California	3,926,478	-0.51	37.35	36.84
Shoshone, Idaho	11,064	-0.16	37.07	36.91
Crook, Oregon	24,645	-0.19	37.00	36.81
Benewah, Idaho	10,426	-0.13	36.72	36.59
Salt Lake, Utah	1,387,960	0.02	36.04	36.06

[a] Population numbers based on Woods & Poole data. Woods & Poole Economics, Inc. (2015). Complete Demographic Database. Washington, DC. http://www.woodsandpoole.com/index.php.

While the count of modeled nonattainment counties is much less certain than the average changes in air quality, in 2045, there are no 24-hour $PM_{2.5}$ design values that are projected to change from being above the level of the standard in the reference case to being below the level of the standard in the proposed control case.

As described in Chapter 4 of this RIA, $PM_{2.5}$ in the atmosphere can be primary or secondary and its composition, transport, and accumulation is complex. The air quality modeling projects 24-hour $PM_{2.5}$ design value decreases as a result of emissions changes from the proposed rule in the vast majority of counties. However, there are a handful of counties where 24-hour $PM_{2.5}$ design values are projected to increase. These increases are likely due to elevated secondary $PM_{2.5}$ formation rates resulting from increased oxidant levels which occur during stagnant cold weather due to reductions in NO_X.

6.4.4 **Nitrogen Dioxide Concentration Impacts of Proposed Rulemaking**

This section summarizes the annual average NO_2 air quality impacts of the proposed rule in 2045, based on our CMAQ modeling. Our modeling indicates that annual average NO_2 concentrations would decrease as a result of the proposed rule, if finalized as proposed. Figure 6-8 presents the changes in annual NO_2 concentrations in 2045.

As shown in Figure 6-8, our modeling indicates that by 2045 annual NO_2 concentrations in the majority of the country would decrease between 0.01 and 0.1 ppb due to the proposed rule. However, decreases in annual NO_2 concentrations would be greater than 0.2 ppb along many highway corridors and greater than 0.3 ppb in most urban areas. The absolute reductions correspond to reductions of greater than 5 percent in annual NO_2 concentrations across much of the country, see Figure 6-9. Although we didn't model changes in 1-hour concentrations, the proposed rule would also likely decrease 1-hour NO_2 concentrations and help any potential nonattainment areas attain and maintenance areas maintain the NO_2 standard.[Q] We expect this rule will also contribute to reductions in NO_2 concentrations near roadways, although our air quality modeling is not of sufficient resolution to capture that impact.[R]

[Q] As noted in Chapter 6.1.3, there are currently no nonattainment areas for the NO_2 NAAQS.
[R] The 12 km grid resolution of the air quality modeling domain does not allow us to analyze the concentration gradients of NO_2 and other pollutants which are likely to occur within a few hundred meters near roads.

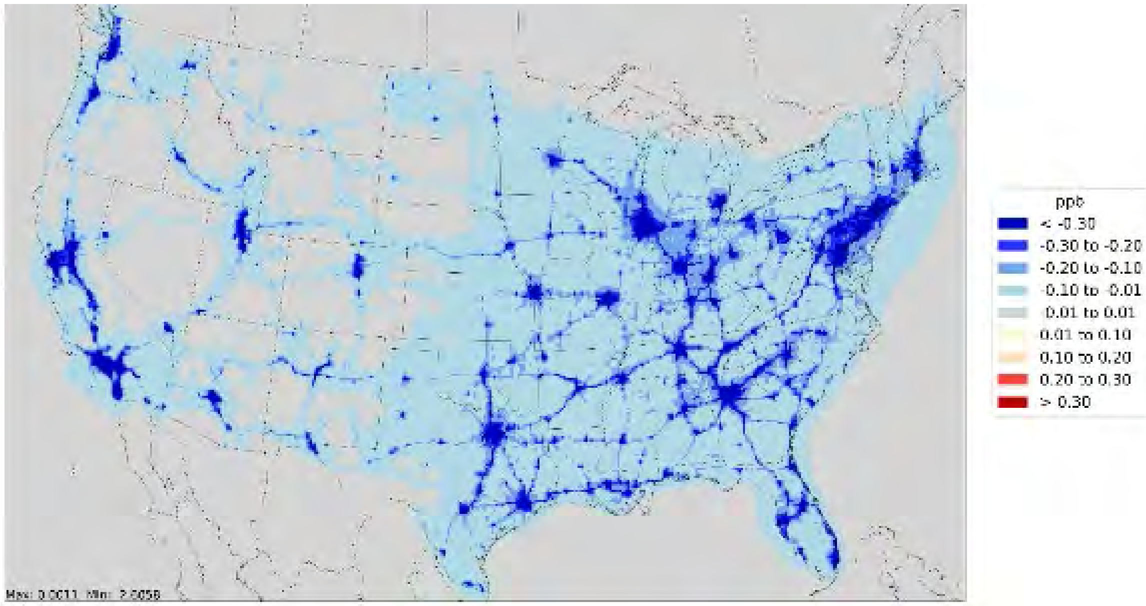

Figure 6-8: Projected Absolute Change in Annual Ambient NO2 Concentrations in 2045

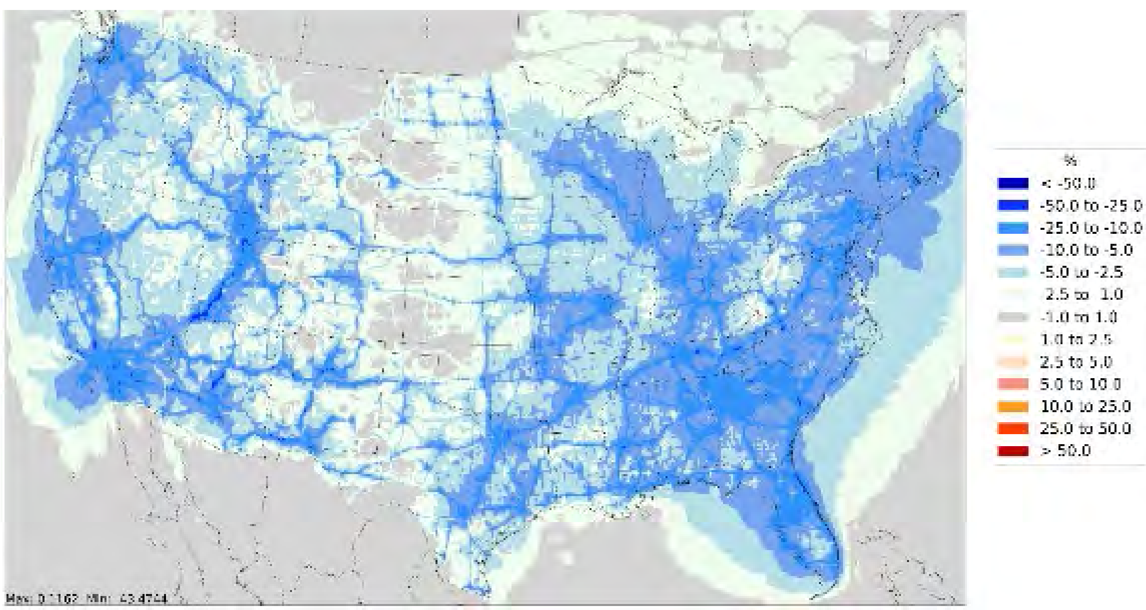

Figure 6-9: Percent Change in Annual Ambient NO_2 Concentrations in 2045

6.4.5 <u>**Carbon Monoxide Concentration Impacts of Proposed Rulemaking**</u>

This section summarizes the annual average CO air quality impacts of the proposed rule in 2045, based on our CMAQ modeling. Our modeling indicates that annual average CO concentrations would decrease as a result of the proposed rule. Figure 6-10 and Figure 6-11 present the absolute and percent changes in annual CO concentrations in 2045.

As shown in Figure 6-10, our modeling indicates that by 2045 annual CO concentrations in the majority of the country would decrease between 0.02 and 0.5 ppb due to the proposed rulemaking. However, decreases in annual CO concentrations would be greater than 1.5 ppb in some urban areas. The absolute reductions correspond to percent changes of less than 1 percent across the country, except for the Phoenix area, where there are some larger decreases between 1 and 2 percent. Although we didn't model changes in 8-hour or 1-hour concentrations, the proposed standards would also likely decrease 1-hour and 8-hour CO concentrations and help any potential nonattainment areas attain and maintenance areas maintain the CO standard.[S]

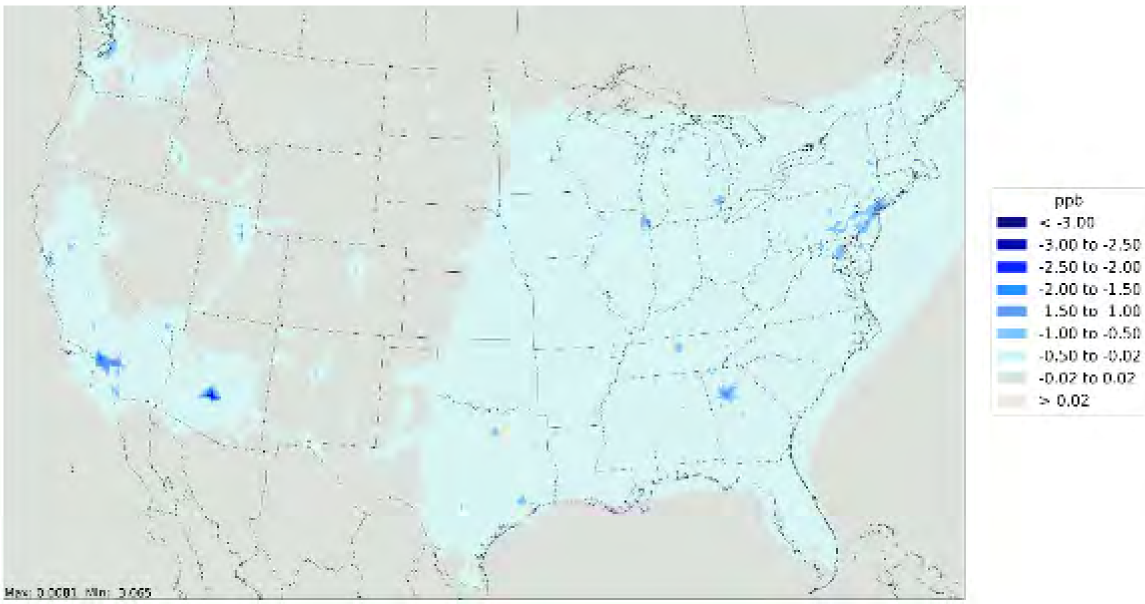

Figure 6-10: Absolute Change in Annual Ambient CO Concentrations in 2045

[S] As noted in Chapter 6.1.4, there are currently no nonattainment areas for the CO NAAQS.

Figure 6-11: Percent Change in Annual Ambient CO Concentrations in 2045

6.4.6 **Air Toxics Impacts of Proposed Rulemaking**

This section summarizes the changes in annual average air toxic (acetaldehyde, benzene, formaldehyde and naphthalene) concentrations in 2045 due to the proposed rule. Our modeling indicates that the proposed rule would have relatively little impact on national average ambient concentrations of the modeled air toxics in 2045. Annual percent changes are less than 1% for air toxics across most of the country. Annual absolute changes in ambient concentrations are generally less than 0.001 µg/m^3 for benzene and naphthalene (Figure 6-12 and Figure 6-13 below). There are small increases in acetaldehyde across the country, see Figure 6-14. The increases in acetaldehyde likely occur because species that lead to production or recycling of acetaldehyde increase as their reactions with nitrogen oxides decrease. For formaldehyde there are decreases across most of the country and a few areas with increases, see Figure 6-15. The increases in formaldehyde concentration due to the proposed rule are likely related to higher concentrations of OH radicals in areas where ozone increases due to NO_X emissions reductions (see Chapter 4.1.1.1).

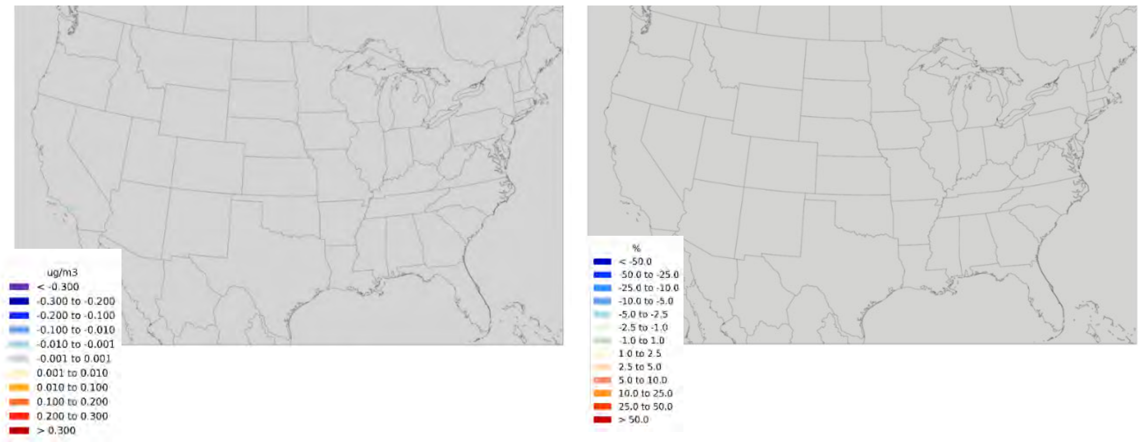

Figure 6-12: Changes in Ambient Benzene Concentrations in 2045 due to Proposed Rule: Absolute Changes in µg/m³ (left) and Percent Changes (right)

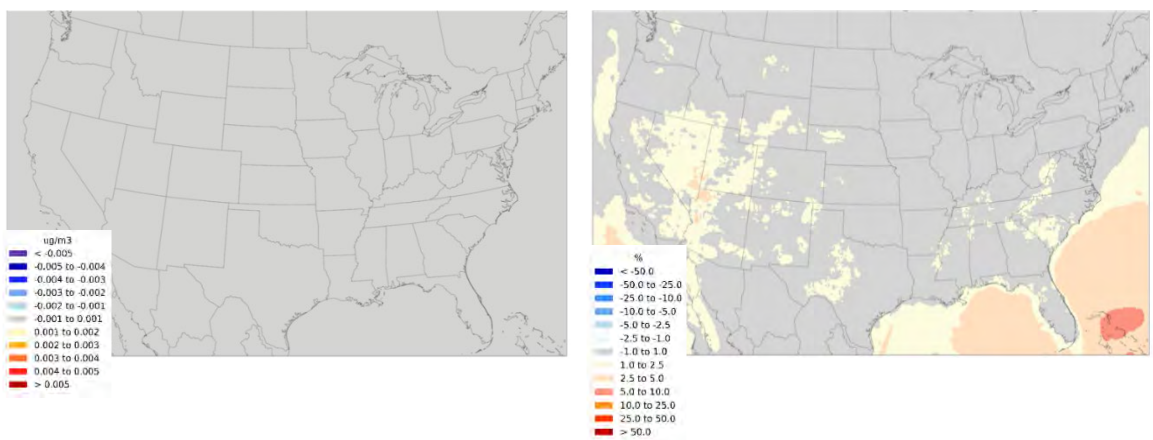

Figure 6-13: Changes in Ambient Naphthalene Concentrations in 2045 due to Proposed Rule: Absolute Changes in µg/m³ (left) and Percent Changes (right)

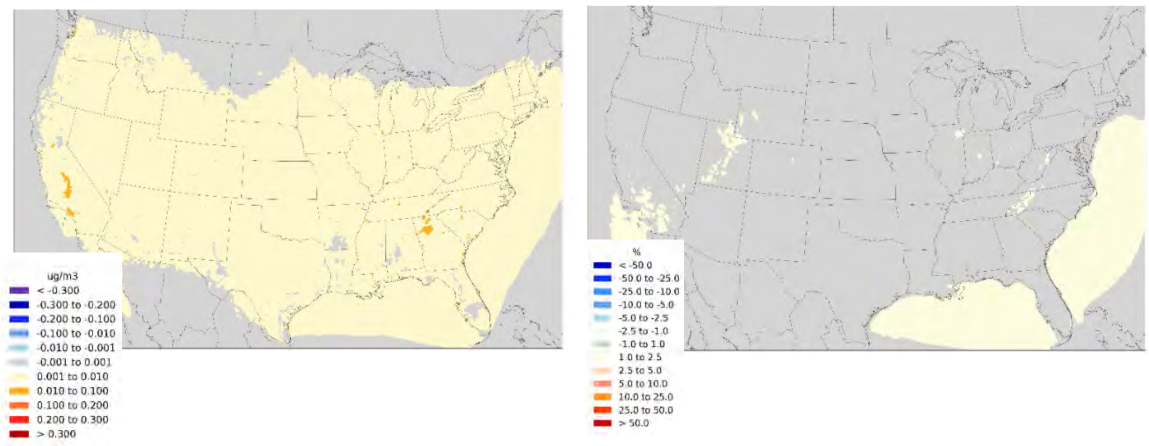

Figure 6-14: Changes in Ambient Acetaldehyde Concentrations in 2045 due to Proposed Rule: Absolute Changes in μg/m³ (left) and Percent Changes (right)

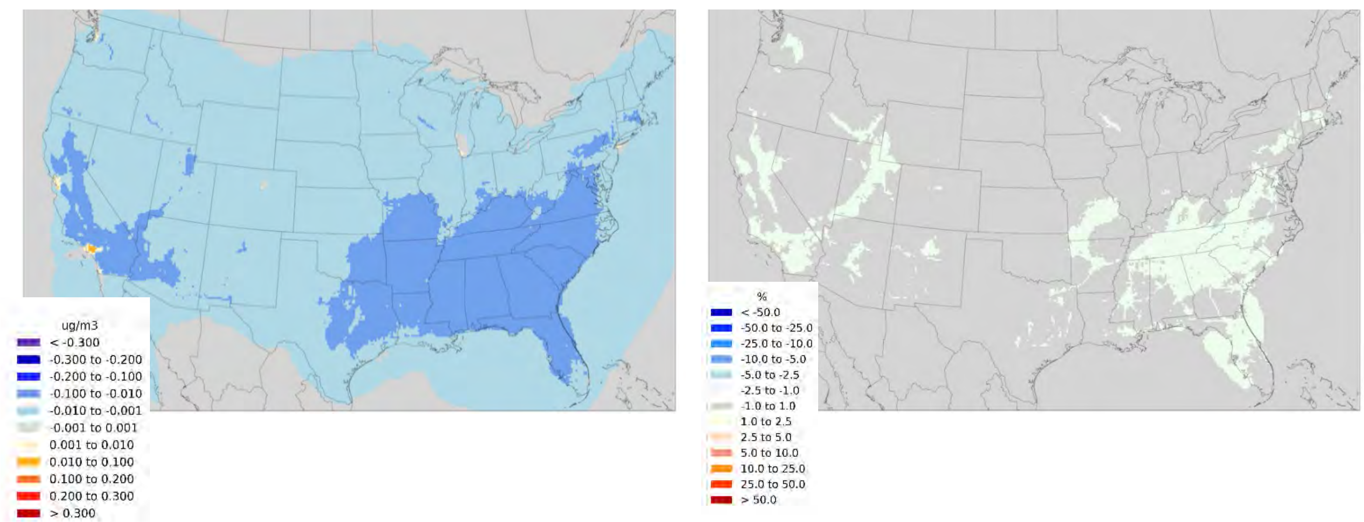

Figure 6-15: Changes in Ambient Formaldehyde Concentrations in 2045 due to Proposed Rule: Absolute Changes in μg/m³ (left) and Percent Changes (right)

6.4.7 **Visibility Impacts of Proposed Rulemaking**

Air quality modeling was used to project visibility conditions in 145 Mandatory Class I Federal areas across the U.S. with and without the proposed rule in 2045. The results show that in 2045, the proposed rule would improve projected visibility on the 20% most impaired days in all modeled areas.[T] The average visibility on the 20 percent most impaired days at all modeled

[T] The level of visibility impairment in an area is based on the light-extinction coefficient and a unitless visibility index, called a "deciview", which is used in the valuation of visibility. The deciview metric provides a scale for

Mandatory Class I Federal areas is projected to improve by 0.04 deciviews, or 0.37 percent, in 2045. The greatest improvement in visibility would occur in San Gorgonio and San Jacinto Wilderness Areas in California, where visibility is projected to improve by 1.56 percent (0.21 deciviews) in 2045 due to the proposed rule. The AQM TSD contains the full visibility results from 2045 for the 145 analyzed areas.[22]

6.4.8 Deposition Impacts of Proposed Rulemaking

Our air quality modeling projects decreases in nitrogen deposition due to the proposed rule. Figure 6-16 shows that by 2045 the proposed rule would result in decreases in nitrogen deposition over much of the eastern US and in urban areas of the western US, with the largest decreases in Atlanta and Los Angeles. Figure 6-17 indicates those decreases correspond to annual percent decreases of more than one percent over much of the country, with some localized decreases of over 4 percent. As discussed in Chapter 4.2.3.1, there is considerable evidence that nitrogen deposition adversely affects terrestrial, wetland, freshwater, and estuarine ecosystems. The reductions in nitrogen deposition expected from this rule, along with other actions to reduce NOx emissions, will reduce acidification and nitrogen enrichment across the US, including the Chesapeake Bay and other water bodies. This will lead to improved ecosystem functions, reduced coastal eutrophication, increased recreational demand, and other beneficial effects.

perceived visual changes over the entire range of conditions, from clear to hazy. Under many scenic conditions, the average person can generally perceive a change of one deciview. The higher the deciview value, the worse the visibility. Thus, an improvement in visibility is a decrease in deciview value.

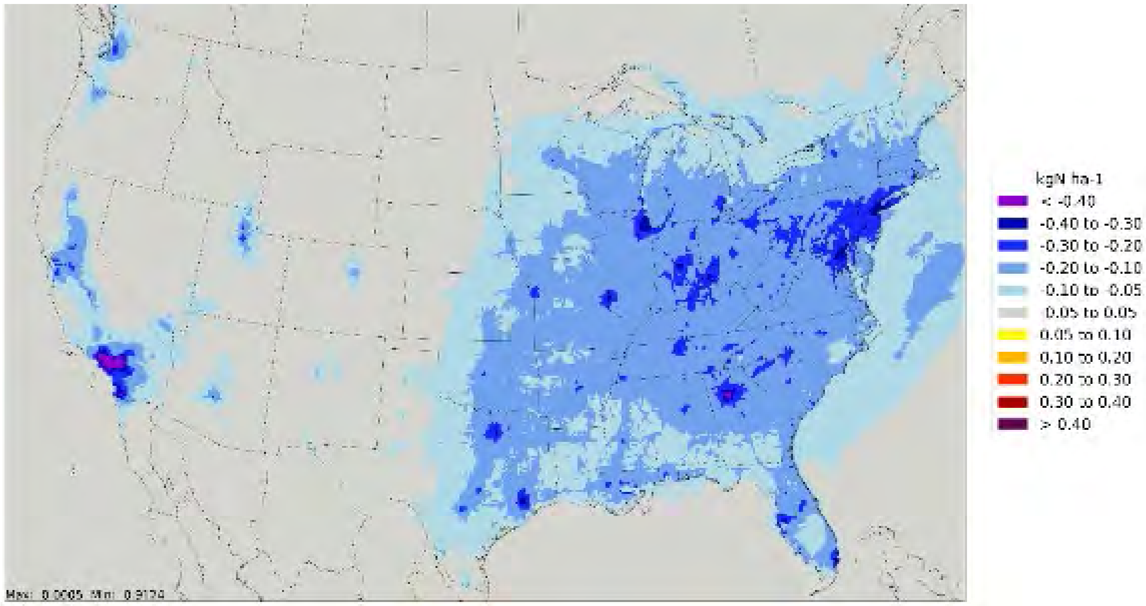

Figure 6-16: Absolute Change in Annual Deposition of Nitrogen in 2045

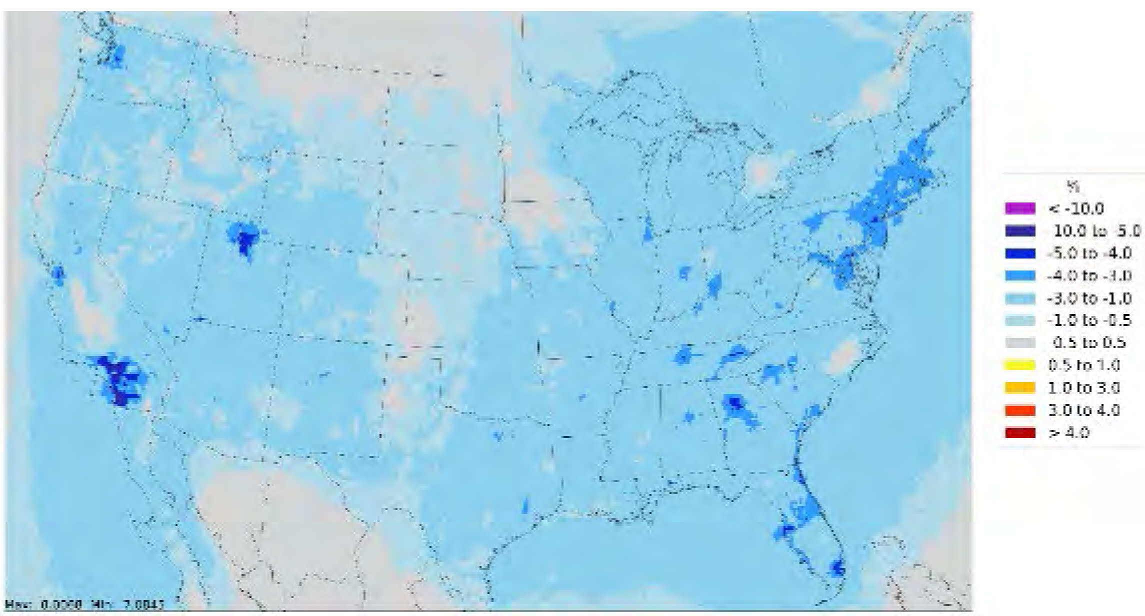

Figure 6-17: Percent Change in Annual Deposition of Nitrogen in 2045

6.4.9 <u>**Demographic Analysis of Air Quality**</u>

When feasible, EPA's Office of Transportation and Air Quality conducts full-scale photochemical air quality modeling to demonstrate how its national mobile source regulatory actions affect ambient concentrations of regional pollutants throughout the United States. As described in Chapter 6.2, the air quality modeling we conducted for the proposal also supports our analysis of future projections of $PM_{2.5}$ and ozone concentrations in a "baseline" scenario absent the rule and in a "control" scenario that assumes the rule is in place.[U] These baseline and control scenarios are also used as inputs to the health benefits analysis. As demonstrated in Chapter 6.4 and Chapter 8, the ozone and $PM_{2.5}$ improvements that are projected to result from the rule, and the health benefits associated with those pollutant reductions, will be substantial.

This air quality modeling data can also be used to conduct an analysis of how human exposure to future air quality varies with sociodemographic characteristics relevant to potential environmental justice concerns in scenarios with and without the rule in place. Although the spatial resolution of the air quality modeling is not sufficient to capture very local heterogeneity of human exposures, particularly the pollution concentration gradients near roads, the analysis does allow estimates of demographic trends at a national scale. We developed this approach by considering the purpose and specific characteristics of this rulemaking, as well as the nature of known and potential exposures to the air pollutants controlled by the standards. The heavy-duty standards apply nationally and will be implemented consistently across roadways throughout the U.S. The pollutant predominantly controlled by the standard is NO_X. Reducing emissions of NO_X will reduce formation of ozone and secondarily formed $PM_{2.5}$, which will reduce human exposures to regional concentrations of ambient ozone and $PM_{2.5}$. These reductions will be geographically widespread. Taking these factors into consideration, this demographic analysis evaluates the exposure outcome distributions that will result from this rule at the national scale with a focus on locations that are projected to have the highest baseline concentrations of $PM_{2.5}$ and ozone.

To analyze trends in exposure outcomes, we sorted projected 2045 baseline air quality concentrations from highest to lowest concentration and created two groups: areas within the contiguous U.S. with the worst air quality (grid cells with the highest 5 percent of concentrations) and the rest of the country (remaining 95 percent of grid cells). This approach can then answer two principal questions to determine disparity of air quality on the basis of race and ethnicity:

1. What is the racial and ethnic composition of areas with the worst baseline air quality in 2045?
2. Are those with the worst air quality benefitting more from the heavy-duty vehicle and engine standards?

We found that in the 2045 baseline, nearly double the number of people of color live within areas with the worst ozone and $PM_{2.5}$ air pollution compared to non-Hispanic Whites (NH-Whites).[V] We also found that (in absolute terms) the largest predicted improvements in both

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[U] Air quality modeling was performed for the proposed rule, which used emission reductions that are very similar to the emission reductions projected for the final rule. Given the similar structure of the proposed and final programs, we expect consistent geographic distribution of emissions reductions and modeled improvements in air quality, and that the air quality modeling conducted at the time of proposal adequately represents the final rule. Specifically, we expect this rule will decrease ambient concentrations of air pollutants, including significant improvements in ozone concentrations in 2045 as demonstrated in the air quality modeling analysis.

[V] The demographic analysis uses air quality modeling that has a contiguous U.S. domain. The analysis does not characterize distributional trends in areas of the U.S. that fall outside of this domain.

ozone and $PM_{2.5}$ are estimated to occur in areas with the worst baseline air quality, and a larger number of people of color are projected to reside in these areas.

6.4.9.1 Data and Methods

We began with projected 2045 baseline and control scenarios of modeled $PM_{2.5}$ and ozone concentration data (described in RIA Chapter 8.2.1). Ambient air quality concentration data (annual average µg/m^3 for $PM_{2.5}$ and May-September daily maximum 8-hour average ppb for ozone) was estimated at a standard grid resolution of 12km x 12km across the contiguous United States (CONUS).[W] Using 2045 baseline air quality data as our reference scenario, we sorted baseline air quality concentrations from highest to lowest concentration and compared two air quality concentration groups – grid cells in the highest 5 percent of the distribution of baseline concentrations and grid cells in the remaining 95 percent.[X]

Figure 6-18The maps in Figure 6-18 display the spatial distribution of grid cells with baseline concentrations in the highest 5 percent for both $PM_{2.5}$ and ozone concentrations. We retain this distinction throughout the analysis in order to track how air quality is distributed by air quality concentration group in the baseline and how the rule impacts air quality in these same grid cells with the standards in place.

The analysis also used population projections stratified by race/ethnicity, age, and sex are based on proprietary economic forecasting models developed by Woods and Poole in 2015.[27] The Woods and Poole database contains county-level projections of population by age, sex, and race out to 2050, relative to a baseline using the 2010 Census data. Population projections for each county are determined simultaneously with every other county in the U.S to consider patterns of economic growth and migration.[Y] The projected population for 2045 was extracted from the Environmental Benefits Mapping and Analysis Program – Community Edition (BenMAP-CE)[Z] at the same 12km x 12km grid resolution as the air quality data. Race and ethnicity of individuals projected to live in a given area were compiled into two broad categories, "people of color" and "Non-Hispanic White (NH-White)."[AA] We chose to aggregate race and ethnicity categories in this way to address the uncertainty present with population projections far into the future – it is difficult to predict with precision patterns of economic growth and migration (see Section 6.4.9.3 for more discussion about uncertainty). In 2045, there are 409 million people projected to be living in the contiguous United States; 208 million are projected to

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[W] Note that the ambient $PM_{2.5}$ and ozone air quality concentration data used in this analysis are different than the $PM_{2.5}$ and ozone design value metrics presented in RIA Chapter 6. Design values are pollutant concentrations that determine whether a monitoring site meets the NAAQS for a given pollutant.

[X] Using higher and lower percentiles to compare risks, exposures and outcomes has been applied by EPA's Office of Air and Radiation in previous distributional analyses of regulatory air quality modeling (see MATS, CSAPR, PM NAAQS) and is consistent with EPA's EJ Technical Guidance.

[Y] More information about the population projections can be found in the Technical Support Document (TSD) for the Final Revised Cross-State Air Pollution Rule Update for the 2008 Ozone Season NAAQS: *https://www.epa.gov/sites/default/files/2021-03/documents/estimating_pm2.5-_and_ozone-attributable_health_benefits_tsd.pdf*.

[Z] More information about BenMAP-CE can be found here: *https://www.epa.gov/benmap*. Additional information about the population projections used in this analysis can be found in Appendix J of the BenMAP-CE User's Manual: *https://www.epa.gov/benmap/benmap-ce-manual-and-appendices*.

[AA] "People of color" includes Black, Asian, Native American, Hawaiian/Pacific Islander, and Hispanic populations.

be NH-White and 201 million are projected to be people of color. To put these projections into perspective, 2010 populations for the contiguous United States were 201 million for NH-White and 106 million for people of color.

Additionally, this analysis looked at the distribution of poverty status within the same air quality concentration groups – 12km x 12km grid cells in the highest 5 percent of the distribution of baseline concentrations and grid cells in the remaining 95 percent. We applied county-level poverty status derived from the Census' American Community Survey (ACS) 5-year estimates from 2015 to 2019, which represents the fraction of county-level population below and above 200% of the poverty line.[BB] We note that measures of "current" poverty are not necessarily predictors of future poverty status; poverty status in the 2045 population could be different in terms of both scale and geographic location. However, for the purposes of this analysis, we believe applying a "current" measure of those who live above and below 200% of the poverty line is illustrative.

For each pollutant and air quality concentration group (i.e., highest 5% of concentration or remaining 95% of grid cells), we calculated the average baseline, control, and reduction in concentrations. We then summed the population by group (people of color or NH-White; populations above or below 200% of the poverty line) for each air quality concentration group.

(a)

(b)

Figure 6-18: Distributional maps of populated 12km grid cells across the contiguous United States in 2045. Darker areas represent the location of grid cells within the highest 5 percent of baseline concentrations for (a) $PM_{2.5}$ and (b) ozone

EPA received comments related to the methods the Agency used to analyze the distribution of impacts of the heavy-duty vehicle and engine standards. After consideration of comments, we have retained the demographic analysis from the proposal based upon the data and methods described above. However, in response to comments that the Agency consider the disparate impacts of the rule through the analysis of race/ethnicity-stratified impacts, we have added an analysis of the demographic composition of population-weighted national average air quality impacts. For scenarios with and without the rule in place, we present national average air quality

[BB] County-level poverty status was mapped to the 12km x 12km grid cell domain using spatial weighting in BenMAP-CE.

concentrations that are weighted by specific race and ethnicity populations. Using the same air quality and population data described above, we sum the product of each projected CMAQ grid-cell population and its corresponding CMAQ grid-cell air quality concentration and then divide by the total population by race/ethnicity. As described in Section 6.4.9.3, we caution that the population projection data by race and ethnicity is uncertain and that the spatial resolution of the air quality modeling is not sufficient to capture local heterogeneity of human exposures.

6.4.9.2 Results

Of the approximately 48,000 populated CMAQ grid cells that encompass the contiguous United States, nearly 2,400 are in the highest 5 percent of the baseline distribution. For $PM_{2.5}$, the concentration at the 95th percentile is 7.76 $\mu g/m^3$ (median: 5.18 $\mu g/m^3$), and for ozone it is 49.91 ppb (median: 38.34 ppb). In 2045, 144 million people are projected to live within the highest 5 percent of grid cells for $PM_{2.5}$ and 39 million are projected to live in areas with the highest concentrations of ozone (Figure 6-18).

As shown in Table 6-7, in 2045, the number of people of color projected to live within the grid cells with the highest baseline concentrations of ozone (26 million) is nearly double that of NH-Whites (14 million). Thirteen percent of people of color are projected to live in areas with the worst baseline ozone, compared to seven percent of NH-Whites. The rule will reduce human exposures to ambient ozone for all population groups, but the 39 million people living in areas with the worst air quality will experience a greater reduction in ozone than the 370 million people living in the remaining 95 percent of grid cells.

Table 6-7: Demographic Analysis of Projected 2045 Ozone Concentrations (ppb), Sorted by Average Baseline Ozone Air Quality: NH-White and People of Color

			Seasonal Average Ozone Concentrations in ppb (5% to 95% Range)		
	2045 Population - millions		Baseline	Control	Reduction
All 12km x 12km Grid Cells in CONUS (n=47,795)	Total Population in CONUS – All Grid Cells	409	39.18 (29.91 - 49.91)	38.71 (29.42 -49.50)	0.47 (0.16 - 0.90)
	Non-Hispanic White Population in CONUS	208			
	People of Color Population in CONUS	201			
Highest 5% of Baseline Ozone Concentrations (n=2,391)	Population in Highest 5% (% of Total in CONUS)	39 (10%)	52.59 (50.01 - 58.20)	51.95 (49.57 - 57.22)	0.64 (0.27 - 1.44)
	Non-Hispanic White (% of NH-W in CONUS)	14 (7%)			
	People of Color (% of POC in CONUS)	26 (13%)			
Remaining 95% of Baseline Ozone Concentrations (n=45,404)	Population in Remaining 95% (% of Total in CONUS)	370 (90%)	38.47 (29.82 - 48.70)	38.01 (29.34 - 48.31)	0.46 (0.16 - 0.87)
	Non-Hispanic White (% of NH-W in CONUS)	194 (93%)			
	People of Color (% of POC in CONUS)	176 (86%)			

$PM_{2.5}$ results have a similar pattern to what we observe for ozone. As shown in Table 6-8, in 2045, the number of people of color projected to live within the grid cells with the highest baseline concentrations of $PM_{2.5}$ (93 million) is nearly double that of NH-Whites (51 million). Forty-six percent of people of color are projected to live in areas with the worst baseline $PM_{2.5}$, compared to 25 percent of NH-Whites. Those in areas with the worst air quality will experience a greater reduction in $PM_{2.5}$ than those in the remaining 95 percent of grid cells.

Table 6-8: Demographic Analysis of Projected 2045 $PM_{2.5}$ Concentrations (μg/m^3), Sorted by Average $PM_{2.5}$ Baseline Air Quality: NH-White and People of Color

			Annual Average $PM_{2.5}$ Concentrations in μg/m^3 (5% to 95% Range)		
	2045 Population - millions		Baseline	Control	Reduction
All 12km x 12km Grid Cells in CONUS (n=47,795)	Total Population in CONUS – All Grid Cells	409	5.23 (2.65 - 7.76)	5.21 (2.65 - 7.72)	0.022 (0.003 - 0.052)
	Non-Hispanic White Population in CONUS	208			
	People of Color Population in CONUS	201			
Highest 5% of Baseline $PM_{2.5}$ Concentrations (n=2,391)	Population in Highest 5% (% of Total in CONUS)	144 (35%)	9.03 (7.80 - 12.07)	8.99 (7.76 - 12.01)	0.044 (0.008 -0.097)
	Non-Hispanic White (% of NH-W in CONUS)	51 (25%)			
	People of Color (% of POC in CONUS)	93 (46%)			
Remaining 95% of Baseline $PM_{2.5}$ Concentrations (n=45,404)	Population in Remaining 95% (% of Total in CONUS)	265 (65%)	5.03 (2.62 - 7.24)	5.01 (2.62 - 7.20)	0.020 (0.003 - 0.049)
	Non-Hispanic White (% of NH-W in CONUS)	156 (75%)			
	People of Color (% of POC CONUS)	108 (54%)			

In Table 6-9 and Table 6-10, we looked at populations above and below 200% of the federal poverty line. Using 2045 population estimates, 126 million people are projected to live below 200% of the poverty line, 13 million (10 percent) of whom will also be living in an area with the worst baseline concentrations of ozone. Similarly, 10 percent of people projected to live above 200% of the poverty line will also be living in an area with the worst baseline concentrations of ozone. For $PM_{2.5}$, 37 percent of those living below 200% of the poverty line will also be living in areas with the worst baseline concentrations of $PM_{2.5}$, compared to 35 percent of the population above 200% of the poverty line projected to live in those same areas. While some disparity exists for $PM_{2.5}$, overall, the demographic results for those living above and below 200% of the poverty line are not as pronounced in the areas with the worst air quality as they are for race and ethnicity.

Table 6-9: Demographic Analysis of Projected 2045 Ozone Concentrations (ppb), Sorted by Average Baseline Ozone Air Quality: Poverty Status

			Seasonal Average Ozone Concentrations in ppb (5% to 95% Range)		
	2045 Population - millions		Baseline	Control	Reduction
All 12km x 12km Grid Cells in CONUS (n=47,795)	Total Population in CONUS – All Grid Cells	409	39.18 (29.91 - 49.91)	38.71 (29.42 -49.50)	0.47 (0.16 - 0.90)
	Population Below 200% of the Poverty Line in CONUS [a]	126			
	Population Above 200% of the Poverty Line in CONUS	283			
Highest 5% of Baseline Ozone Concentrations (n=2,391)	Population in Highest 5% (% of Total in CONUS)	39 (10%)	52.59 (50.01 - 58.20)	51.95 (49.57 - 57.22)	0.64 (0.27 - 1.44)
	Population Below 200% of the Poverty Line (% Below 200% in CONUS)	13 (10%)			
	Population Above 200% of the Poverty Line (% Below 200% in CONUS)	26 (10%)			
Remaining 95% of Baseline Ozone Concentrations (n=45,404)	Population in Lowest 95% (% of Total in CONUS)	370 (90%)	38.47 (29.82 - 48.70)	38.01 (29.34 - 48.31)	0.46 (0.16 - 0.87)
	Population Below 200% of the Poverty Line (% Below 200% in CONUS)	112 (90%)			
	Population Above 200% of the Poverty Line (% Below 200% in CONUS)	257 (90%)			

[a] Note that the poverty measure used here is based on ACS 5-year estimates from 2015 to 2019 at the county level representing the fraction of county-level population below and above 200% of the poverty line. Counts of 2045 population reflect projections based on 2010 Census Data and population growth factors estimated by Woods & Poole (2015). Measures of "current" poverty are not necessarily predictors of future poverty status.

Table 6-10: Demographic Analysis of Projected 2045 $PM_{2.5}$ Concentrations (µg/m^3), Sorted by Average $PM_{2.5}$ Baseline Air Quality: Poverty Status

			Annual Average $PM_{2.5}$ Concentrations in µg/m^3 (5% to 95% Range)		
	2045 Population - millions		Baseline	Control	Reduction
All 12km x 12km Grid Cells in CONUS (n=47,795)	Total Population in CONUS – All Grid Cells	409	5.23 (2.65 - 7.76)	5.21 (2.65 - 7.72)	0.022 (0.003 - 0.052)
	Population Below 200% of the Poverty Line in CONUS [a]	126			
	Population Above 200% of the Poverty Line in CONUS	283			
Highest 5% of Baseline $PM_{2.5}$ Concentrations (n=2,391)	Population in Highest 5% (% of Total in CONUS)	144 (35%)	9.03 (7.80 - 12.07)	8.99 (7.76 - 12.01)	0.044 (0.008 -0.097)
	Population Below 200% of the Poverty Line (% Below 200% in CONUS)	46 (37%)			
	Population Above 200% of the Poverty Line (% Below 200% in CONUS)	98 (35%)			
Remaining 95% of Baseline $PM_{2.5}$ Concentrations (n=45,404)	Population in Lowest 95% (% of Total in CONUS)	265 (65%)	5.03 (2.62 - 7.24)	5.01 (2.62 - 7.20)	0.020 (0.003 - 0.049)
	Population Below 200% of the Poverty Line (% Below 200% in CONUS)	79 (63%)			
	Population Above 200% of the Poverty Line (% Below 200% in CONUS)	185 (65%)			

[a] Note that the poverty measure used here is based on ACS 5-year estimates from 2015 to 2019 at the county level representing the fraction of county-level population below and above 200% of the poverty line. Counts of 2045 population reflect projections based on 2010 Census Data and population growth factors estimated by Woods & Poole (2015). Measures of "current" poverty are not necessarily predictors of future poverty status.

While the demographic analyses in Table 6-7 and Table 6-8 demonstrate the possible disparity that exists between NH-Whites and people of color, we have expanded the analysis of air quality impacts experienced by specific race and ethnic groups. In Table 6-11, we present the national population-weighted average ozone concentrations for each specific race and ethnicity category in scenarios without (baseline) and with (control) the rule in place. We also present the reduction in ozone (from baseline to control) for each race and ethnicity category along with the relative reduction from baseline expressed as a percentage. To highlight the changes in each category, results are color-coded by air quality (concentrations increase from light blue to dark blue) and by air quality improvements (reductions increase from light green to dark green).

On a population-weighted basis, all race and ethnicity population categories experience reductions in exposure to ozone as a result of the rule. NH-Black populations experience the lowest concentrations of ozone (both with and without the rule in place), while also experiencing the greatest reductions. NH-Native Americans are projected to live in areas with the highest ozone concentrations while also experiencing slightly smaller reductions from this rule compared to other race and ethnicity populations. In relative terms, the percent reduction in ozone experienced by each race and ethnicity category ranges from a 1.33% reduction from baseline (NH-Native American) to a 1.92% reduction from baseline (NH-Black).

In Table 6-12, we present the national population-weighted average $PM_{2.5}$ concentrations for each specific race and ethnicity category using the same color-coding scheme described for Table 6-11. On a population-weighted basis, all race and ethnicity population categories experience reductions in exposure to $PM_{2.5}$ as a result of the rule. The largest reductions in $PM_{2.5}$ are projected to occur in areas where NH-Black populations reside. NH-Native Americans experience the lowest concentrations of $PM_{2.5}$ (both with and without the rule in place) and are projected to receive slightly smaller reductions from this rule compared to other race and ethnicity populations. Hispanic populations are projected to experience the highest $PM_{2.5}$ concentrations in both the baseline and control scenarios. In relative terms, the percent reduction in $PM_{2.5}$ experienced by each race and ethnicity category ranges from a 0.40% reduction from baseline (Hispanic, NH-Asian, NH-Native American) to a 0.52% reduction from baseline (NH-Black).

Table 6-11: Demographic Analysis of National Average 2045 Ozone Concentrations (ppb), by Race/Ethnicity[a]

	2045 Population (million)	Baseline Ozone Concentration	Control Ozone Concentration	Reduction in Ozone	% Reduction in Ozone
All Race/Ethnicity	409	39.30	38.66	0.64	1.64%
NH-White	208	38.61	37.94	0.67	1.72%
Hispanic	108	41.10	40.51	0.60	1.45%
NH-Black	55	37.55	36.83	0.72	1.92%
NH-Asian	35	40.46	39.91	0.56	1.38%
NH-Native American	3	41.34	40.78	0.55	1.33%

[a] National averages are weighted by population. We sum the product of each projected CMAQ grid-cell population and its corresponding CMAQ grid-cell air quality concentration and then divide by the total population.

Table 6-12: Demographic Analysis of National Average 2045 $PM_{2.5}$ Concentrations ($\mu g/m^3$), by Race/Ethnicity[a]

	2045 Population (million)	Baseline $PM_{2.5}$ Concentration	Control $PM_{2.5}$ Concentration	Reduction in $PM_{2.5}$	% Reduction in $PM_{2.5}$
All Race/Ethnicity	409	7.26	7.23	0.034	0.47%
NH-White	208	6.83	6.79	0.035	0.51%
Hispanic	108	7.90	7.87	0.031	0.40%
NH-Black	55	7.50	7.41	0.039	0.52%
NH-Asian	35	7.64	7.61	0.030	0.40%
NH-Native American	3	6.18	6.15	0.025	0.40%

[a] National averages are weighted by population. We sum the product of each projected CMAQ grid-cell population and its corresponding CMAQ grid-cell air quality concentration and then divide by the total population.

6.4.9.3 Uncertainty in the Demographic Analysis

The results of this demographic analysis are dependent on the available input data and its associated uncertainty. As we note in both the air quality modeling and health benefits chapters, uncertainties exist along the entire pathway from emissions to air quality to population projections and exposure. The demographic analysis (including poverty status) is subject to these same sources of uncertainty.

A limitation of this analysis is the 12km x 12km horizontal grid spacing of the air quality modeling domain. Such resolution is unable to capture the heterogeneity of human exposures to pollutants within that area, especially pollutant concentration gradients that exist near roads. EPA is considering how to better estimate the near-roadway air quality impacts of its regulatory actions and how those impacts are distributed across populations. Because the heavy-duty standards apply nationally and will be implemented consistently across roadways throughout the U.S., we can still make useful observations of demographic trends at a national scale using the air quality modeling data at a 12km x 12km resolution.

Another key source of uncertainty is the accuracy of the projected baseline concentrations of $PM_{2.5}$ and ozone because we use modeled 2045 baseline air quality as the basis for our comparisons. Assumptions that influence projections of future air quality include emissions in the future baseline (stationary source emissions are only projected out to year 2028 and held constant out to 2045) and the meteorology used to model air quality (2016 conditions). With this uncertainty in mind, we prefer to examine the air quality impacts of the standards by comparing the baseline scenario to the control scenario in order to highlight incremental changes in air quality due to the standards. By looking at the incremental change, any underlying uncertainty present in both the modeled baseline and control air quality data is largely offset. However, when we rank grid cells from dirtiest to cleanest using 2045 baseline concentrations, the uncertainties associated with the baseline take on greater importance when interpreting the results of the analysis.

There is also inherent uncertainty in the Woods & Poole-based populations projected out to 2045. As mentioned above, the population projections are based on proprietary economic forecasting models developed by Woods and Poole in 2015 and are relative to a baseline using the 2010 Census data. Underlying the population projections are forecasted variables such as income, employment, and population. Each of these forecasts require many assumptions: economy-wide modeling to project income and employment, net migration rates based on employment opportunities and taking into account fertility and mortality, and the estimation of age/sex/race distributions at the county-level based on historical rates of mortality, fertility, and migration. To the extent these patterns and assumptions have changed since the population projections were estimated, and to the extent that these patterns and assumptions may change in the future, we would expect the projections of future population would be different than those used in this analysis.

For the analysis of exposure trends organized by baseline concentration, we attempted to address some of this population projection uncertainty by compiling race and ethnicity into two broad categories, “people of color” and “NH-White.” Such broad groupings help avoid overly precise interpretations of inherently uncertain projections of population and demographics, especially when looking at areas of worst air quality against the remaining areas across the contiguous United States. EPA continues to investigate how best to incorporate population projections into our analyses to disaggregate populations of concern by relevant socioeconomic variables, and to identify the interactions between demographic changes and air quality changes. In response to this commitment, while acknowledging the uncertainty in future population projections by specific race and ethnicity category, we have expanded the demographic analysis of air quality impacts to include specific race and ethnicity groups. Tables 6-11 and 6-12 report air quality exposure trends by race and ethnicity at the national level.

The measure of poverty status used in this analysis is based on data from the American Community Survey representing the rate of poverty between 2015-2019 and is not projected to reflect poverty status in the future. This assumption is inherently uncertain since measures of “current” poverty are not necessarily predictors of future poverty status.

We intend to continue to refine demographic analyses in future rulemakings including potentially assessing how much of the results may or may not be driven by emissions changes compared to projected changes in demographics.

Finally, we note that the air quality scenario we modeled to support the air quality, benefits, and demographic analyses for this rulemaking is based on modeling conducted for the proposal. Despite the differences between the modeled scenario and the final standards, the emission reductions used in the air quality modeling analysis for the proposed rule are similar to those associated with the final standards, see Chapter 5.4. Both scenarios result in reductions in emissions of VOC and $PM_{2.5}$ and large reductions in emissions of NO_X, and we expect that the final rule will also lead to substantial improvements in air quality.

Chapter 6 References

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Chapter 7 Program Costs

In this chapter, EPA presents estimates of the costs associated with the emissions-reduction technologies that manufacturers could add in response to the final standards. We present these not only in terms of the upfront technology costs per engine as presented in Chapter 3 of this RIA, but also how those costs will change in the years following implementation. We also present the costs associated with the final regulatory useful life provisions that correspond to those standards, as well as costs associated with the warranty provisions. These technology costs are presented in terms of direct manufacturing costs and associated indirect costs--i.e., research and development (R&D), administrative costs, marketing, and other costs of running a company. We term the sum of these direct and indirect costs "technology costs" or "technology package costs." They represent the costs incurred by manufacturers--i.e., regulated entities--to comply with the final program.[A] The analysis also includes estimates of the possible operating costs associated with the final program. These operating costs represent estimated costs incurred by users of MY 2027 and later heavy-duty vehicles.[B] All costs are presented in 2017 dollars unless noted otherwise.

The costs presented here are grouped into three main categories, as described below:

- Technology Package Costs: these are the direct costs of new or modified technology—that EPA projects manufacturers will add—and the associated indirect costs that will be involved with bringing those technologies to market (research, development, warranty, etc.). In our analysis, these costs are expected to be incurred by manufacturers of new HD engines and vehicles. "Direct" costs represent the direct manufacturing costs of the technologies we expect to be used to comply with the final standards over the final useful lives. We use those costs to estimate the year-over-year manufacturing costs going forward from the first year of implementation. "Indirect" costs include the indirect costs of the technologies we expect to be used to comply with the final standards, in part due to the useful life provisions. Indirect costs also include costs expected under the final program due to the warranty provisions.

- Operating Costs: these are the costs associated with the truck and bus operation that are projected to be impacted by the final program. For example, costs associated with tire replacement are not included since the final standards are not expected to impact tire replacement, but costs associated with repair of the more costly emission-related components are included. These costs are estimated to be incurred by purchasers/owners of new MY 2027 HD vehicles.

[A] More precisely, these technology costs represent costs that manufacturers are expected to attempt to recapture via new vehicle sales. As such, profits are included in the indirect cost calculation. Clearly, profits are not a "cost" of compliance--EPA is not imposing new regulations to force manufacturers to make a profit. However, profits are necessary for manufacturers in the heavy-duty industry, a competitive for-profit industry, to sustain their operations. As such, manufacturers are expected to make a profit on the compliant vehicles they sell, and we include those profits in estimating technology costs.

[B] Importantly, the final standards, useful lives, and warranty periods apply only to new, MY 2027 and later heavy-duty vehicles. The legacy fleet is not subject to the new requirements and, therefore, users of prior model year vehicles will not incur the operating costs we estimate.

- Program Costs: these are the new technology package costs and operating costs combined (the sum of numbers 1 and 2, above). These costs represent our best estimate of the costs to society. As such, any taxes (e.g., fuel taxes) are excluded since taxes represent a transfer payment from one member of society to another with no net cost to society. Total program costs under the final program are presented in terms of calendar year 2045 costs, present value costs, and annualized costs (see Table 7-51 and Table 7-52).[C]

The cost analysis is done using a tool written in Python and contained in the docket. The Python tool along with some documentation is contained in the docket to this rule and on our website.[1]

7.1 Technology Package Costs

As noted, individual technology piece costs were presented in Chapter 3. Those costs are, in general, the direct manufacturing costs (DMC) estimated for the first year of implementation of the final standards for the final useful lives. Those costs are used here as a starting point in estimating program costs. Following the year in which costs are first incurred for each phase, we have applied a learning effect to represent the cost reductions expected to occur via the "learning by doing" phenomenon.[2] This provides a year-over-year cost for each technology as applied to new engine sales. We have then applied industry standard "retail price equivalent (RPE)" markup factors industry-wide, with adjustments discussed below, to estimate indirect costs. Both the learning effects applied to direct costs and the application of markup factors to estimate indirect costs are consistent with the cost estimation approaches used in EPA's past transportation-related regulatory programs.[3] The sum of the direct and indirect costs represents our estimate of technology costs per vehicle on a year-over-year basis. These technology costs multiplied by estimated sales then represent the total technology costs associated with the final program.

This cost calculation approach presumes that the expected technologies will be purchased by original equipment manufacturers (OEMs) from their suppliers. So, while the DMC estimates include the indirect costs and profits incurred by the supplier, the indirect cost markups we apply cover the indirect costs incurred by OEMs to incorporate the new technologies into their vehicles and to cover profit margins typical of the heavy-duty truck industry. We discuss the indirect costs markups in more detail in Section 7.1.2.

These technology package costs (both direct and indirect), while first incurred by manufacturers of new engines, are presumed to be passed on to the consumers of those engines (i.e., heavy-duty truck makers and, ultimately, their purchasers/owners).

Note that, throughout this discussion of costs we use the term regulatory class which is roughly equivalent to a service class; we use the term regulatory class for consistency with our MOVES model and its classification system so that our costs align with our inventory estimates and the associated benefits discussed in Chapters 5 and 8.

[C] The costs presented in Table 7-98 and Table 7-99 are presented again in Table ES-2,which summarizes the net benefits of the final standards.

7.1.1 **Direct Manufacturing Costs**

To produce a unit of output, manufacturers incur direct and indirect costs. Direct costs include cost of materials and labor costs to manufacture that unit. Indirect costs are discussed in the following section. The direct manufacturing costs presented here include individual technology costs for emission-related engine components and for exhaust aftertreatment systems (EAS).

Notably, for this analysis we include not only the marginal increased costs associated with the final program, but also the emission control system costs for the "no action" baseline case (Table 7-5 and Table 7-6).[D] Throughout this discussion we refer to baseline technology costs, or baseline costs, which are meant to reflect our cost estimate of engine systems--that portion that is emission-related--and the exhaust aftertreatment costs absent the impacts of final program. This inclusion of baseline system costs contrasts with EPA's approach in recent Greenhouse Gas rules or the light-duty Tier 3 criteria pollutant rule where we estimated costs relative to a "no action" baseline case, which obviated the need to estimate baseline costs. We have included baseline costs in this analysis because the final emissions warranty and regulatory useful life provisions are expected to have some impact on not only the new technology added to comply with the final program, but also on any existing emission control systems (see Chapter 2 for more details on the final Emissions Warranty and Regulatory Useful Life). The new warranty and useful life provisions will increase costs not only for the new technology added in response to the new standards, but also for the technology already in place (to which the new technology is added) because the new warranty and useful life provisions will apply to the entire emission-control system, not just the new technology added in response to the new standards. The baseline direct manufacturing costs detailed below are thus meant to reflect that portion of baseline case engine hardware and aftertreatment systems for which new indirect costs will be incurred due to the final warranty and useful life provisions, even absent any changes in the level of emission standards.

We have estimated the baseline engine costs based on recently completed studies by the International Council on Clean Technology (ICCT) as discussed in more detail below. The baseline EAS costs were presented in Chapter 3 of this RIA. The estimated marginal technology costs associated with the final standards were also presented in Chapter 3 of this RIA.

As noted, the costs shown in Table 7-5 and Table 7-6 include costs for the baseline case.[E] For the baseline diesel engine-related costs associated with emission control (i.e., a portion of the fuel system, the EGR system, etc.), we have relied on a white paper done by the International Council on Clean Transportation (ICCT) entitled, "Costs of Emission Reduction Technologies for Heavy-Duty Diesel Vehicles."[4] In Table 14 of that paper, ICCT presented technology costs to meet U.S. standards at different stages for a 12L engine. The different stages of U.S. standards were the 1998, 2004, 2007 and 2010 standards. Relevant portions of ICCT's Table 14—those portions associated with engine-related technologies—are shown in Table 7-1. For the fuel system and the turbo charger, ICCT shows only 50 percent of the total cost and states in the text

[D] See Chapter 5 for more information about the baseline and how that baseline is characterized. For this cost analysis, the baseline, or no action, case consists of MY 2019 engines and emission control systems. See also Section VI for more information about the emission inventory baseline and how that baseline is characterized. Why we include costs for the no action case is described in this section.
[E] See RIA Chapters 1.1 and 1.2 for more information on emission control technologies available on current, or baseline, engines.

that, for components that serve other purposes in addition to emission control (e.g., the fuel system delivers fuel to power the engine and the turbo charger serves to increase engine power in addition to their emission control functions), only 50% of the cost is considered in their analysis.[5] ICCT notes that their costs are likely conservative since they do not consider learning effects applied to the cost estimates associated with each regulatory stage. Lastly, ICCT notes that their cost estimates are stated in 2015 dollars.

Table 7-1: ICCT Cost Estimates of 12L Diesel Engine-Related Emission Control Costs Associated with Past US Emission Standards (2015 dollars)

Air/fuel control and engine out emissions	US 1998	US2004	US2007	US2010
Fuel system—50% of total cost	-	376	38	41
Variable Geometry Turbo (extra cost)—50% of total cost	-	-	185	-
Exhaust Gas Recirculation (EGR) system	-	439	-	-
EGR cooling	-	108	-	-
Total for air/fuel control and engine out emissions	-	923	223	41

In this analysis, EPA has made use of these ICCT cost estimates by first doubling the fuel system and turbo charger costs to get the full cost of those systems (i.e., to undo the halving of those costs done by ICCT). We then added to that result the EGR costs to get a total cost of $1,827. We have then scaled them based on engine displacement in a manner consistent with our approach to estimating exhaust aftertreatment costs (EAS, see Chapter 3 of this RIA). The engine displacements used in our EAS cost estimates were 7, 8 and 13 liters for light, medium and heavy heavy-duty engines, respectively. We have estimated the class 2b and 3 engines as equivalent to the light heavy-duty (7L) and the urban bus engines as equivalent to the medium-duty (8L) engines. We then adjusted the costs from the ICCT study's 2015 dollars to 2017 dollars consistent with the FRM analysis. The resultant diesel engine-related costs used in this analysis are shown in Table 7-2.

Table 7-2: Diesel Engine-Related Emission Control System Costs in the "No Action" Baseline*

	Class 2b3	Light HDE	Medium HDE	Heavy HDE	Urban bus
Engine displacement	7	7	8	13	8
Displacement based scalar	7/12=0.58	7/12=0.58	8/12=0.67	13/12=1.08	8/12=0.67
Baseline cost, 2015 dollars (1,827 times Displacement based scalar)	$1,066	$1,066	$1,218	$1,979	$1,218
Baseline cost, 2017 dollars (1.03 GPD deflator Baseline cost in 2015 dollars)*	$1,097	$1,097	$1,254	$2,038	$1,254

* See Table 7-8 and associated text for information on the GDP deflators used in this analysis; costs shown are by MOVES regulatory class; there are a small number of diesel engines used in LHD2b3 that are engine (rather than chassis) certified and are, therefore, expected to incur costs associated with the final rule.

For the baseline gasoline engine-related costs associated with emission control (i.e., a portion of the fuel system, etc.), we have relied on a white paper done by ICCT entitled, "Estimated Cost of Emission Reduction Technologies for Light-Duty Vehicles."[6] In Table 4-10 of that paper, ICCT presented technology costs to meet U.S. light-duty Tier 2, Bin 5 for a 4.5L engine. The ICCT estimate shown was $306. ICCT notes that their cost estimates are stated in 2011 dollars.

In this analysis, EPA has made use of this ICCT cost estimate by scaling them based on engine displacement from the 4.5L light-duty displacement assumed by ICCT to a more typical gasoline HD engine displacement of 7L. The resultant gasoline engine-related costs used in this analysis are shown in Table 7-3.

Table 7-3: Gasoline Engine-Related Emission Control System Costs in the "No Action" Baseline*

	Light HDE	Medium HDE	Heavy HDE
Engine displacement	7	7	7
Displacement based scalar	7/4.5=1.56	7/4.5=1.56	7/4.5=1.56
Baseline cost, 2011 dollars (306 times Displacement based scalar)	$476	$476	$476
Baseline cost, 2017 dollars (1.099 GPD deflator Baseline cost in 2011 dollars)*	$523	$523	$523

* See Table 7-8 and associated text for information on the GDP deflators used in this analysis; note that there are no engine certified LHD2b3 gasoline engines and, therefore, none are expected to incur costs associated with this final rule.

For the baseline CNG engine-related costs associated with emission control (a portion of the fuel system, etc.), we have relied on the ICCT baseline gasoline costs presented in Table 7-3, but have scaled those costs based on more typical diesel engine displacements because CNG engines tend to be converted diesel engines but with fuel systems more typical of gasoline engines. The diesel engine displacements used for scaling gasoline costs were presented in Table 7-2. The resultant baseline engine-related CNG costs are shown in Table 7-4.

Table 7-4: CNG Engine-Related Emission Control System Costs in the "No Action" Baseline*

	Heavy HDE	Urban bus
Engine displacement	12	9
Displacement based scalar	12/4.5=2.67	8/4.5=2.0
Baseline cost, 2011 dollars (306 times Displacement based scalar)	$816	$612
Baseline cost, 2017 dollars (1.099 GPD deflator * Baseline cost in 2011 dollars)*	$896	$672

* See Table 7-8 and associated text for information on the GDP deflators used in this analysis.

For cylinder deactivation costs under the final standards, we have used FEV-conducted teardown-based cylinder deactivation costs as presented in Chapter 3 of this RIA.[7] The marginal technology costs for exhaust aftertreatment components--also detailed and presented in Chapter 3 of this RIA--are updated relative to the proposal. In the proposal, we used an ICCT methodology with extensive revision by EPA. In this final analysis, the exhaust aftertreatment costs are based on FEV-conducted teardown-based costs.[8]

The cost basis (the year dollars) for many of these costs were also presented in a mixed set of cost basis years. For the cost analysis presented here, we use 2017 dollars throughout the analysis for consistency with the proposal which used 2017 dollars. The costs presented in Chapter 3 are repeated in Table 7-5 for diesel regulatory classes, in Table 7-6 for gasoline regulatory classes, and in **Error! Reference source not found.** for CNG regulatory classes with the exception that all values presented here are updated to a consistent 2017 dollar basis.[F] See Chapter 5.2 for a discussion of regulatory classes. Table 7-8 shows the gross-domestic product price deflators used to adjust to 2017 dollars. Note that we have estimated costs for regulatory classes that exist in our MOVES runs (see Chapter 5 of this RIA) to remain consistent with the inventory impacts we have estimated. Note also that, throughout this section, we use several acronyms, including heavy-duty engine (HDE, exhaust aftertreatment system (EAS), and compressed natural gas (CNG).

Table 7-5: Diesel Technology and Package Direct Manufacturing Costs per Engine by Regulatory Class for the Baseline and Final Program, 2017 dollars

MOVES Regulatory Class	Technology	Baseline	Final Program (MY2027 increment to Baseline)
Class 2b3	**Package**	3,681	1,920
	Engine hardware	1,097	0
	Closed crankcase	0	0
	Cylinder deactivation	0	196
	EAS	2,585	1,724
Light HDE	**Package**	3,699	1,957
	Engine hardware	1,097	0
	Closed crankcase	18	37
	Cylinder deactivation	0	196
	EAS	2,585	1,724
Medium HDE	**Package**	3,808	1,817
	Engine hardware	1,254	0
	Closed crankcase	18	37
	Cylinder deactivation	0	147
	EAS	2,536	1,634
Heavy HDE	**Package**	5,816	2,316
	Engine hardware	2,037	0
	Closed crankcase	18	37
	Cylinder deactivation	0	206
	EAS	3,761	2,074
Urban bus	**Package**	3,884	1,850
	Engine hardware	1,254	0
	Closed crankcase	18	37
	Cylinder deactivation	0	147
	EAS	2,613	1,666

[F] The MY 2019 engine and aftertreatment costs estimates presented in RIA Chapters 3.1.5 and 3.2.3 are used as the MY 2027 baseline cost in the tables in this RIA Chapter 7.1.1.

Table 7-6: Gasoline Technology and Package Direct Manufacturing Costs per Engine by Regulatory Class for the Baseline and Final Program, 2017 dollars

MOVES Regulatory Class	Technology	Baseline	Final Program (MY2027 increment to Baseline)
Light HDE	**Package**	2,681	688
	Engine hardware	522	0
	Aftertreatment	2,158	664
	ORVR	0	24
Medium HDE	**Package**	2,681	688
	Engine hardware	522	0
	Aftertreatment	2,158	664
	ORVR	0	24
Heavy HDE	**Package**	2,681	688
	Engine hardware	522	0
	Aftertreatment	2,158	664
	ORVR	0	24

Table 7-7: CNG Technology and Package Direct Manufacturing Costs per Engine by Regulatory Class for the Baseline and Final Program, 2017 dollars

MOVES Regulatory Class	Technology	Baseline	Final Program (MY2027 increment to Baseline)
Heavy HDE	**Package**	8,585	25
	Engine hardware	896	0
	Aftertreatment	7,689	25
Urban bus	**Package**	6,438	19
	Engine hardware	672	0
	Aftertreatment	5,766	19

Table 7-8: GDP Price Deflators* Used to Adjust Costs to 2017 Dollars

Cost Basis Year	Conversion Factor
2011	1.098
2015	1.029
2017	1.000
2018	0.977
2019	0.960

* Based on the National Income and Product Accounts, Table 1.1.9 Implicit Price Deflators for Gross Domestic Product, Bureau of Economic Analysis, U.S. Department of Commerce, April 28, 2022.

The direct costs are then adjusted to account for learning effects going forward from the first year of implementation. To make that adjustment, the following equation is used.[9]

$$y_{t+1} = \left(\frac{x_{t+1}}{x_t}\right)^b y_t$$

Where,
y_t = cost (or price) of a given item at time t
x_t = cumulative production of a given item at time t
x_{t+1} = cumulative production of a given item at time t+1

b = the learning rate

In this cost analysis, EPA makes an adjustment to this standard formula by inserting a "seed volume factor" meant to represent the number of years of sales of a technology leading to learning prior to the year for which the technology's cost estimate is intended (model year 2027 as indicated in Table 7-5, Table 7-6, and **Error! Reference source not found.**). In other words, manufacturers may sell some of the systems expected for compliance with the final standards in years prior to the first year of the new standards, thereby learning and realizing cost reductions in that time. The value of this seed volume factor might be 0 to represent a new, unsold technology, or any value >0 to represent a relatively new technology having had some sales in prior years. A seed volume factor of 0 places the technology at the beginning of the learning curve and, therefore, subsequent learning effects (i.e., cost reductions) will be most rapid in the years immediately following the first year of implementation. An increasing seed volume factor serves to move the effects of learning further along the curve making subsequent learning effects less dramatic in the years immediately following the first year of the analysis. Figure 7-1 shows the effect of the seed volume factor on the levels of learning applied to the direct manufacturing costs assuming constant sales year-over-year and a learning rate of -0.245.[G]

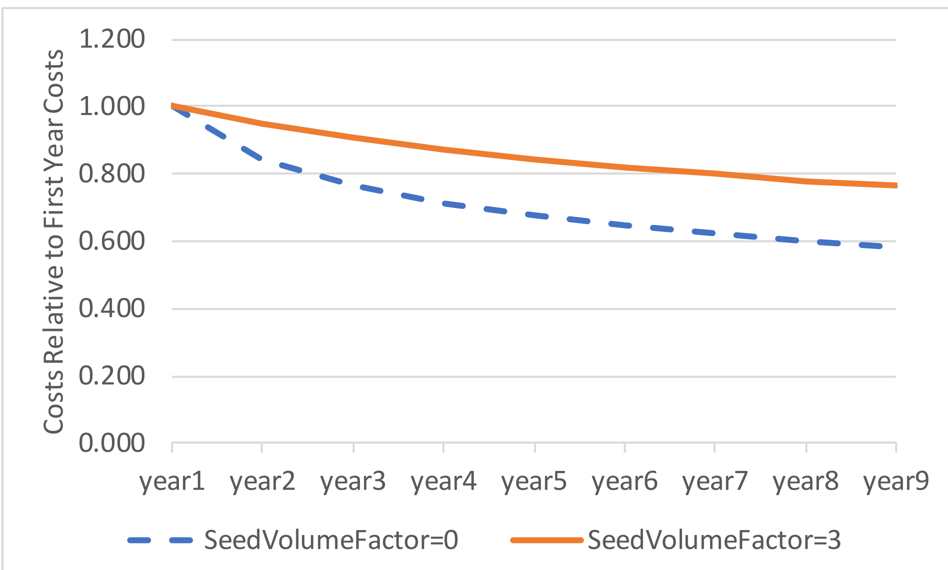

Figure 7-1: Costs Relative to First Year Costs using Different Seed Volume Factors

In the end, the learning effects are calculated using the following formula.

[G] In effect, the "seed volume factor" sets the cumulative number of units produced by an organization, which is a required data point to estimate the conventional form of the learning curve. Throughout this analysis, we have used a learning rate of -0.245 as developed for EPA by ICF and a foremost Subject Matter Expert, see "Cost Reduction through Learning in Manufacturing Industries and in the Manufacture of Mobile Sources, Final Report and Peer Review Report," EPA-420-R-16-018, November 2016.

$$y_t = \left(\frac{x_t + (Sales_{t=0} * SeedVolumeFactor)}{Sales_{t=0} + (Sales_{t=0} * SeedVolumeFactor)}\right)^b y_{t=0}$$

Where,
b = the learning rate (-0.245 in this analysis)
$y_{t=0}$ = estimated direct cost in the first year of implementation (e.g., MY2027 or MY2030)
$Sales_{t=0}$ = sales in the first year of implementation (e.g., MY2027 or MY2030)
SeedVolumeFactor = 0 or greater to represent the number of years of learning already having occurred on a technology
x_t = the cumulative sales of vehicles complying with the new standard (equal to first year sales in the first year of implementation, first year sales plus second year sales in the second year of implementation, etc.)

To illustrate the seed volume factor, if we assume the learning rate, b, is -0.245 (the value used in this analysis), the direct cost in the first year of implementation, $y_{t=0}$, is \$100, the sales in the first year of implementation, $Sales_{t=0}$, is 1000 engines, and the seed volume factor is 0 (i.e., no learning having occurred prior to the first year of the analysis), then the cost, y_t, would be \$100. This is because the cumulative production in year t, x_t, equals $Sales_{t=0}$ in the first year of implementation leaving the formula as:

$$(1)^{-0.245} * \$100 = \$100$$

If the sales in the following year were an additional 1000 engines, the cost would decrease to \$84 since x_t would now be 2000 and the formula would be:

$$\left(\frac{2000}{1000}\right)^{-0.245} * \$100 = \$84$$

If the seed volume factor is set to 3 (i.e., to approximate 3 years of learning prior to the first year of the analysis, then the cost in 2027 would again be \$100 since the formula would be

$$\left(\frac{1000 + (3 * 1000)}{1000 + (3 * 1000)}\right)^{-0.245} * \$100 = \$100$$

However, in 2028, the formula would be

$$\left(\frac{2000 + (3 * 1000)}{1000 + (3 * 1000)}\right)^{-0.245} * \$100 = \$95$$

With 3 years of learning estimated to have already occurred, the costs going forward from the first year reduce at a slower pace than in the previous example. Therefore, increasing the seed volume factor results in less rapid learning effects going forward. The seed volume factors used in this analysis are shown in Table 7-9. A factor of 10 has been used for the baseline technologies since those technologies will have undergone considerable learning by MY 2027. We have used a factor of 3 for the final standards to reflect at least 3 years of sales with technologies very similar to those expected under the final standards thereby resulting in conservative learning-based cost reductions moving forward from MY 2027.

Table 7-9: Seed volume factors used in this analysis

Fuel	Regulatory Class	Baseline	Final Program (MY 2027 increment to Baseline)
Diesel	Class 2b3	10	3
	Light HDE	10	3
	Medium HDE	10	3
	Heavy HDE	10	3
	Urban Bus	10	3
Gasoline	Light HDE	10	3
	Medium HDE	10	3
	Heavy HDE	10	3
CNG	Heavy HDE	10	3
	Urban Bus	10	3

Learning factors were applied on a technology package cost basis, and MOVES projected sales volumes were used to determine first year sales ($Sales_{t=0}$) and cumulative sales (x_t). The resultant direct manufacturing costs and how those costs reduce over time are presented in both Chapter 7.1.3 (on a per vehicle basis) and in Chapter 7.3.1 (on a total cost basis). The resultant learning effects are shown in Table 7-10 for diesel HDE, Table 7-11 for gasoline HDE, and Table 7-12 for CNG HDE.

Table 7-10: Learning Effects Applied to Direct Manufacturing Costs for Diesel HDE

Model Year	Class 2b3		Light HDE		Medium HDE		Heavy HDE		Urban bus	
	Baseline	FRM	Baseline	FRM	Baseline	FRM	Baseline	FRM	Baseline	FRM
2027	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000
2028	0.979	0.947	0.979	0.946	0.979	0.946	0.979	0.947	0.979	0.947
2029	0.960	0.906	0.959	0.904	0.959	0.904	0.960	0.906	0.960	0.905
2030	0.943	0.873	0.941	0.869	0.942	0.870	0.942	0.872	0.942	0.871
2031	0.927	0.845	0.924	0.840	0.925	0.840	0.926	0.843	0.926	0.842
2032	0.913	0.821	0.909	0.814	0.909	0.815	0.911	0.818	0.910	0.817
2033	0.900	0.800	0.894	0.792	0.895	0.793	0.897	0.796	0.896	0.795
2034	0.887	0.781	0.880	0.771	0.881	0.773	0.884	0.777	0.883	0.775
2035	0.875	0.765	0.867	0.753	0.868	0.755	0.871	0.760	0.870	0.758
2036	0.864	0.750	0.854	0.737	0.856	0.738	0.860	0.744	0.858	0.742
2037	0.854	0.736	0.843	0.722	0.844	0.724	0.849	0.730	0.847	0.727
2038	0.844	0.724	0.831	0.708	0.833	0.710	0.838	0.716	0.836	0.714
2039	0.835	0.712	0.821	0.695	0.822	0.697	0.828	0.704	0.826	0.701
2040	0.826	0.701	0.810	0.683	0.812	0.685	0.819	0.693	0.816	0.690
2041	0.817	0.691	0.800	0.672	0.802	0.674	0.809	0.682	0.807	0.679
2042	0.809	0.682	0.791	0.661	0.793	0.664	0.801	0.672	0.798	0.669
2043	0.801	0.673	0.782	0.651	0.784	0.654	0.792	0.663	0.789	0.659
2044	0.794	0.664	0.773	0.642	0.776	0.645	0.784	0.654	0.781	0.650
2045	0.787	0.656	0.765	0.633	0.767	0.636	0.776	0.645	0.773	0.641

Table 7-11: Learning Effects Applied to Direct Manufacturing Costs for Gasoline HDE

Model Year	Light HDE		Medium HDE		Heavy HDE	
	Baseline	FRM	Baseline	FRM	Baseline	FRM
2027	1.000	1.000	1.000	1.000	1.000	1.000
2028	0.979	0.946	0.979	0.946	0.979	0.946
2029	0.959	0.904	0.959	0.904	0.959	0.904
2030	0.941	0.869	0.941	0.869	0.941	0.869
2031	0.924	0.840	0.924	0.840	0.924	0.840
2032	0.909	0.814	0.909	0.814	0.909	0.814
2033	0.894	0.792	0.894	0.792	0.894	0.792
2034	0.880	0.772	0.880	0.771	0.880	0.771
2035	0.867	0.753	0.867	0.753	0.867	0.753
2036	0.855	0.737	0.854	0.737	0.854	0.737
2037	0.843	0.722	0.843	0.722	0.843	0.722
2038	0.832	0.708	0.831	0.708	0.831	0.708
2039	0.821	0.695	0.820	0.695	0.820	0.695
2040	0.811	0.683	0.810	0.683	0.810	0.683
2041	0.801	0.672	0.800	0.672	0.800	0.672
2042	0.791	0.662	0.791	0.661	0.791	0.661
2043	0.782	0.652	0.782	0.651	0.782	0.651
2044	0.774	0.642	0.773	0.642	0.773	0.642
2045	0.765	0.633	0.765	0.633	0.765	0.633

Table 7-12: Learning Effects Applied to Direct Manufacturing Costs for CNG HDE

Model Year	Heavy HDE		Urban bus	
	Baseline	FRM	Baseline	FRM
2027	1.000	1.000	1.000	1.000
2028	0.979	0.947	0.979	0.947
2029	0.960	0.905	0.960	0.905
2030	0.942	0.870	0.942	0.871
2031	0.925	0.841	0.926	0.842
2032	0.910	0.816	0.910	0.817
2033	0.895	0.794	0.896	0.795
2034	0.882	0.774	0.883	0.775
2035	0.869	0.756	0.870	0.758
2036	0.857	0.740	0.858	0.742
2037	0.845	0.725	0.847	0.727
2038	0.834	0.711	0.836	0.714
2039	0.824	0.699	0.826	0.701
2040	0.814	0.687	0.816	0.690
2041	0.804	0.676	0.807	0.679
2042	0.795	0.666	0.798	0.669
2043	0.786	0.656	0.789	0.659
2044	0.778	0.647	0.781	0.650
2045	0.770	0.638	0.773	0.641

7.1.2 <u>Indirect Costs</u>

The indirect costs presented here are all the costs estimated to be incurred by manufacturers of new heavy-duty engines and vehicles associated with producing the unit of output that are not direct costs. For example, they may be related to production (such as research and development (R&D)), corporate operations (such as salaries, pensions, and health care costs for corporate staff), or selling (such as transportation, dealer support, and marketing). Indirect costs are generally recovered by allocating a share of the costs to each unit of good sold. Although it is possible to account for direct costs allocated to each unit of goods sold, it is more challenging to account for indirect costs allocated to a unit of goods sold. To ensure that regulatory analysis capture the changes in indirect costs, markup factors, which relate total indirect costs to total direct costs, have been developed and used by EPA and other stakeholders. These factors are often referred to as retail price equivalent (RPE) multipliers. RPE multipliers provide, at an aggregate level, the relative shares of revenues, where:

Revenue = Direct Costs + Indirect Costs

so that:

Revenue/Direct Costs = 1 + Indirect Costs/Direct Costs = RPE

and,

Indirect Costs = Direct Costs x (RPE - 1).

If the relationship between revenues and direct costs (i.e., RPE) can be shown to equal an average value over time, then an estimate of direct costs can be multiplied by that average value to estimate revenues, or total costs. Further, that difference between estimated revenues, or total costs, and estimated direct costs can be taken as the indirect costs. EPA has frequently used these multipliers[10] to predict the resultant impact on costs associated with manufacturers' responses to regulatory requirements and we are using that approach in this analysis.

Using RPE multipliers implicitly assumes that incremental changes in direct manufacturing costs produce common incremental changes in all indirect cost contributors as well as net income. In the past, EPA has expressed a concern with using the RPE multiplier for all technologies, because it is not likely that the indirect costs of vehicle modifications are the same for all technologies.[11] For example, less complex technologies could require fewer R&D efforts or less warranty coverage than more complex technologies. In addition, some simple technological adjustments may, for example, have no effect on the number of corporate personnel and the indirect costs attributable to those personnel. The use of RPEs, with their assumption that all technologies have the same proportion of indirect costs, is likely to overestimate the costs of less complex technologies and underestimate the costs of more complex technologies. EPA developed an alternative indirect cost methodology--the Indirect Cost Multiplier (ICM)--to address those concerns.[12]

The cost of different technologies was an important distinction in modeling efforts supporting EPA's greenhouse gas rulemakings (GHG) since a variety of GHG technologies were available to manufacturers and EPA wanted to project which combinations of technologies were most cost effective toward achieving compliance. For this final rule, we do not have that consideration since we are projecting the same technologies for all vehicles within a given regulatory class and are not attempting to project from a variety of technologies which are most cost effective toward achieving compliance. For that reason, EPA is using the RPE approach to estimate indirect costs in this analysis.

RPEs themselves are also inherently difficult to estimate because the accounting statements of manufacturers do not neatly categorize all cost elements as either direct or indirect costs. Hence, each researcher developing an RPE estimate must apply a certain amount of judgment to the allocation of the costs.[13]

This cost analysis estimates indirect costs by applying markup factors used in past rulemakings setting new greenhouse gas standards for heavy-duty trucks.[14] The markup factors are based on financial filings with the Securities and Exchange Commission for several engine and engine/truck manufacturers in the heavy-duty industry as detailed in a study done by RTI International for EPA.[15] The RPE factors developed by RTI for HD engine manufacturers, HD truck manufacturers and for the HD truck industry as a whole are shown in Table 7-13.[H] Also shown in Table 7-13 are the RPE factors developed by RTI for light-duty vehicle manufacturers.[16]

Table 7-13: Retail Price Equivalent Factors in the Heavy-Duty and Light-Duty Industries

Cost Contributor	HD Engine Manufacturer	HD Truck Manufacturer	HD Truck Industry	LD Vehicle Industry
Direct manufacturing cost	1.00	1.00	1.00	1.00
Warranty	0.02	0.04	0.03	0.03
R&D	0.04	0.05	0.05	0.05
Other (admin, retirement, health, etc.)	0.17	0.22	0.29	0.36
Profit (cost of capital)	0.05	0.05	0.05	0.06
RPE	1.28	1.36	1.42	1.50

For this analysis, EPA has based cost estimates for diesel and CNG regulatory classes on the HD Truck Industry values shown in Table 7-13.[I] Because most of the changes apply to engines, we first considered using the HD Engine Manufacturer values. However, the industry is becoming more vertically integrated and the costs we are trying to estimate are those that occur at the end purchaser, or retail, level. For that reason, we believe that the truck industry values represent the factors of most interest to this analysis. For gasoline regulatory classes, we have used the LD Vehicle Industry values shown in Table 7-13. We have chosen those values since they more closely represent the cost structure of manufacturers in that industry--Ford, General Motors, and Fiat Chrysler.

Of the cost contributors listed in Table 7-13, Warranty and R&D are the elements of indirect costs that the final requirements are expected to impact. As discussed in Chapter 2 of this RIA, EPA is lengthening the required warranty period, which we expect to increase the contribution of warranty costs to indirect costs. EPA is also lengthening the regulatory useful life, which we expect to result in increased R&D expenses as systems are developed to deal with the longer life

[H] The engine manufacturers included were Hino and Cummins; the truck manufacturers included were PACCAR, Navistar, Daimler and Volvo. Where gaps existed such as specific line items not reported by these companies due to differing accounting practices, data from the Heavy Duty Truck Manufacturers Industry Report by Supplier Relations LLC (2009) and Census (2009) data for Other Engine Equipment Manufacturing Industry (NAICS 333618) and Heavy Duty Truck Manufacturing Industry (NAICS 336120) were used to fill the gaps. This is detailed in the study report at Appendix A.1.

[I] Note that the report used the term “HD Truck” while EPA generally uses the term “HD vehicle;” they are equivalent when referring to this report.

during which compliance with standards will be required. We expect that the minor OBD-related R&D efforts as discussed in Section IV.C of the preamble will be done given the R&D estimated here. Profit is listed to highlight that profit is being considered and included in the analysis. All other indirect cost elements--those encapsulated by the "Other" category, including General and Administrative Costs, Retirement Costs, Healthcare Costs, and other overhead costs--as well as Profits, are expected to scale according to their historic levels of contribution.

As mentioned, Warranty and R&D are the elements of indirect costs that the final requirements are expected to impact. Warranty expenses are the costs that a business expects to incur, or has already incurred, for the repair or replacement of goods that it has sold. The total amount of warranty expense is limited by the warranty period that a business typically allows. After the warranty period for a product has expired, a business no longer incurs a warranty liability; thus, a longer warranty period results in a longer period of liability for a product. At the time of sale, companies are expected to set aside money in a warranty liability account to cover any potential future warranty claims. If and when warranty claims are made by customers, the warranty liability account is debited and a warranty claims account is credited to cover warranty claim expenses.

In the proposed analysis, to address the expected increased indirect cost contributions associated with warranty (increased funding of the warranty liability account) due to the proposed longer warranty requirements, we applied scaling factors commensurate with the changes in proposed Option 1 or Option 2 to the number of miles included in the warranty period (i.e., VMT-based scaling factors). Industry commenters took exception to this approach, arguing that it resulted in underestimated costs associated with warranty. To support their comments, the Truck and Engine Manufacturers Association (EMA) submitted data that showed costs associated with actual warranty claims for roughly 250,000 heavy heavy-duty vehicles. The chart included in the EMA comments is shown in Figure 7-2 and is in the public docket for this rule.

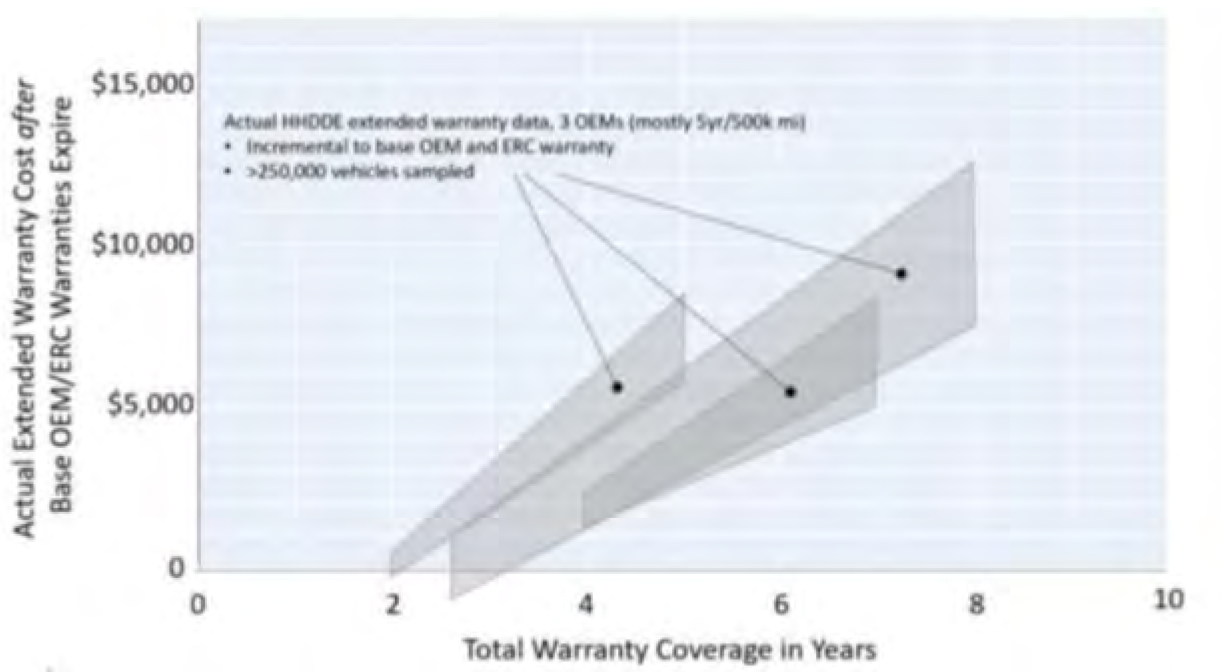

Figure 7-2: Warranty Costs Submitted as Part of the Comments from the Truck and Engine Manufacturers Association; see EPA-HQ-OAR-2019-0055-1203-A1, page 151

EPA considers this EMA comment and supporting information to be persuasive, not only because it represents data, but also because it represents data from three manufacturers and over 250,000 vehicles. However, the data are for heavy HDE, so it is not possible to determine an appropriate cost per year for light or medium HDE from the data directly. Also, the data represent actual warranty claims without any mention of the warranty claims rate (i.e., the share of engines sold that are making the warranty claims represented in the data). This latter issue makes it difficult to determine the costs that might be imposed on all new engines sold to cover the future warranty claims for the relatively smaller fraction of engines that incur warranty repair. In other words, if all heavy HDE purchases are helping to fund a warranty liability account, it is unclear if the $1,000 per year per engine is the right amount or if $1,000 per year is needed on only that percent of engines that will incur warranty repair. In the end, warranty costs imposed on new engine sales should be largely recouped by purchasers of those engines in the form of reduced emission repair expenses. EPA believes it is highly unlikely that any manufacturer would use their warranty program as a profit generator under the $1,000 per engine approach, especially in a market as competitive as the HD engine and vehicle industry. The possibility exists that the costs associated with the longer warranty coverage required by this rule will (1) converge towards those of the better performing OEMs; and (2) drop over time via something analogous to the learning by doing phenomenon described earlier. If true, we have probably overestimated the costs estimated here as attributable to this rule.

Thus, after careful consideration of these comments regarding warranty, and the engineering judgement of EPA subject matter experts, we revised our approach to estimating warranty costs, and for the final rule we have estimated warranty costs assuming a cost of $1,000 (2018 dollars or $977 in 2017 dollars) per estimated number of years of warranty coverage for a heavy heavy-duty diesel engine or heavy-duty vehicle equipped with such an engine. For other regulatory (engine) classes, we have scaled that value by the ratio of their estimated baseline emission-control system direct cost to the estimated emission-control system direct cost of the baseline heavy heavy-duty diesel engine. We use the baseline heavy heavy-duty diesel engine direct cost here because it should be consistent with the data behind the $1,000 per year value. The resulting warranty costs per year for a MY2027 HDE are as shown in Table 7-14. Importantly, these are emission-related warranty costs.

Table 7-14: Warranty Costs per Year of Estimated Warranty Coverage (2017 dollars)*

MOVES Regulatory Class	Scaling Approach	Diesel	Gasoline	CNG
Class 2b3	Base 2b3 DMC / Base Diesel Heavy HDE DMC	618		
Light HDE	Base Light HDE DMC / Base Diesel Heavy HDE DMC	621	450	
Medium HDE	Base Medium HDE DMC / Base Diesel Heavy HDE DMC	639	449	
Heavy HDE	Base Heavy HDE DMC / Base Diesel Heavy HDE DMC	977	448	1,442
Urban bus	Base Urban bus DMC / Base Diesel Heavy HDE DMC	652		1,081

* The Base Diesel HDE DMC would be the $5,816 value shown in Table 7-5.

As noted, we have used the estimated number of years of warranty coverage, not the regulated number of years. In other words, a long-haul tractor accumulating over 100,000 miles per year will reach any regulated warranty mileage prior to a refuse truck accumulating under 40,000 miles per year, assuming both are in the same regulatory class and, therefore, have the same warranty provisions. In all cases, we estimate the number of years of warranty coverage by determining the minimum number of years to reach either the regulated number of years, the regulated number of miles, or the regulated number of hours of operation. In making this estimation, whether for warranty or for useful life, we start with the required age/miles/hours and a typical number of miles driven per year. The typical miles driven is calculated as the average number of miles driven during the first 7 years of operation, according to our MOVES model. Using that value, and the average speeds for each vehicle according to our MOVES model, we can calculate the age at which the required miles and required hours (if applicable) will be reached. The ages at which warranty and useful life are estimated to be reached are then determined as the minimum of the required age, the calculated age based on miles per year, and the calculated age based on hours per year (if applicable). The results for both warranty and useful life are shown in Table 7-15.

Table 7-15: Ages when Warranty and Useful Life are Estimated to be Reached, MY2027 Diesel HD Vehicles

	Baseline		FRM Control	
	Warranty	**Useful Life**	**Warranty**	**Useful Life**
Class 2b3				
Light Commercial Trucks	3.3	7.2	10.0	15.0
Long-Haul Single Unit Trucks	1.4	3.2	6.0	7.7
Passenger Trucks	3.3	7.2	10.0	15.0
Short-Haul Single Unit Trucks	2.3	5.0	9.5	12.2
Light HDE				
Long-Haul Single Unit Trucks	1.4	3.2	6.0	7.7
Other Buses	1.3	2.9	5.5	7.1
School Buses	3.8	8.5	10.0	15.0
Short-Haul Single Unit Trucks	2.3	5.0	9.5	12.2
Transit Buses	1.3	2.9	5.5	7.1
Medium HDE				
Long-Haul Single Unit Trucks	3.6	5.3	8.0	10.0
Motor Homes	5.0	10.0	10.0	12.0
Other Buses	3.3	4.9	7.4	9.2
Refuse Trucks	4.3	6.3	9.6	12.0
School Buses	5.0	10.0	10.0	12.0
Short-Haul Combination Trucks	1.7	2.6	3.9	4.9
Short-Haul Single Unit Trucks	5.0	8.4	10.0	12.0
Transit Buses	3.3	4.9	7.4	9.2
Heavy HDE				
Long-Haul Combination Trucks	1.8	3.1	3.2	4.7
Long-Haul Single Unit Trucks	5.0	10.0	10.0	11.0
Motor Homes	5.0	10.0	10.0	11.0
Other Buses	5.0	10.0	10.0	11.0
Refuse Trucks	5.0	10.0	10.0	11.0
School Buses	5.0	10.0	10.0	11.0
Short-Haul Combination Trucks	3.5	6.0	6.2	9.0
Short-Haul Single Unit Trucks	5.0	10.0	10.0	11.0
Urban Bus				
Transit Buses	2.6	10.0	10.0	11.0

Table 7-16: Ages when Warranty and Useful Life are Estimated to be Reached, MY2027 Gasoline HD Vehicles

	Baseline		FRM Control	
	Warranty	**Useful Life**	**Warranty**	**Useful Life**
Light HDE				
Long-Haul Single Unit Trucks	1.4	3.2	4.6	5.7
Motor Homes	5.0	10.0	10.0	15.0
Other Buses	1.3	2.9	4.2	5.3
School Buses	3.8	8.5	10.0	15.0
Short-Haul Single Unit Trucks	2.3	5.0	7.3	9.1
Transit Buses	1.3	2.9	4.2	5.3
Medium HDE				
Long-Haul Single Unit Trucks	1.4	3.2	4.6	5.7
Motor Homes	5.0	10.0	10.0	15.0
Short-Haul Single Unit Trucks	2.3	5.0	7.3	9.1
Heavy HDE				
Long-Haul Single Unit Trucks	1.4	3.2	4.6	5.7
Motor Homes	5.0	10.0	10.0	15.0
Short-Haul Single Unit Trucks	2.3	5.0	7.3	9.1

Table 7-17: Ages when Warranty and Useful Life are Estimated to be Reached, MY2027 CNG HD Vehicles

	Baseline		FRM Control	
	Warranty	**Useful Life**	**Warranty**	**Useful Life**
Heavy HDE				
Long-Haul Single Unit Trucks	5.0	10.0	10.0	11.0
Other Buses	5.0	10.0	10.0	11.0
Refuse Trucks	5.0	10.0	10.0	11.0
School Buses	5.0	10.0	10.0	11.0
Short-Haul Combination Trucks	3.5	6.0	6.2	9.0
Short-Haul Single Unit Trucks	5.0	10.0	10.0	11.0
Urban Bus				
Transit Buses	2.6	10.0	10.0	11.0

Lastly, with respect to warranty, we have estimated that many of the regulated products are sold today with a warranty period longer than the required warranty period. In the proposal, we calculated baseline warranty costs only for the required warranty periods. In the final analysis, we calculate baseline warranty costs for the warranty periods with which most are actually sold. For diesel and CNG heavy HDE, we assume all are sold with warranties covering 250,000 miles, and for diesel and CNG medium HDE, we assume half are sold with warranties covering 150,000 miles. For all other engines and associated fuel types, we have not estimated any use of extended warranties in the baseline.

We carry these annual warranty costs for both the baseline and the final standards despite the addition of new technology. We believe this is reasonable for two reasons: (1) the source data mentioned above included several years of data during which there must have been new technology introductions, yet annual costs appear to have remained generally steady; and, (2) the R&D we expect to be done, discussed next, is expected to improve overall durability, which should serve to help maintain historical annual costs.

For R&D, we have maintained the approach used in the proposal, although it is applied using the final useful life provisions. For R&D on a Class 8 truck, the final standards would extend regulatory useful life from 10 years, 22,000 hours, or 435,000 miles to 11 years, 32,000 hours, or 650,000 miles. We have applied a scaling factor of 1.49 (650/435) to the 0.05 R&D contribution factor for MYs 2027 and later. We apply this same methodology to estimating R&D for other vehicle categories. We estimate that once the development efforts into longer useful life are complete, increased expenditures will return to their normal levels of contribution. Therefore, we have implemented R&D scalars for three years (MY 2027 through MY 2029). In MY 2030 and later, the R&D scaling factors are no longer applied.

The VMT-based scaling factors applied to R&D cost contributors used in our cost analysis of final standards are shown in Table 7-18 for diesel and CNG regulatory classes and in Table 7-19 for gasoline regulatory classes.

Table 7-18: Scaling Factors Applied to RPE R&D Contribution Factors to Reflect Changes in their Contributions, Diesel & CNG Regulatory Classes

Scenario	MOVES Regulatory Class	MY2027 through MY2029	MY2030+
Baseline	Class 2b3	1.00	1.00
	Light HDE	1.00	1.00
	Medium HDE	1.00	1.00
	Heavy HDE	1.00	1.00
	Urban Bus	1.00	1.00
Final Program	Class 2b3	2.45	1.00
	Light HDE	2.45	1.00
	Medium HDE	1.89	1.00
	Heavy HDE	1.49	1.00
	Urban Bus	1.49	1.00

Table 7-19: Scaling Factors Applied to RPE Contribution Factors to Reflect Changes in their Contributions, Gasoline Regulatory Classes

Scenario	MOVES Regulatory Class	MY2027 through MY2029	MY2030+
Baseline	Light HDE	1.00	1.00
	Medium HDE	1.00	1.00
	Heavy HDE	1.00	1.00
Final Program	Light HDE	1.82	1.00
	Medium HDE	1.82	1.00
	Heavy HDE	1.82	1.00

Lastly, as mentioned in section 7.1.1, the markups for estimating indirect costs are applied to our estimates of the absolute direct manufacturing costs for emission-control technology shown in Table 7-5, Table 7-6 and Table 7-7, not just the incremental costs associated with the final program (i.e., the Baseline+Final costs).[J] Table 7-20 provides an illustrative example using a baseline technology cost of $5000, an incremental cost of $1000, and an indirect cost R&D contribution of 0.05 with a simple scalar of 1.5 associated with a longer useful life period. In this

[J] Increased indirect costs are included for the baseline technology because the final warranty and useful life provisions will impact those technologies too, not just the new, incremental technology.

case, the costs could be calculated according to two approaches as shown. By including the baseline costs, we are estimating considerable new R&D costs in the final analysis as illustrated by the example where including baseline costs results in R&D costs of $450 while excluding baseline costs results in R&D costs of just $75.[K]

Table 7-20: Simplified Example of Indirect Warranty Costs Calculated on an Incremental vs. Absolute Technology Package Cost (values are not from the analysis and are for presentation only)

	Using Baseline Costs Only	**Using Absolute Costs**
Baseline direct manufacturing cost (DMC)	$5000	$5000
Action case DMC	$1000	$5000 + $1000 = $6000
Indirect R&D Costs	$1000 x 0.05 x 1.5 = $75	$6000 x 0.05 x 1.5 = $450
Incremental DMC + R&D	$1000 + $75 = $1075	$1000 + $450 = $1450

7.1.3 **Technology Costs per Vehicle**

The following tables present the technology costs estimated for the final program on a per-vehicle basis for MY 2027. Reflected in these tables are learning effects on direct manufacturing costs and scaling effects associated with final program requirements. The sum is also shown and reflects the direct plus indirect cost per vehicle in the specific model year where direct costs refer to estimated direct manufacturing costs and indirect costs refer to estimated costs such as research and development, warranty, and administrative costs incurred by manufacturers in achieving compliance. Note that the indirect costs shown include warranty, R&D, "other," and profit, the latter two which scale with direct costs via the indirect cost contribution factor. While direct costs do not change across the different vehicle types (i.e., long-haul versus short-haul combination), the indirect costs do vary because differing miles driven and operating hours between types of vehicles result in different warranty and useful life estimates in actual use. These differences impact the estimated warranty and R&D costs.

Note that, while we show costs per vehicle here, it is important to remember that these are costs and not prices. We make no effort at estimating how manufacturers will price their products. Manufacturers may pass costs along to purchasers via price increases in a manner consistent with what we show here. However, manufacturers may also price certain products higher than what we show while pricing others lower--the higher-priced products thereby subsidizing the lower-priced products. This is true in any market, not just the heavy-duty highway industry. This is perhaps especially true with respect to the indirect costs we have estimated because, for example, R&D done to improve emission durability can readily transfer across different engines but the hardware added to an engine is uniquely tied to that engine.

Importantly, we present costs here for MY 2027 vehicles, but these costs continue for every model year going forward from there. Consistent with the learning impacts described in section 7.1.1, the costs per vehicle decrease slightly over time, but only the increased R&D costs are expected to decrease significantly. Increased R&D is estimated to occur for three years following

[K] As noted earlier, we have included baseline costs in this analysis because the final emissions warranty and regulatory useful life provisions will be expected to have some impact on not only the new technology added to comply with the final program, but also on any existing emission control systems (See Chapter 2 for more details on proposed Emissions Warranty and Regulatory Useful Life).

and including MY 2027, after which time its contribution to indirect costs is as shown in Table 7-13.

Table 7-21: MY2027 Diesel Class 2b3 Technology Costs per Vehicle Associated with the Final Program, 2017 dollars

	Direct Costs	Indirect Costs	Costs per Vehicle
FRM Baseline			
Light Commercial Trucks	3,681	3,448	7,130
Long-Haul Single Unit Trucks	3,681	2,321	6,003
Passenger Trucks	3,681	3,448	7,130
Short-Haul Single Unit Trucks	3,681	2,837	6,518
FRM Baseline+Final Program			
Light Commercial Trucks	5,601	8,775	14,376
Long-Haul Single Unit Trucks	5,601	6,310	11,912
Passenger Trucks	5,601	8,775	14,376
Short-Haul Single Unit Trucks	5,601	8,477	14,078
Increased Cost of the Final Program			
Light Commercial Trucks	1,920	5,326	7,246
Long-Haul Single Unit Trucks	1,920	3,989	5,909
Passenger Trucks	1,920	5,326	7,246
Short-Haul Single Unit Trucks	1,920	5,640	7,560

Table 7-22: MY2027 Diesel Light HDE Technology Costs per Vehicle Associated with the Final Program, 2017 dollars

	Direct Costs	Indirect Costs	Costs per Vehicle
FRM Baseline			
Long-Haul Single Unit Trucks	3,699	2,332	6,031
Other Buses	3,699	2,263	5,962
School Buses	3,699	3,829	7,528
Short-Haul Single Unit Trucks	3,699	2,851	6,550
Transit Buses	3,699	2,263	5,962
FRM Baseline + Final Program			
Long-Haul Single Unit Trucks	5,656	6,353	12,009
Other Buses	5,656	6,064	11,720
School Buses	5,656	8,830	14,485
Short-Haul Single Unit Trucks	5,656	8,530	14,186
Transit Buses	5,656	6,064	11,720
Increased Cost of the Final Program			
Long-Haul Single Unit Trucks	1,957	4,021	5,978
Other Buses	1,957	3,800	5,757
School Buses	1,957	5,001	6,957
Short-Haul Single Unit Trucks	1,957	5,680	7,636
Transit Buses	1,957	3,800	5,757

Table 7-23: MY2027 Diesel Medium HDE Technology Costs per Vehicle Associated with the Final Program, 2017 dollars

	Direct Costs	Indirect Costs	Costs per Vehicle
FRM Baseline			
Long-Haul Single Unit Trucks	3,808	3,774	7,582
Motor Homes	3,808	4,682	8,490
Other Buses	3,808	3,597	7,404
Refuse Trucks	3,808	4,217	8,025
School Buses	3,808	4,682	8,490
Short-Haul Combination Trucks	3,808	2,595	6,402
Short-Haul Single Unit Trucks	3,808	4,682	8,490
Transit Buses	3,808	3,597	7,404
FRM Baseline + Final Program			
Long-Haul Single Unit Trucks	5,625	7,572	13,197
Motor Homes	5,625	8,839	14,464
Other Buses	5,625	7,175	12,799
Refuse Trucks	5,625	8,564	14,189
School Buses	5,625	8,839	14,464
Short-Haul Combination Trucks	5,625	4,930	10,555
Short-Haul Single Unit Trucks	5,625	8,839	14,464
Transit Buses	5,625	7,175	12,799
Increased Cost of the Final Program			
Long-Haul Single Unit Trucks	1,817	3,798	5,615
Motor Homes	1,817	4,157	5,974
Other Buses	1,817	3,578	5,395
Refuse Trucks	1,817	4,347	6,164
School Buses	1,817	4,157	5,974
Short-Haul Combination Trucks	1,817	2,335	4,153
Short-Haul Single Unit Trucks	1,817	4,157	5,974
Transit Buses	1,817	3,578	5,395

Table 7-24: MY2027 Diesel Heavy HDE Technology Costs per Vehicle Associated with the Final Program, 2017 dollars

	Direct Costs	Indirect Costs	Costs per Vehicle
FRM Baseline			
Long-Haul Combination Trucks	5,816	4,025	9,841
Long-Haul Single Unit Trucks	5,816	7,151	12,967
Motor Homes	5,816	7,151	12,967
Other Buses	5,816	7,151	12,967
Refuse Trucks	5,816	7,151	12,967
School Buses	5,816	7,151	12,967
Short-Haul Combination Trucks	5,816	5,658	11,473
Short-Haul Single Unit Trucks	5,816	7,151	12,967
FRM Baseline + Final Program			
Long-Haul Combination Trucks	8,132	6,535	14,667
Long-Haul Single Unit Trucks	8,132	13,139	21,271
Motor Homes	8,132	13,139	21,271
Other Buses	8,132	13,139	21,271
Refuse Trucks	8,132	13,139	21,271
School Buses	8,132	13,139	21,271
Short-Haul Combination Trucks	8,132	9,474	17,606
Short-Haul Single Unit Trucks	8,132	13,139	21,271
Increased Cost of the Final Program			
Long-Haul Combination Trucks	2,316	2,510	4,827
Long-Haul Single Unit Trucks	2,316	5,988	8,304
Motor Homes	2,316	5,988	8,304
Other Buses	2,316	5,988	8,304
Refuse Trucks	2,316	5,988	8,304
School Buses	2,316	5,988	8,304
Short-Haul Combination Trucks	2,316	3,816	6,132
Short-Haul Single Unit Trucks	2,316	5,988	8,304

Table 7-25: MY2027 Diesel Urban Bus Technology Costs per Vehicle Associated with the Final Program, 2017 dollars

	Direct Costs	Indirect Costs	Costs per Vehicle
FRM Baseline	3,884	3,238	7,122
FRM Baseline+Final Program	5,734	8,901	14,635
Increased Cost of the Final Program	1,850	5,663	7,512

Table 7-26: MY2027 Gasoline HDE Technology Costs per Vehicle Associated with the Final Program, 2017 dollars

	Direct Costs	Indirect Costs	Costs per Vehicle
FRM Baseline			
Long-Haul Single Unit Trucks	2,681	1,905	4,585
Motor Homes	2,681	3,511	6,192
Other Buses	2,681	1,855	4,535
School Buses	2,681	2,989	5,670
Short-Haul Single Unit Trucks	2,681	2,280	4,961
Transit Buses	2,681	1,855	4,535
FRM Baseline+Final Program			
Long-Haul Single Unit Trucks	3,369	3,784	7,153
Motor Homes	3,369	6,223	9,592
Other Buses	3,369	3,624	6,993
School Buses	3,369	6,223	9,592
Short-Haul Single Unit Trucks	3,369	4,986	8,355
Transit Buses	3,369	3,624	6,993
Increased Cost of the Final Program			
Long-Haul Single Unit Trucks	688	1,880	2,568
Motor Homes	688	2,712	3,401
Other Buses	688	1,770	2,458
School Buses	688	3,234	3,923
Short-Haul Single Unit Trucks	688	2,706	3,394
Transit Buses	688	1,770	2,458

Table 7-27: MY2027 CNG Heavy HDE Technology Costs per Vehicle Associated with the Final Program, 2017 dollars

	Direct Costs	Indirect Costs	Costs per Vehicle
FRM Baseline			
Long-Haul Single Unit Trucks	8,585	10,556	19,141
Other Buses	8,585	10,556	19,141
Refuse Trucks	8,585	10,556	19,141
School Buses	8,585	10,556	19,141
Short-Haul Combination Trucks	8,585	8,351	16,936
Short-Haul Single Unit Trucks	8,585	10,556	19,141
FRM Baseline+Final Program			
Long-Haul Single Unit Trucks	8,610	17,988	26,598
Other Buses	8,610	17,988	26,598
Refuse Trucks	8,610	17,988	26,598
School Buses	8,610	17,988	26,598
Short-Haul Combination Trucks	8,610	12,577	21,187
Short-Haul Single Unit Trucks	8,610	17,988	26,598
Increased Cost of the Final Program			
Long-Haul Single Unit Trucks	25	7,431	7,457
Other Buses	25	7,431	7,457
Refuse Trucks	25	7,431	7,457
School Buses	25	7,431	7,457
Short-Haul Combination Trucks	25	4,225	4,251
Short-Haul Single Unit Trucks	25	7,431	7,457

Table 7-28: MY2027 CNG Urban Bus Technology Costs per Vehicle Associated with the Final Program, 2017 dollars

	Direct Costs	Indirect Costs	Costs per Vehicle
FRM Baseline	6,438	5,367	11,806
FRM Baseline+Final Program	6,457	13,490	19,948
Increased Cost of the Final Program	19	8,123	8,142

7.2 Operating Costs

We have estimated three impacts on operating costs expected to be incurred by users of new MY 2027 and later heavy-duty vehicles: increased diesel exhaust fluid (DEF) consumption by diesel vehicles due to increased DEF dose rates to enable compliance with more stringent NO_X standards; decreased fuel costs by gasoline vehicles due to new onboard refueling vapor recovery systems that allow burning (in engine) of otherwise evaporated hydrocarbon emissions; and, emission repair impacts. For the repair impacts, we expect that the longer duration warranty period will result in lower owner/operator-incurred repair costs due to fewer repairs being paid for by owners/operators since more costs will be borne by the manufacturer, and that the longer duration useful life periods will result in increased emission control system durability. We have estimated the net effect on repair costs and describe our approach, along with increased DEF consumption and reduced gasoline consumption below.

As noted in the introductory text to this chapter, the operating costs we estimate here are for the heavy-duty truck operation impacted by the final program (e.g., repair of emission-related components). These costs (and savings) are incurred by heavy-duty truck purchasers/owners.

7.2.1 Costs Associated with Increased Diesel Exhaust Fluid (DEF) Consumption in Diesel Engines

To estimate baseline case DEF consumption in heavy-duty vehicles with diesel engines, this analysis uses the relationship, shown below, of DEF dose rate relative to the reduction in NO_X over the SCR catalyst.[L,17]

$$NOx\ reduction = -73.679x + 0.0149$$

where x is equal to the DEF dose rate. This relationship was developed giving consideration to FTP emissions. By estimating the FTP NO_X reduction across the SCR catalyst, the DEF dose rate can be calculated. NO_X reduction is estimated from the difference between estimated engine-out and FTP tailpipe NO_X emissions; these variables along with the calculated DEF dose rate for the baseline case are shown in Table 7-29.

[L] The relationship between DEF dose rate and NOx reduction across the SCR catalyst is based on methodology presented in the Technical Support Document to the 2012 Non-conformance Penalty rule (the NCP Technical Support Document, or NCP TSD).

Table 7-29: Diesel Exhaust Fluid Consumption Rates for Diesel Vehicles in the Baseline Case

	Value
Engine-out NOX (FTP g/hp-hr)	4.0
Tailpipe NOX (FTP mg/hp-hr)	200
DEF Dose Rate (% of fuel consumed)	5.18%

To estimate DEF consumption impacts under the final program, which involves changes to not only the new FTP emission standards but also the new SET and LLC standards along with new off-cycle standards, we developed a new approach to estimate DEF consumption. For this analysis, we scaled DEF consumption with the NO_X reductions achieved under each of the alternatives. To do this, we considered the molar mass of NO_X, the molar mass of urea, the molar ratio of NO to NO_2, the mass concentration of urea in DEF along with the density of DEF to estimate the theoretical gallons of DEF consumed per ton of NO_X reduced at 442 gallons/ton. The theoretical DEF dosing rates was then compared to the data collected from the CARB Stage 3 test program for the hot FTP, SET and LLC (see Chapter 3 of the RIA). The data from this testing showed that the NO_X specific DEF dosing was 536, 478, and 568 gallons/ton for the hot FTP, SET and LLC, respectively. Since this data takes into account any over dosing that occurs for part of the cycle, and NO_2/NO ratio being greater than 1 for parts of the cycle, we have adjusted the theoretical 442 gallons/ton NO_X to the average of the hot FTP, SET and LLC, which is 527 gallons/ton. These values are shown in Table 7-30.

Table 7-30: Derivation of DEF Consumption per Ton of NO_X Reduced

	Value
Molar mass of NO_X (g/mol)	46.0055
Molar mass of urea (g/mol)	60.07
Molar ratio of NO to NO_2	1
Mass concentration of urea in DEF	0.325
Density of DEF (g/mL)	1.09
Theoretical gallons DEF/ton NO_X reduced	442
Proposed gallons DEF/ton NO_X reduced	527

The final calculation of DEF consumption in each option is to multiply the gallons of diesel fuel consumed by 5.18%, to account for the baseline DEF consumption and add to that amount 527 gallons of DEF for each ton of NO_X reduced from the baseline. Both the gallons of diesel fuel consumed and the tons of NO_X reduced are taken directly from the year-over-year MOVES results.

The gallons of DEF consumed are then multiplied by the estimated price of DEF per gallon. This analysis uses the DEF prices presented in the NCP Technical Support Document with growth beyond 2042 projected at the same 1.3 percent rate as noted in the NCP TSD. Note that the DEF prices presented in Table 7-31 update the NCP TSD's 2011 prices to 2017 dollars using the GDP deflator presented in Table 7-8.

Table 7-31: Diesel Exhaust Fluid Price per Gallon (2017 dollars)

Calendar Year	DEF Price/Gallon
2027	3.25
2028	3.30
2029	3.33
2030	3.37
2031	3.42
2032	3.46
2033	3.52
2034	3.56
2035	3.60
2036	3.65
2037	3.69
2038	3.75
2039	3.79
2040	3.85
2041	3.89
2042	3.94
2043	4.00
2044	4.04
2045	4.10

The impacts on DEF costs are shown in Table 7-32. Note that the impacts of the final program are the increased costs shown in Table 7-32, the baseline and final program costs are shown to provide a sense of scale for the increased costs. Because these are operating costs which occur over time, we present them at both 3 and 7 percent discount rates.

Table 7-32: MY2027 Lifetime DEF Costs per Diesel Vehicle Associated with Final NOx Standards, 2017 dollars

	3% Discount Rate				7% Discount Rate			
	Light HDE	Medium HDE	Heavy HDE	Urban Bus	Light HDE	Medium HDE	Heavy HDE	Urban Bus
FRM Baseline								
Long-Haul Combination Trucks			34,009				25,768	
Long-Haul Single Unit Trucks	3,759	5,686	6,823		2,937	4,443	5,331	
Motor Homes		1,489	1,764			1,068	1,265	
Other Buses	9,118	11,285	11,688		6,695	8,286	8,582	
Refuse Trucks		8,435	8,787			6,317	6,581	
School Buses	2,331	3,030	3,187		1,712	2,225	2,340	
Short-Haul Combination Trucks		16,323	17,154			12,735	13,384	
Short-Haul Single Unit Trucks	2,733	4,144	4,975		2,100	3,184	3,823	
Transit Buses	9,192	11,254		11,742	6,750	8,263		8,622
FRM Baseline+Final Program								
Long-Haul Combination Trucks			37,621				28,580	
Long-Haul Single Unit Trucks	4,011	6,215	7,916		3,136	4,865	6,200	
Motor Homes		1,617	2,016			1,162	1,450	
Other Buses	9,805	12,277	13,594		7,209	9,040	10,011	
Refuse Trucks		9,182	10,246			6,895	7,696	
School Buses	2,501	3,293	3,671		1,839	2,424	2,702	
Short-Haul Combination Trucks		17,575	19,378			13,727	15,154	
Short-Haul Single Unit Trucks	2,949	4,573	5,864		2,268	3,522	4,517	
Transit Buses	9,867	12,149		13,410	7,253	8,945		9,863
Increased Cost of the Final Program								
Long-Haul Combination Trucks			3,612				2,812	
Long-Haul Single Unit Trucks	252	529	1,094		199	422	869	
Motor Homes		128	253			94	185	
Other Buses	687	992	1,906		514	754	1,428	
Refuse Trucks		747	1,459			579	1,115	
School Buses	170	263	484		127	199	362	
Short-Haul Combination Trucks		1,251	2,224			992	1,771	
Short-Haul Single Unit Trucks	216	429	889		168	337	694	
Transit Buses	675	896		1,669	504	681		1,241

7.2.2 <u>Costs Associated with Changes in Fuel Consumption on Gasoline Engines</u>

This analysis estimates a small decrease in fuel costs, i.e., fuel savings, by vehicles equipped with gasoline engines because the final ORVR system will be expected to capture previously evaporated fuel and then burn that fuel in the engine (see Table 7-6 for our estimated ORVR direct manufacturing cost). To estimate these impacts, we first converted grams of hydrocarbon captured by the ORVR system to milliliters of gasoline that will ultimately be burned in the engine. Based on that relationship, we estimate that 1.48 milliliters of gasoline will be consumed for each gram of hydrocarbon emissions reduced under the final program. We estimated this value, 1.48, by assuming that the ORVR system will exchange captured butane for gasoline on an energy basis and Tier 3 certification fuel has a density of 0.7482 g/ml, or 2832 g/gal at 60 degrees F.[18] We then used a butane energy density of 45.8 MJ/kg, or 19752 Btu/lb,[19] and the Tier 3 certification fuel energy density of 17890 Btu/lb,[20] giving a ratio of 1.117 grams of gasoline displaced for each gram of butane burned. Using the density of Tier 3 certification fuel, we get

1.117 / 2832 = 0.0003943 gallons, or 1.48 ml, of gasoline saved for each gram of butane captured since the owner/operator is no longer paying for evaporated fuel as it will be burned in the engine. Using AEO 2019 reference case gasoline prices, the impacts on fuel costs for MY2027 gasoline HD vehicles are shown in Table 7-33 (note that negative values indicate lower fuel costs, or fuel savings). In the aggregate, we estimate that the ORVR requirements in the final program will result in an annual reduction of approximately 0.3 million (calendar year 2027) to 4.9 million (calendar year 2045) gallons of gasoline, representing roughly 0.1 percent of gasoline consumption from impacted vehicles.

Table 7-33: MY2027 Lifetime Fuel Costs per Gasoline Vehicle Associated with ORVR Requirements, 2017 dollars

	3% Discount Rate			7% Discount Rate		
	Light HDE	**Medium HDE**	**Heavy HDE**	**Light HDE**	**Medium HDE**	**Heavy HDE**
FRM Baseline						
Long-Haul Single Unit Trucks	120,876	150,530	192,727	94,841	118,108	151,216
Motor Homes	30,329	38,339	48,887	21,905	27,691	35,309
Other Buses	273,223			201,982		
School Buses	69,242			51,188		
Short-Haul Single Unit Trucks	86,494	109,427	139,754	66,791	84,501	107,918
Transit Buses	269,797			199,449		
FRM Baseline+Final Program						
Long-Haul Single Unit Trucks	120,744	150,349	192,470	94,739	117,969	151,019
Motor Homes	30,271	38,260	48,781	21,864	27,635	35,233
Other Buses	272,656			201,570		
School Buses	69,110			51,092		
Short-Haul Single Unit Trucks	86,397	109,292	139,566	66,717	84,399	107,777
Transit Buses	269,245			199,047		
Increased Cost of the Final Program						
Long-Haul Single Unit Trucks	-132	-181	-257	-102	-139	-197
Motor Homes	-58	-79	-106	-41	-56	-75
Other Buses	-567			-412		
School Buses	-132			-96		
Short-Haul Single Unit Trucks	-97	-135	-187	-74	-102	-141
Transit Buses	-552			-402		

7.2.3 <u>Emission-Related Repair Cost Impacts Associated with the Final Program</u>

The final extended warranty and useful life requirements will have an impact on emission-related repair costs incurred by heavy-duty vehicle owners. Researchers have noted the relationships among quality, reliability, and warranty for a variety of goods.[21] Wu, for instance, examines how analyzing warranty data can provide "early warnings" on product problems that can then be used for design modifications.[22] Guajardo et al. describe one of the motives for warranties to be "incentives for the seller to improve product quality;" specifically for light-duty vehicles, they find that buyers consider warranties to substitute for product quality, and to complement service quality.[23] The other rationales are protection for consumers against failures, provision of product quality information to consumers, and a means to distinguish consumers according to their risk preferences. Murthy and Jack, for new products, and Saidi-Mehrabad et al.

for second-hand products, consider the role of warranties in improving a buyer's confidence in quality of the good.[24,25]

On the one hand, we expect owner-incurred emission repair costs to decrease due to the final program because the longer emission warranty requirements will result in more repair costs covered by the OEMs. Further, we expect improved serviceability in an effort by OEMs to decrease repair costs they will incur. We also expect that the longer useful life periods in the final program will result in more durable parts to ensure regulatory compliance over the longer timeframe. On the other hand, we also expect that the more costly emission control systems required by the final program may result in higher repair costs which might increase owner-incurred costs outside the warranty and useful life periods.

As discussed in Chapter 7.1.2, we have estimated increased OEM costs associated with increased warranty liability (i.e., longer warranty periods), and for more durable parts resulting from the longer useful life periods. These costs are accounted for via increased warranty costs and increased research and development (R&D) costs. We also included additional aftertreatment costs in the direct manufacturing costs to address the increased useful life requirements (e.g., larger catalyst volume; see Chapters 2 and 3 of the RIA for detailed discussions). We estimate that these efforts will help to reduce emission repair costs during the emission warranty and regulatory useful life periods, and possibly beyond.

In the proposal, to estimate impacts on emission repair costs, we began with an emission repair cost curve derived from an industry white paper. [26] Some commenters took exception to our approach, preferring instead that we use what they consider to be a more established repair and maintenance cost estimate from the American Transportation Research Institute.[27] Given the duration of the ATRI study and the amount of data behind it, we have moved to using that study in this final analysis (the November 2021 study).

In the ATRI study, 10 years of repair and maintenance costs are presented. Note that the costs presented by ATRI are for all repair and maintenance, not just emission-related repair and maintenance and not just emission-related repair—the real focus of our analysis since our emission-related warranty and useful life provisions are geared only toward emission-related systems. Also, in the ATRI study, they do not provide a dollar basis for their results. In looking at a prior 2019 ATRI study,[28] we found identical values presented in all applicable years as were presented in the 2021 study (see Table 8 of the 2021 study and Table 9 of the 2019 study). This led us to believe that the costs presented had not been updated to a consistent dollar basis but instead were reported in nominal terms in each year. We then converted the ATRI costs to 2017 dollars using the deflators presented in Table 7-8 and calculated the average value over the 10 years of data to arrive at the 0.158 dollars per mile and 6.31 dollars per hour values used as a starting point in our analysis.

Table 7-34: Repair and Maintenance Costs per Mile

Calendar Year	ATRI Study, Dollars per mile	EPA Assumed Dollar Basis	ATRI Cost in 2017 Dollars, Dollars per mile
2011	0.152	2011	0.167
2012	0.138	2012	0.149
2013	0.148	2013	0.157
2014	0.158	2014	0.164
2015	0.156	2015	0.161
2016	0.166	2016	0.169
2017	0.167	2017	0.167
2018	0.171	2018	0.167
2019	0.149	2019	0.143
2020	0.148	2020	0.140
Average	0.155		0.158

Table 7-35: Repair and Maintenance Costs per Hour of Operation

Calendar Year	ATRI Study, Dollars per hour	EPA Assumed Dollar Basis	ATRI Cost in 2017 Dollars, Dollars per hour
2011	6.07	2011	6.66
2012	5.52	2012	5.95
2013	5.92	2013	6.27
2014	6.31	2014	6.56
2015	6.23	2015	6.41
2016	6.65	2016	6.78
2017	6.58	2017	6.58
2018	6.72	2018	6.56
2019	5.87	2019	5.63
2020	6.00	2020	5.69
Average	6.19		6.31

The ATRI study is a comprehensive study of tractor-trailer fleet freight operators. As such, the costs presented in Table 7-34 and Table 7-35 are taken as representative of diesel heavy HDE equipped vehicles. Given the different emission control system costs of light and medium HDEs, we considered it necessary to adjust the 0.158 and 6.31 dollar values to be more applicable to light and medium HD vehicles. To do this, we used the same approach as described above in scaling diesel HDE engine warranty costs for engines of other sizes and other fuels. We also wanted to adjust the repair and maintenance to reflect only emission-related repair, eliminating not only non-emission-related repairs but also all maintenance. To do so, we used the approach used for the same purpose and described in the proposal. To estimate the emission repair portion of these costs, we used the figure on page 3 of the Fleet Advantage whitepaper which showed the percent of total repair and maintenance costs attributable to different systems on the vehicle.[29] The details of that chart are recreated below in Table 7-36 along with EPA's estimates for what portion of the repair and maintenance costs could be considered to be emission repairs.

As shown, our analysis estimates that 10.8 percent of repair and maintenance costs are emission repairs that our warranty and useful life provisions are meant to impact. In general, the maintenance/repair shares are estimated to be 50/50 repair/maintenance, with the exception of "Preventive maintenance" and "Exhaust system." Preventive maintenance, by definition, is not repair, and thus 100 percent is considered maintenance. For the exhaust system, we estimate that 80 percent of those costs are repair costs, with maintenance costs limited to DPF cleaning. The share of emission-related vs. non-emission-related items was broken down, in general, by those that are clearly emission-related, where we attribute 100 percent to the emission-related category, versus those that are clearly not emission-related where 100 percent is attributed to the non-emission-related category. Not shown are the non-emission repair share which totals 35.0 percent, the emission maintenance share which is 7.2 percent and the non-emission maintenance share which is 47.0 percent.

Table 7-36: Percentage of Total Repair & Maintenance Costs Attributable to Different Vehicle Systems[30]

	Fleet Advantage	EPA Estimates				
System	**Percent of Total Repair & Maintenance Cost**	**Maintenance Share**	**Repair Share**	**Non-Emission-Related Share**	**Emission-Related Share**	**Emission Repair Share (EPA Estimate)**
Tires, tubes, liners & valves	43%	50%	50%	100%	0%	0.0%
Preventive maintenance	12%	100%	0%	100%	0%	0%
Brakes	9%	50%	50%	100%	0%	0%
Expendable items	8%	50%	50%	100%	0%	0%
Lighting	5%	50%	50%	100%	0%	0%
Cranking	5%	50%	50%	100%	0%	0%
Power plant	3%	50%	50%	0%	100%	1.5%
Exhaust system	6%	20%	80%	0%	100%	4.8%
Fuel system	6%	50%	50%	0%	100%	3%
Engine/motor	3%	50%	50%	0%	100%	1.5%
Total	100%	54.2%	45.8%	82.0%	18.0%	10.8%

The end results are shown in Table 7-37 and Table 7-38 which show the dollar per mile and dollar per hour values, respectively, used in this analysis for emission-related repairs. Note that the costs shown are scaled upward as described in the text (i.e., package direct cost divided by the baseline package direct cost within each regulatory class and fuel type) for emission-related repairs done beyond the useful life. Which set of repair cost values we used, dollar per mile or dollar per hour of operation, is shown in Table 7-39 along with the average speeds used to estimate the number of hours of operation per year (MOVES vehicle miles travelled divided by average speed). We have applied the dollar per hour value for most vocational vehicles and the dollar per mile value for others.

Table 7-37: Emission-Related Repair Costs per Mile, 2017 dollars per mile *

		Repair & Maintenance			Emission-Related Repair (10.8% of Repair & Maintenance)		
	Scaling Approach	Diesel	Gasoline	CNG	Diesel	Gasoline	CNG
Class 2b3	Base Class 2b3 DMC / Base Diesel Heavy HDE DMC	0.100			0.011		
Light HDE	Base Light HDE DMC / Base Diesel Heavy HDE DMC	0.101	0.073		0.011	0.008	
Medium HDE	Base Medium HDE DMC / Base Diesel Heavy HDE DMC	0.103	0.073		0.011	0.008	
Heavy HDE	Base Heavy HDE DMC / Base Diesel Heavy HDE DMC	0.158	0.073	0.232	0.017	0.008	0.025
Urban Bus	Base Urban bus DMC / Base Diesel Heavy HDE DMC	0.098		0.162	0.011		0.018

* The Base Diesel Heavy HDE DMC would be the $5,816 value shown in Table 7-5.

Table 7-38: Emission-Related Repair Costs per Hour of Operation, 2017 dollars per hour *

		Repair & Maintenance			Emission-Related Repair (10.8% of Repair & Maintenance)		
	Scaling Approach	Diesel	Gasoline	CNG	Diesel	Gasoline	CNG
Class 2b3	Base Class 2b3 DMC / Base Diesel Heavy HDE DMC	3.99			0.431		
Light HDE	Base Light HDE DMC / Base Diesel Heavy HDE DMC	4.03	2.91		0.435	0.314	
Medium HDE	Base Medium HDE DMC / Base Diesel Heavy HDE DMC	4.11	2.91		0.444	0.314	
Heavy HDE	Base Heavy HDE DMC / Base Diesel Heavy HDE DMC	6.31	2.91	9.27	0.714	0.314	1.00
Urban Bus	Base Urban bus DMC / Base Diesel Heavy HDE DMC	3.91		6.47	0.422		0.699

* The Base Diesel Heavy HDE DMC would be the $5,816 value shown in Table 7-5.

Table 7-39: Repair Cost per Unit-Attribute and Average Speeds used in Calculating Emission-Related Repair Costs

MOVES Sourcetype	Repair cost per unit attribute	Average Speed (miles per hour) *
Passenger Trucks	Dollars per mile	42.9
Light Commercial Trucks	Dollars per mile	41.2
Other Buses	Dollars per hour	42.5
Transit Buses	Dollars per hour	43.0
School Buses	Dollars per hour	43.0
Refuse Trucks	Dollars per hour	44.0
Short-Haul Single Unit Trucks	Dollars per hour	43.9
Long-Haul Single Unit Trucks	Dollars per mile	44.5
Motor Homes	Dollars per mile	43.9
Short-Haul Combination Trucks	Dollars per mile	47.7
Long-Haul Combination Trucks	Dollars per mile	51.5

* Sonntag, Darrell. Population and Activity of Onroad Vehicles in MOVES_CTI_NPRM. Attachment to Memorandum to Docket EPA-HQ-OAR-2019-0055: "Updates to MOVES for Emissions Analysis of the HD 2027 NPRM." May 2021.

As noted above, given that future engines and vehicles will be equipped with new, more costly technology, it is possible that the annual repair costs for vehicles under the final program will be higher than the annual repair costs in the baseline. We have included such an increase for the period beyond useful life by scaling the emission-related repair costs shown in Table 7-37 and Table 7-38 by the direct manufacturing costs of the applicable regulatory class divided by its direct manufacturing costs in the baseline. In other words, if the final program direct manufacturing cost for a diesel HDE is $8,455 and its baseline cost is $5,816, the 0.017 emission-related repair cost per mile value shown in Table 7-37 would become 0.025 (8455 divided by 5816 times 0.017). This is perhaps conservative because it seems reasonable to assume that the R&D used to improve durability during the useful life period would also improve durability beyond it. Nonetheless, we also think it is reasonable to include an increase in repair costs, relative to the baseline case, because the period beyond useful life is of marginally less concern to manufacturers.[M] Table 7-40, Table 7-41 and Table 7-42 show the emission-related repair cost per mile or cost per hour values used in the analysis for the period beyond the estimated useful life.

[M] This is not meant to suggest that manufacturers no longer care about their products beyond their regulatory useful life, but rather to support the expectation that regulatory pressures--i.e., regulatory compliance during the useful life--tend to focus resources.

Table 7-40: Diesel Emission-Related Repair Cost per Mile or Cost per Hour used for the Period Beyond Useful Life

		Baseline		Proposal	
	Scaling Approach	Cents per mile	Cents per hour	Cents per mile	Cents per hour
Class 2b3					
Light Commercial Trucks	Final Class 2b3 DMC / Base Class 2b3 DMC	1.08		1.74	
Long-Haul Single Unit Trucks		1.08		1.74	
Passenger Trucks		1.08		1.74	
Short-Haul Single Unit Trucks			43.1		69.4
Light HDE					
Long-Haul Single Unit Trucks	Final Light HDE DMC / Base Light HDE DMC	1.09		1.75	
Other Buses			43.3		70.1
School Buses			43.3		70.1
Short-Haul Single Unit Trucks			43.3		70.1
Transit Buses			43.3		70.1
Medium HDE					
Long-Haul Single Unit Trucks	Final Medium HDE DMC / Base Medium HDE DMC	1.12		1.75	
Motor Homes		1.12		1.75	
Other Buses			44.6		69.7
Refuse Trucks			44.6		69.7
School Buses			44.6		69.7
Short-Haul Combination Trucks		1.12		1.75	
Short-Haul Single Unit Trucks			44.6		69.7
Transit Buses			44.6		69.7
Heavy HDE					
Long-Haul Combination Trucks	Final Heavy HDE DMC / Base Heavy HDE DMC	1.71		2.48	
Long-Haul Single Unit Trucks		1.71		2.48	
Motor Homes		1.71		2.48	
Other Buses			68.1		99.1
Refuse Trucks			68.1		99.1
School Buses			68.1		99.1
Short-Haul Combination Trucks		1.71		2.48	
Short-Haul Single Unit Trucks			68.1		99.1
Urban Bus					
Transit Buses	Final Urban bus HDE DMC / Base Urban bus HDE DMC		45.5		71.0

Table 7-41: Gasoline Emission-Related Repair Cost per Mile or Cost per Hour used for the Period Beyond Useful Life

		Baseline		Proposal	
	Scaling Approach	Cents per mile	Cents per hour	Scaling Approach	Cents per mile
Light HDE					
Long-Haul Single Unit Trucks	Final Light HDE DMC / Base Light HDE DMC	0.79		0.99	
Motor Homes		0.79		0.99	
Other Buses			31.4		39.5
School Buses			31.4		39.5
Short-Haul Single Unit Trucks			31.4		39.5
Transit Buses			31.4		39.5
Medium HDE					
Long-Haul Single Unit Trucks	Final Medium HDE DMC / Base Medium HDE DMC	0.79		0.99	
Motor Homes		0.79		0.99	
Short-Haul Single Unit Trucks			31.4		39.5
Heavy HDE					
Long-Haul Single Unit Trucks	Final Heavy HDE DMC / Base Heavy HDE DMC	0.79		0.99	
Motor Homes		0.79		0.99	
Short-Haul Single Unit Trucks			31.4		39.5

Table 7-42: CNG Emission-Related Repair Cost per Mile or Cost per Hour used for the Period Beyond Useful Life

		Baseline		Proposal	
	Scaling Approach	Cents per mile	Cents per hour	Scaling Approach	Cents per mile
Heavy HDE					
Long-Haul Single Unit Trucks	Final Heavy HDE DMC / Base Heavy HDE DMC	2.52		2.53	
Other Buses			100.6		100.9
Refuse Trucks			100.6		100.9
School Buses			100.6		100.9
Short-Haul Combination Trucks		2.52		2.53	
Short-Haul Single Unit Trucks			100.6		100.9
Urban Bus					
Transit Buses	Final Urban bus HDE DMC / Base Urban bus HDE DMC		75.4		75.7

As done for warranty costs, we have used estimated ages for when warranty and useful life are reached, using the required miles, ages and hours along with the estimated miles driven and hours of operation for each specific type of vehicle (refer to Table 7-15, Table 7-16, and Table 7-17 to see how many years of warranty and useful life are actually estimated for each type of diesel, gasoline and CNG vehicle, respectively). As noted above, this means that warranty and useful life ages are reached in different years for a long-haul combination truck driving over 100,000 miles per year or over 2,000 hours per year and a refuse truck driven around 40,000 miles per year or operating less than 1,000 hours per year. The resultant MY 2027 lifetime emission-related repair costs are shown in Table 7-43 for diesel HD vehicles, Table 7-44 for gasoline HD vehicles and Table 7-45 for CNG HD vehicles. Since these costs occur over time, we present them using both a 3 percent and a 7 percent discount rate.

Note that these costs assume that all emission-related repair costs are paid by manufacturers during the warranty period, and beyond the warranty period the emission-related repair costs are incurred by owners/operators.

Table 7-43: MY2027 Lifetime Emission-Related Repair Costs per Diesel Vehicle, 2017 dollars

	3% Discount Rate					7% Discount Rate				
	Class 2b3	Light HDE	Medium HDE	Heavy HDE	Urban bus	Class 2b3	Light HDE	Medium HDE	Heavy HDE	Urban bus
FRM Baseline										
Light Commercial Trucks	1,502					1,030				
Long-Haul Combination Trucks				22,041					16,138	
Long-Haul Single Unit Trucks	3,192	3,208	2,493	3,060		2,429	2,440	1,790	2,109	
Motor Homes			613	936				394	602	
Other Buses		4,292	3,668	4,719			3,083	2,499	3,074	
Passenger Trucks	1,502					1,030				
Refuse Trucks			2,222	3,110				1,506	2,065	
School Buses		1,148	1,050	1,604			771	684	1,045	
Short-Haul Combination Trucks			6,635	8,088				5,003	5,823	
Short-Haul Single Unit Trucks	1,790	1,799	1,292	1,973		1,311	1,318	876	1,338	
Transit Buses		4,242	3,625		3,941		3,047	2,469		2,732
FRM Baseline+Final Program										
Light Commercial Trucks	790					449				
Long-Haul Combination Trucks				25,070					17,497	
Long-Haul Single Unit Trucks	2,264	2,284	1,531	1,524		1,497	1,509	956	906	
Motor Homes			480	728				272	415	
Other Buses		4,090	3,261	3,454			2,598	1,978	1,979	
Passenger Trucks	790					449				
Refuse Trucks			1,408	2,038				819	1,180	
School Buses		667	772	1,174			378	439	673	
Short-Haul Combination Trucks			7,029	6,436				4,960	4,225	
Short-Haul Single Unit Trucks	758	764	721	1,115		447	451	421	655	
Transit Buses		4,042	3,224		2,394		2,567	1,955		1,370
Increased Cost of the Final Program										
Light Commercial Trucks	-712					-581				
Long-Haul Combination Trucks				3,028					1,359	
Long-Haul Single Unit Trucks	-929	-924	-962	-1,536		-932	-931	-834	-1,203	
Motor Homes			-132	-207				-122	-187	
Other Buses		-203	-406	-1,265			-486	-520	-1,095	
Passenger Trucks	-712					-581				
Refuse Trucks			-814	-1,072				-687	-885	
School Buses		-481	-278	-430			-393	-245	-372	
Short-Haul Combination Trucks			394	-1,651				-43	-1,598	
Short-Haul Single Unit Trucks	-1,032	-1,035	-570	-857		-864	-867	-455	-684	
Transit Buses		-200	-402		-1,547		-480	-514		-1,362

Table 7-44: MY2027 Lifetime Emission-Related Repair Costs per Gasoline Vehicle, 2017 dollars

	3% Discount Rate			7% Discount Rate		
	Light HDE	Medium HDE	Heavy HDE	Light HDE	Medium HDE	Heavy HDE
FRM Baseline						
Long-Haul Single Unit Trucks	2,324	2,324	2,324	1,768	1,768	1,768
Motor Homes	431	431	431	278	278	278
Other Buses	3,111			2,234		
School Buses	832			559		
Short-Haul Single Unit Trucks	1,304	1,304	1,304	955	955	955
Transit Buses	3,074			2,208		
FRM Baseline+Final Program						
Long-Haul Single Unit Trucks	1,831	1,831	1,831	1,271	1,271	1,271
Motor Homes	275	275	275	156	156	156
Other Buses	2,898			1,917		
School Buses	442			252		
Short-Haul Single Unit Trucks	764	764	764	483	483	483
Transit Buses	2,865			1,895		
Increased Cost of the Final Program						
Long-Haul Single Unit Trucks	-493	-493	-493	-497	-497	-497
Motor Homes	-156	-156	-156	-122	-122	-122
Other Buses	-212			-317		
School Buses	-390			-306		
Short-Haul Single Unit Trucks	-540	-540	-540	-471	-471	-471
Transit Buses	-210			-313		

Table 7-45: MY2027 Lifetime Emission-Related Repair Costs per CNG Vehicle, 2017 dollars

	3% Discount Rate		7% Discount Rate	
	Heavy HDE	**Urban Bus**	**Heavy HDE**	**Urban Bus**
FRM Baseline				
Long-Haul Single Unit Trucks	4,517		3,113	
Other Buses	6,966		4,537	
Refuse Trucks	4,590		3,048	
School Buses	2,368		1,542	
Short-Haul Combination Trucks	11,938		8,595	
Short-Haul Single Unit Trucks	2,912		1,975	
Transit Buses		6,532		4,529
FRM Baseline+Final Program				
Long-Haul Single Unit Trucks	1,720		1,029	
Other Buses	3,807		2,194	
Refuse Trucks	2,260		1,317	
School Buses	1,294		746	
Short-Haul Combination Trucks	7,723		5,143	
Short-Haul Single Unit Trucks	1,248		737	
Transit Buses		2,822		1,626
Increased Cost of the Final Program				
Long-Haul Single Unit Trucks	-2,797		-2,084	
Other Buses	-3,158		-2,344	
Refuse Trucks	-2,330		-1,732	
School Buses	-1,074		-797	
Short-Haul Combination Trucks	-4,215		-3,452	
Short-Haul Single Unit Trucks	-1,664		-1,238	
Transit Buses		-3,710		-2,903

7.3 Program Costs

Using the cost elements outlined above, we have estimated the costs associated with the final program as presented in the following tables. Costs are broken into two main categories: Technology Costs and Operating Costs. Technology costs are broken further into direct costs and the indirect costs elements (warranty costs, R&D costs, other indirect costs and profits) to arrive at total technology costs incurred by manufacturers (i.e., regulated entities). Operating costs are broken into urea/DEF costs (diesel only), fuel savings (gasoline only) and repair costs to arrive at total operating costs incurred by owner/operators of new MY2027 and later HD vehicles. Section 7.3.1 presents the total technology costs for the final program and the updated costs for proposed Option 2, both relative to the updated baseline case costs. Section 7.3.2 presents the operating costs, similarly grouped. Section 7.3.3 presents the total program costs relative to the baseline case for the final program and the updated proposed Option 2. Costs are presented in 2017 dollars in undiscounted annual values along with present and equivalent annualized values (PV and EAV, respectively) at both 3 and 7 percent discount rates with discounted values discounted to the 2027 calendar year.

7.3.1 **Total Technology Costs**

The tables shown here show direct manufacturing, warranty, R&D, profits, other indirect costs and total technology costs incurred by manufacturers. Values shown for a given calendar year are undiscounted values while discounted present values (PV) and equivalent annualized values (EAV) are presented at both 3 and 7 percent discount rates with values discounted to 2027. All values are shown in 2017 dollars.

Table 7-46: Technology Cost Impacts of the Final Program Relative to the Baseline Case, Millions of 2017 dollars *

Calendar Year	Direct Manufacturing Costs	Warranty Costs	R&D Costs	Other Indirect Costs	Profits	Total Technology Costs
2027	1,100	2,100	210	340	58	3,800
2028	1,100	2,100	200	320	55	3,700
2029	1,000	2,100	190	310	53	3,700
2030	1,000	2,100	51	300	52	3,500
2031	1,000	2,200	50	300	51	3,600
2032	990	2,200	49	290	50	3,600
2033	980	2,200	49	290	50	3,600
2034	980	2,300	49	290	49	3,600
2035	960	2,300	48	280	49	3,700
2036	950	2,300	48	280	48	3,700
2037	950	2,400	48	280	48	3,700
2038	950	2,400	48	280	48	3,700
2039	950	2,500	47	280	48	3,800
2040	950	2,500	47	280	48	3,800
2041	950	2,500	47	280	48	3,900
2042	950	2,600	47	280	48	3,900
2043	950	2,600	47	280	48	3,900
2044	950	2,700	48	280	48	4,000
2045	950	2,700	48	280	48	4,100
PV, 3%	14,000	33,000	1,100	4,200	720	53,000
PV, 7%	10,000	24,000	900	3,000	520	38,000
EAV, 3%	990	2,300	78	290	50	3,700
EAV, 7%	1,000	2,300	87	290	51	3,700

* Values show 2 significant digits. Note that the Information Collection Request costs addressed in Section XII of the preamble would fall within the "Other" indirect costs shown here.

Table 7-47: Technology Cost Impacts of the Updated Proposed Option 2 Relative to the Baseline Case, Millions of 2017 dollars *

Calendar Year	Direct Manufacturing Costs	Warranty Costs	R&D Costs	Other Indirect Costs	Profits	Total Technology Costs
2027	1,100	370	190	340	58	2,100
2028	1,100	370	180	320	55	2,000
2029	1,000	370	180	310	53	1,900
2030	1,000	370	51	300	52	1,800
2031	1,000	370	50	300	51	1,800
2032	990	380	49	290	50	1,800
2033	980	380	49	290	50	1,700
2034	980	380	49	290	49	1,700
2035	960	390	48	280	49	1,700
2036	950	390	48	280	48	1,700
2037	950	390	48	280	48	1,700
2038	950	400	48	280	48	1,700
2039	950	400	47	280	48	1,700
2040	950	410	47	280	48	1,700
2041	950	410	47	280	48	1,700
2042	950	420	47	280	48	1,700
2043	950	420	47	280	48	1,700
2044	950	430	48	280	48	1,800
2045	950	440	48	280	48	1,800
PV, 3%	14,000	5,600	1,100	4,200	720	26,000
PV, 7%	10,000	4,000	850	3,000	520	19,000
EAV, 3%	990	390	75	290	50	1,800
EAV, 7%	1,000	390	82	290	51	1,800

* Values show 2 significant digits. Note that the Information Collection Request costs addressed in Section XII of the preamble would fall within the "Other" indirect costs shown here.

7.3.2 Total Operating Costs

The tables shown here show emission repair costs, urea costs, pre-tax fuel costs and total operating costs incurred by owner/operators of new MY 2027 and later HD vehicles. Values shown for a given calendar year are undiscounted values while discounted values are presented at both 3 and 7 percent discount rates. All values are shown in 2017 dollars.

Note that some values are shown as negative costs, or savings. This is expected with respect to the fuel costs since the ORVR requirements are expected to reduce gasoline consumption due to the capture of previously evaporated emissions. With respect to the emission repair costs, new MY 2027 and later HD vehicles will have no emission-related repair costs in the early years (MYs 2027 through 2030) since all of those vehicles will be covered under warranty. However, by MY 2031 and later, those early compliant vehicles will begin to experience emission-related repairs thereby countering the lack of emission-related repairs on new MY 2031 and later vehicles. This pattern will continue, with older vehicles eventually experiencing emission-related repairs that outweigh the comparatively smaller number of new vehicles covered under warranty. This explains the lower magnitude of emission repair savings in the later years shown in the tables.

Table 7-48: Operating Cost Impacts of the Final Program Relative to the Baseline Case, Millions of 2017 dollars *

Calendar Year	Emission Repair Costs	Urea Costs	Pre-tax Fuel Costs	Total Operating Costs
2027	0	57	-0.39	57
2028	-47	120	-0.82	70
2029	-300	180	-1.3	-120
2030	-430	250	-1.7	-190
2031	-500	330	-2.2	-170
2032	-570	410	-2.7	-160
2033	-610	470	-3.4	-140
2034	-640	530	-4.1	-110
2035	-660	580	-4.8	-82
2036	-660	630	-5.4	-38
2037	-600	680	-6.0	65
2038	-540	720	-6.6	170
2039	-490	760	-7.2	270
2040	-450	800	-7.8	340
2041	-410	840	-8.3	410
2042	-390	870	-8.8	470
2043	-370	910	-9.3	530
2044	-350	940	-9.7	570
2045	-340	970	-10	620
PV, 3%	-6,200	7,700	-69	1,400
PV, 7%	-4,300	4,900	-43	600
EAV, 3%	-430	540	-4.8	99
EAV, 7%	-420	480	-4.2	58

* Values show 2 significant digits.

Table 7-49: Operating Cost Impacts of the Updated Proposed Option 2 Relative to the Baseline Case, Millions of 2017 dollars *

Calendar Year	Emission Repair Costs	Urea Costs	Pre-tax Fuel Costs	Total Operating Costs
2027	0	55	-0.39	55
2028	-47	110	-0.82	65
2029	-190	170	-1.3	-17
2030	-250	240	-1.7	-17
2031	-300	310	-2.2	-0.079
2032	-220	370	-2.7	160
2033	-140	420	-3.4	280
2034	-71	470	-4.1	400
2035	-8.4	520	-4.8	500
2036	62	560	-5.4	610
2037	150	600	-6.0	740
2038	230	640	-6.6	860
2039	300	670	-7.2	970
2040	350	710	-7.8	1,100
2041	400	740	-8.3	1,100
2042	430	770	-8.8	1,200
2043	460	800	-9.3	1,300
2044	490	830	-9.7	1,300
2045	510	860	-10	1,400
PV, 3%	1,100	6,800	-69	7,900
PV, 7%	310	4,400	-43	4,700
EAV, 3%	76	480	-4.8	550
EAV, 7%	30	430	-4.2	450

* Values show 2 significant digits.

Note that the ORVR requirements will result in previously evaporated gasoline being used by in the engines of gasoline vehicles. We have estimated the cost savings that owner/operators will experience and present those in Section 7.2.2. In this section, we also show the pre-tax fuel savings that are ultimately part of the benefit-cost analysis presented in Chapter 9. Table 7-50 shows the impacts on fuel tax revenues that will be expected from these changes under the final program.

Table 7-50: Fuel Cost and Transfer Impacts of the Final Program Relative to the Baseline Case, Millions of 2017 dollars

Calendar Year	Retail Fuel Costs	Pre-tax Fuel Costs	Tax Revenues
2027	-0.47	-0.39	-0.076
2028	-0.97	-0.82	-0.16
2029	-1.5	-1.3	-0.26
2030	-2.1	-1.7	-0.35
2031	-2.7	-2.2	-0.44
2032	-3.3	-2.7	-0.53
2033	-4.1	-3.4	-0.65
2034	-4.9	-4.1	-0.77
2035	-5.7	-4.8	-0.88
2036	-6.4	-5.4	-0.99
2037	-7.1	-6.0	-1.1
2038	-7.8	-6.6	-1.2
2039	-8.5	-7.2	-1.3
2040	-9.1	-7.8	-1.3
2041	-9.7	-8.3	-1.4
2042	-10	-8.8	-1.5
2043	-11	-9.3	-1.5
2044	-11	-9.7	-1.6
2045	-12	-10	-1.7
PV, 3%	-81	-69	-12
PV, 7%	-51	-43	-7.7
EAV, 3%	-5.6	-4.8	-0.85
EAV, 7%	-4.9	-4.2	-0.75

7.3.3 Total Program Costs

The tables shown here present technology costs, operating costs and the sum of the two for final program and the updated proposed Option 2. Values shown for a given calendar year are undiscounted values while discounted values are presented at both 3 and 7 percent discount rates. All values are shown in 2017 dollars.

Table 7-51: Total Technology & Operating Cost Impacts of the Final Program Relative to the Baseline Case, Millions of 2017 dollars

Calendar Year	Total Technology Costs	Total Operating Costs	Sum
2027	3,800	57	3,900
2028	3,700	70	3,800
2029	3,700	-120	3,600
2030	3,500	-190	3,400
2031	3,600	-170	3,400
2032	3,600	-160	3,400
2033	3,600	-140	3,500
2034	3,600	-110	3,500
2035	3,700	-82	3,600
2036	3,700	-38	3,600
2037	3,700	65	3,800
2038	3,700	170	3,900
2039	3,800	270	4,000
2040	3,800	340	4,200
2041	3,900	410	4,300
2042	3,900	470	4,400
2043	3,900	530	4,500
2044	4,000	570	4,600
2045	4,100	620	4,700
PV, 3%	53,000	1,400	55,000
PV, 7%	38,000	600	39,000
EAV, 3%	3,700	99	3,800
EAV, 7%	3,700	58	3,800

* Values show 2 significant digits.

Table 7-52: Total Technology & Operating Cost Impacts of the Updated Proposed Option 2 Relative to the Baseline Case, Millions of 2017 dollars

Calendar Year	Total Technology Costs	Total Operating Costs	Sum
2027	2,100	55	2,100
2028	2,000	65	2,100
2029	1,900	-17	1,900
2030	1,800	-17	1,800
2031	1,800	-0.079	1,800
2032	1,800	160	1,900
2033	1,700	280	2,000
2034	1,700	400	2,100
2035	1,700	500	2,200
2036	1,700	610	2,300
2037	1,700	740	2,500
2038	1,700	860	2,600
2039	1,700	970	2,700
2040	1,700	1,100	2,800
2041	1,700	1,100	2,900
2042	1,700	1,200	2,900
2043	1,700	1,300	3,000
2044	1,800	1,300	3,100
2045	1,800	1,400	3,100
PV, 3%	26,000	7,900	34,000
PV, 7%	19,000	4,700	23,000
EAV, 3%	1,800	550	2,300
EAV, 7%	1,800	450	2,300

* Values show 2 significant digits.

Chapter 7 References

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17 "Nonconformance Penalties for On-highway Heavy-duty Diesel Engines: Technical Support Document," EPA-420-R-12-014, Figure 3-1 at page 37.

18 “Tier 3 Certification Fuel Impacts Test Program, Appendix A, Table A-1,” EPA-420-R-18-004.

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Chapter 8 Estimated Benefits

8.1 Overview

The highway heavy-duty engines and vehicles subject to the final rule are significant sources of mobile source air pollution, including directly-emitted $PM_{2.5}$ as well as NO_X and VOC emissions (both precursors to ozone formation and secondarily-formed $PM_{2.5}$). The final program will reduce exhaust emissions of these pollutants from the regulated engines and vehicles, which will in turn reduce ambient concentrations of ozone and $PM_{2.5}$. Estimated emission reductions are presented in Chapter 5, and air quality impacts of the standards are presented in Chapter 6. Exposures to these pollutants are linked to adverse environmental and human health impacts, such as premature deaths and non-fatal illnesses (see Chapter 4).

This chapter describes the methods used to estimate health benefits from reducing concentrations of ozone and $PM_{2.5}$. As noted in Chapter 6, full-scale photochemical air quality modeling was performed for the proposal. No further air quality modeling has been conducted to reflect the emissions impacts of the final program. Because air quality modeling results are necessary to quantify estimates of avoided mortality and illness attributable to changes in ambient $PM_{2.5}$ and ozone, we present the benefits from the proposal as a proxy for the health benefits associated with the final program. Chapter 5 describes the differences in emissions between those used to estimate the air quality impacts of the proposal and those that will be achieved by the final program. Emission reductions associated with the final program are similar to those used in the air quality modeling conducted for the proposal (Section 5.5.4). We therefore conclude that the health benefits from the proposal are a fair characterization of those that will be achieved due to the substantial improvements in air quality attributable to the final program.

Using the air quality modeling from the proposal, we have quantified and monetized health impacts in 2045, representing projected impacts associated with a year when the program will be fully implemented and when most of the regulated fleet will have turned over. There are also benefits associated with the standards that, if quantified and monetized, would increase the total monetized benefits. These unquantified benefits are discussed in Section 8.8 of this chapter. Overall, we estimate that the final program will lead to a substantial decrease in adverse $PM_{2.5}$- and ozone-related health impacts in 2045.

The approach we used to estimate health benefits is consistent with the approach described in the technical support document (TSD) that was published for the final Revised Cross-State Air Pollution Rule (CSAPR) Update RIA. [1,2,A] Estimating the health benefits of reductions in $PM_{2.5}$ and ozone exposure begins with estimating the change in exposure for each individual and then estimating the change in each individual's risks for those health outcomes affected by exposure. The benefit of the reduction in each health risk is based on the exposed individual's willingness to pay (WTP) for the risk change, assuming that each outcome is independent of one another. The greater the magnitude of the risk reduction from a given change in concentration, the greater the individual's WTP, all else equal. The social benefit of the change in health risks equals the

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[A] On March 15, 2021, EPA finalized the Revised Cross-State Air Pollution Rule Update for the 2008 ozone National Ambient Air Quality Standards (NAAQS). Starting in the 2021 ozone season, the rule will require additional emissions reductions of nitrogen oxides (NO_X) from power plants in 12 states. https://www.epa.gov/csapr/revised-cross-state-air-pollution-rule-update.

sum of the individual WTP estimates across all of the affected individuals. We conduct this analysis by adapting primary research – specifically, air pollution epidemiology studies and economic value studies – from similar contexts. This approach is sometimes referred to as "benefits transfer." Below we describe the procedure for quantifying and monetizing the health benefits associated with reduced human exposure to $PM_{2.5}$ and ozone.

8.2 Health Impact Assessment for $PM_{2.5}$ and Ozone

There are four distinct steps the Agency follows when conducting a health impacts assessment, each of which are described in this section: (1) prepare air quality modeling data for health impacts analysis; (2) select air pollution health endpoints to quantify; (3) calculate counts of air pollution effects using a health impact function; (4) specify the health impact function with concentration-response parameters drawn from the epidemiological literature.

8.2.1 Preparing Air Quality Modeling Data for Health Impacts Analysis

In RIA Chapter 5, we present the emissions that will be reduced by the final rule, including NO_X, direct PM, and VOCs, all of which contribute to ambient concentrations of $PM_{2.5}$ and ozone. These reduced emissions will benefit public health and the environment since exposure to ozone and $PM_{2.5}$ is linked to adverse public health and environmental effects.[B] In RIA Chapter 6, we summarize the air quality modeling methods and results. These air quality results, measured in terms of ambient concentrations of $PM_{2.5}$ and ozone, are in turn associated with human populations to estimate changes in health effects. This section describes how the CMAQ modeling output was converted into a format suitable for the health impacts analysis using EPA's Environmental Benefits Mapping and Analysis Program – Community Edition (BenMAP-CE).[C]

The first step was to extract 2016 base year predicted hourly, surface-layer $PM_{2.5}$ and ozone concentrations for each grid cell directly from the standard CMAQ output files (at a 12-km by 12-km resolution). For ozone, we generated predicted ozone concentration surfaces for each of three different warm seasons defined by the underlying health studies used in the analysis: April-September, May-September, and June-August. These hourly model predictions were then combined with monitored observations obtained from the Agency's Air Quality System (AQS) to interpolate hourly ozone concentrations to 12-km by 12-km grid cells for the contiguous 48

[B] As noted in Chapter 6, full-scale photochemical air quality modeling was performed for the proposal. No further air quality modeling has been conducted to reflect the emissions impacts of the final program. Since air quality modeling results are necessary to quantify estimates of avoided mortality and illness attributable to changes in ambient $PM_{2.5}$ and ozone, we present the air quality and associated benefits from the proposal as a proxy for the health benefits associated with the final program. Chapter 5 describes the differences in emissions between those used to estimate the air quality impacts of the proposal and those that will be achieved by the final program. Emission reductions associated with the final program are similar to those used in the air quality modeling conducted for the proposal. We therefore conclude that the health benefits from the proposal are a fair characterization of those that will be achieved due to the substantial improvements in air quality attributable to the final program. We do not expect the magnitude of any differences to materially impact our cost-benefit conclusions.
[C] BenMAP-CE is an open-source computer program that calculates the number and economic value of air pollution-related deaths and illnesses. The software incorporates a database that includes many of the concentration-response relationships, population files, and health and economic data needed to quantify these impacts. More information about BenMAP-CE, including downloadable versions of the tool and associated user manuals, can be found at EPA's website www.epa.gov/benmap.

states to create gridded 2016 surfaces informed by observational data.[D,E] We then converted these warm-season hourly ozone concentrations to an ozone metric of interest, such as the daily maximum 8-hour average concentration or the daily maximum 1-hour average concentration, again consistent with the underlying health studies used in the analysis. Gridded fields of relative response factors (RRFs) were created for each ozone metric and warm season definition of interest by dividing unadjusted future year (2045) CMAQ concentrations by unadjusted 2016 base year CMAQ concentrations. Separate 12-km gridded RRFs were created for the future year base case and policy cases for each metric/season combination. Then final future year air quality surfaces were created by multiplying each of the RRF surfaces by the 2016 eVNA surface. These surfaces then served as inputs to the health impact functions of the benefits analysis, contained within BenMAP-CE.

For $PM_{2.5}$, we also used the model predictions in conjunction with observed monitor data. CMAQ generates predictions of hourly PM species concentrations for every grid. The species include a primary coarse fraction (corresponding to PM in the 2.5 to 10 micron size range), a primary fine fraction (corresponding to PM less than 2.5 microns in diameter), and several secondary particles (e.g., sulfates, nitrates, and organics). $PM_{2.5}$ is calculated as the sum of the primary fine fraction and all of the secondarily formed particles. A gridded field of $PM_{2.5}$ concentrations was created by interpolating Federal Reference Monitor ambient data and Interagency Monitoring of Protected Visual Environments (IMPROVE) ambient data. Gridded fields of $PM_{2.5}$ species concentrations were created by interpolating EPA Chemical Speciation Network (CSN) ambient data and IMPROVE data. The ambient data were interpolated to the CMAQ 12-km grid. Future-year estimates of $PM_{2.5}$ were calculated using gridded RRFs applied to gridded 2016 ambient $PM_{2.5}$ and $PM_{2.5}$ species concentrations.

The procedures for determining the RRFs are similar to those in EPA's Modeling Guidance for Demonstrating Air Quality Goals for Ozone, $PM_{2.5}$, and Regional Haze.[3] The guidance recommends that model predictions be used in a relative sense to estimate changes expected to occur in each major $PM_{2.5}$ species. The procedure for calculating future-year $PM_{2.5}$ design values is called the "Speciated Modeled Attainment Test (SMAT)." EPA used this procedure to estimate the ambient impacts of the proposal.

Table 8-1 provides ozone and $PM_{2.5}$ metrics for those grid cells in the modeled domain that enter the health impact functions for health benefits endpoints. The population-weighted average reflects the baseline levels and predicted changes for more populated areas of the nation. This measure better reflects the potential benefits through exposure changes to these populations.

[D] The 12-km grid squares contain the population data used in the health benefits analysis model, BenMAP-CE.
[E] This approach is a generalization of planar interpolation that is technically referred to as enhanced Voronoi Neighbor Averaging (eVNA) spatial interpolation. See the BenMAP-CE manual for technical details, available for download at http://www.epa.gov/benmap.

Table 8-1: Summary of CMAQ-Derived Population-Weighted Ozone and $PM_{2.5}$ Air Quality Metrics for Health Benefits Endpoints

	2045	
Statistic[a]	**Baseline**	***Change***[b]
Ozone Metric: National Population-Weighted Average (ppb)[c]		
Daily Maximum 8-Hour Average Concentration – May-September	39	*0.69*
Daily Maximum 8-Hour Average Concentration – April-September	39	*0.64*
Daily Maximum 8-Hour Average Concentration – June-August	38	*0.76*
Daily Maximum 1-Hour Average Concentration – April-September	44	*0.80*
$PM_{2.5}$ Metric: National Population-Weighted Average ($\mu g/m^3$)[c]		
Annual Average Concentration	7.3	*0.034*

[a] Ozone and PM2.5 metrics were calculated at the CMAQ grid-cell level for use in health effects estimates. Ozone metrics were calculated over relevant time periods during daylight hours of each "ozone season." Note that the national, population-weighted PM2.5 and ozone air quality metrics presented in this table represent an average for the entire, gridded U.S. CMAQ domain. These are different than the population-weighted PM2.5 and ozone design value metrics presented in RIA Chapter 6, which represent the average for areas with a current air quality monitor.
[b] The change is defined as the baseline value minus the control-case value.
[c] Calculated by summing the product of the projected CMAQ grid-cell population and the estimated CMAQ grid concentration and then dividing by the total population.

8.2.2 Selecting Air Pollution Health Endpoints to Quantify

As a first step in quantifying ozone and $PM_{2.5}$-related human health impacts, the Agency consults the Integrated Science Assessment for Ozone and Related Photochemical Oxidants (Ozone ISA) and the Integrated Science Assessment for Particulate Matter (PM ISA). These two documents synthesize the toxicological, clinical and epidemiological evidence to determine whether each pollutant is causally related to an array of adverse human health outcomes associated with either short-term (i.e., hours to less than one month) or long-term (i.e., one month to years) exposure; for each outcome, the ISA reports this relationship to be "causal", "likely to be causal", "suggestive of, but not sufficient to infer, a causal relationship", "inadequate to infer a causal relationship" or "not likely to be a causal relationship". The Agency estimates the incidence of air pollution effects for those health endpoints that the ISA classified as either "causal" or "likely-to-be-causal".

In brief, the ISA for ozone found short-term exposures to ozone to have a "causal" relationship with respiratory effects, a "likely to be causal" relationship with metabolic effects and a "suggestive of, but not sufficient to infer, a causal relationship" with central nervous system effects, cardiovascular effects, and total mortality. The ISA reported that long-term exposures to ozone are "likely to be causal" for respiratory effects including respiratory mortality, and "suggestive of, but not sufficient to infer, a causal relationship" for cardiovascular effects, reproductive effects, central nervous system effects, metabolic effects, and total mortality. The PM ISA found short-term exposure to $PM_{2.5}$ to have a "causal" relationship with cardiovascular effects and mortality (i.e., premature death), "likely to be causal" relationship with respiratory effects, and "suggestive of, but not sufficient to infer, a causal relationship" with metabolic effects and nervous system effects. The ISA identified cardiovascular effects and total mortality as have a "causal" relationship with long-term exposure to $PM_{2.5}$. A "likely to be causal" relationship was determined between long-term $PM_{2.5}$ exposures and respiratory effects, nervous system effects, and cancer effects, and the evidence was "suggestive of, but not sufficient to infer, a causal relationship" for male and female reproduction and fertility effects, pregnancy and birth outcomes, and metabolic effects.

Table 8-2 reports the effects we quantified and those we did not quantify in this RIA. The list of benefit categories not quantified is not exhaustive. And, among the effects quantified, it is not always possible to completely quantify the full range of human health impacts or economic values. The table below omits health effects associated with NO_2 exposure, and any welfare effects such as acidification and nutrient enrichment; these effects are described in the Ozone and PM NAAQS RIAs and summarized later in this chapter.[4,5]

Table 8-2: Health Effects of Ambient Ozone and $PM_{2.5}$

Category	Effect	Effect Quantified	Effect Monetized	More Information
Premature mortality from exposure to $PM_{2.5}$	Adult premature mortality from long-term exposure (age 65-99 or age 30-99)	✓	✓	PM ISA
	Infant mortality (age <1)	✓	✓	PM ISA
Nonfatal morbidity from exposure to $PM_{2.5}$	Heart attacks (age > 18)	✓	✓[a]	PM ISA
	Hospital admissions—cardiovascular (ages 65-99)	✓	✓	PM ISA
	Emergency department visits— cardiovascular (age 0-99)	✓	✓	PM ISA
	Hospital admissions—respiratory (ages 0-18 and 65-99)	✓	✓	PM ISA
	Emergency room visits—respiratory (all ages)	✓	✓	PM ISA
	Cardiac arrest (ages 0-99; excludes initial hospital and/or emergency department visits)	✓	✓[a]	PM ISA
	Stroke (ages 65-99)	✓	✓[a]	PM ISA
	Asthma onset (ages 0-17)	✓	✓	PM ISA
	Asthma symptoms/exacerbation (6-17)	✓	✓	PM ISA
	Lung cancer (ages 30-99)	✓	✓	PM ISA
	Allergic rhinitis (hay fever) symptoms (ages 3-17)	✓	✓	PM ISA
	Lost work days (age 18-65)	✓	✓	PM ISA
	Minor restricted-activity days (age 18-65)	✓	✓	PM ISA
	Hospital admissions—Alzheimer's disease (ages 65-99)	✓	✓	PM ISA
	Hospital admissions—Parkinson's disease (ages 65-99)	✓	✓	PM ISA
	Other cardiovascular effects (e.g., other ages)	—	—	PM ISA[b]
	Other respiratory effects (e.g., pulmonary function, non-asthma ER visits, non-bronchitis chronic diseases, other ages and populations)	—	—	PM ISA[b]
	Other nervous system effects (e.g., autism, cognitive decline, dementia)	—	—	PM ISA[b]
	Metabolic effects (e.g., diabetes)	—	—	PM ISA[b]
	Reproductive and developmental effects (e.g., low birth weight, pre-term births, etc.)	—	—	PM ISA[b]
	Cancer, mutagenicity, and genotoxicity effects	—	—	PM ISA[b]
Mortality from exposure to ozone	Premature respiratory mortality from short-term exposure (0-99)	✓	✓	Ozone ISA
	Premature respiratory mortality from long-term exposure (age 30–99)	✓	✓	Ozone ISA
Nonfatal morbidity from exposure to ozone	Hospital admissions—respiratory (ages 65-99)	✓	✓	Ozone ISA
	Emergency department visits—respiratory (ages 0-99)	✓	✓	Ozone ISA
	Asthma onset (0-17)	✓	✓	Ozone ISA
	Asthma symptoms/exacerbation (asthmatics age 5-17)	✓	✓	Ozone ISA
	Allergic rhinitis (hay fever) symptoms (ages 3-17)	✓	✓	Ozone ISA
	Minor restricted-activity days (age 18–65)	✓	✓	Ozone ISA
	School absence days (age 5–17)	✓	✓	Ozone ISA
	Decreased outdoor worker productivity (age 18–65)	—	—	Ozone ISA[b]
	Metabolic effects (e.g., diabetes)	—	—	Ozone ISA[b]
	Other respiratory effects (e.g., premature aging of lungs)	—	—	Ozone ISA[b]
	Cardiovascular and nervous system effects	—	—	Ozone ISA[b]
	Reproductive and developmental effects	—	—	Ozone ISA[b]

[a] Valuation estimate excludes initial hospital and/or emergency department visits.

[b] Not quantified due to data availability limitations and/or because current evidence is only suggestive of causality.

8.2.3 **Calculating Counts of Air Pollution Effects Using the Health Impact Function**

We use BenMAP-CE to quantify individual risk and counts of estimated premature deaths and illnesses attributable to photochemical modeled changes in warm season average ozone concentrations and annual mean $PM_{2.5}$ for the year 2045 using a health impact function.[6] A health impact function combines information regarding the: concentration-response relationship between air quality changes and the risk of a given adverse outcome; population exposed to the air quality change; baseline rate of death or disease in that population; and, air pollution concentration to which the population is exposed.

The following provides an example of a health impact function, in this case for $PM_{2.5}$ mortality risk. We estimate counts of $PM_{2.5}$-related total deaths (yij) during each year i (i=1,…,I where I is the total number of years analyzed) among adults aged 30 and older (a) in each county in the contiguous U.S. j (j=1,…,J where J is the total number of counties) as

$$y_{ij} = \Sigma_a \, y_{ija}$$

$$y_{ija} = mo_{ija} \times (e\beta \cdot \Delta C_{ij} - 1) \times P_{ija}, \quad \text{Eq[1]}$$

where mo_{ija} is the baseline all-cause mortality rate for adults aged a=30-99 in county j in year i stratified in 10-year age groups, β is the risk coefficient for all-cause mortality for adults associated with annual average $PM_{2.5}$ exposure, C_{ij} is the annual mean $PM_{2.5}$ concentration in county j in year i, and P_{ija} is the number of county adult residents aged a=30-99 in county j in year i stratified into 5-year age groups.[F]

The BenMAP-CE tool is pre-loaded with projected population from the Woods & Poole company; cause-specific and age-stratified death rates from the Centers for Disease Control and Prevention, projected to future years; recent-year baseline rates of hospital admissions, emergency department visits and other morbidity outcomes from the Healthcare Cost and Utilization Program and other sources; concentration-response parameters from the published epidemiologic literature cited in the ISAs for fine particles and ground-level ozone; and, cost of illness or WTP unit values for each endpoint.

8.2.4 **Quantifying Ozone-Attributable Premature Mortality**

In 2008, the National Academies of Science (NAS) issued a series of recommendations to EPA regarding the procedure for quantifying and valuing ozone-related mortality due to short-term exposures.[7] Chief among these was that "…short-term exposure to ambient ozone is likely to contribute to premature deaths" and the committee recommended that "ozone-related mortality be included in future estimates of the health benefits of reducing ozone exposures…" The NAS also recommended that "…the greatest emphasis be placed on the multicity and [National Mortality and Morbidity Air Pollution Studies (NMMAPS)] …studies without exclusion of the meta-analyses."

[F] In this illustrative example, the air quality is resolved at the county level. For this RIA, we simulate air quality concentrations at 12km by 12km grids. The BenMAP-CE tool assigns the rates of baseline death and disease stored at the county level to the 12km by 12km grid cells using an area-weighted algorithm. This approach is described in greater detail in the appendices to the BenMAP-CE user manual.

Prior to the 2015 Ozone NAAQS RIA, the Agency estimated ozone-attributable premature deaths using an NMMAPS-based analysis of total mortality, two multi-city studies of cardiopulmonary and total mortality and effect estimates from three meta-analyses of non-accidental mortality.[8,9,10,11,12,13] Beginning with the 2015 Ozone NAAQS RIA, the Agency began quantifying ozone-attributable premature deaths using two newer multi-city studies of non-accidental mortality and one long-term cohort study of respiratory mortality. [14,15,16] The 2020 Ozone ISA included changes to the causality relationship determinations between short-term exposures and total mortality, as well as including more recent epidemiologic analyses of long-term exposure effects on respiratory mortality.[17] In the final 2021 CSAPR RIA, mortality from long-term exposures was estimated using the Turner et al. (2016) study extending and expanding the analysis of the American Cancer Society cohort (ACS). Mortality for short-term exposures was estimated using the risk estimate parameters from Zanobetti et al. (2008) and Katsouyanni et al. (2009), which were pooled using a consistent ozone season (May-Sept) and ozone metric (maximum daily 8-hour average).[18,19,20]

In this RIA, ozone-attributable respiratory deaths are also estimated using the risk estimate parameters described in the final 2021 CSAPR RIA. However, instead of pooling results derived from different ozone air quality surfaces, we have chosen to use only the risk estimates derived from the Katsouyanni et al. (2009) study because the study includes more cities across the United States. Furthermore, this analysis uses modeled ozone concentration data that matches the ozone metric (maximum daily 1-hour average) and season (April-September) used by Katsouyanni et al. (2009) rather than using a default ozone season (May-Sept) and metric (maximum daily 8-hour average) used in the CSAPR RIA.

8.2.5 **Quantifying $PM_{2.5}$-Attributable Premature Mortality**

When quantifying PM-attributable cases of adult mortality, we use risk estimates from two epidemiology studies examining two large population cohorts: the American Cancer Society cohort and the Medicare cohort.[21,22] The 2019 PM ISA concluded that the analyses of the ACS and Medicare cohorts provide strong evidence of an association between long-term $PM_{2.5}$ exposure and premature mortality with support from additional cohort studies. Both the ACS and Medicare cohort studies have separate and distinct attributes that make them well-suited to being used in a PM benefits assessment, so we present $PM_{2.5}$ related effects derived using relative risk estimates from both cohorts.

The PM ISA, which was reviewed by the Clean Air Scientific Advisory Committee of EPA's Science Advisory Board (SAB-CASAC), concluded that there is a causal relationship between mortality and both long-term and short-term exposure to $PM_{2.5}$ based on the entire body of scientific evidence.[23] The PM ISA also concluded that the scientific literature supports the use of a no-threshold log-linear model to portray the PM-mortality concentration-response relationship while recognizing potential uncertainty about the exact shape of the concentration-response relationship. The 2019 PM ISA, which informed the setting of the 2020 PM NAAQS, reviewed available studies that examined the potential for a population-level threshold to exist in the concentration-response relationship. Based on such studies, the ISA concluded that "evidence from recent studies reduce uncertainties related to potential co-pollutant confounding and continues to provide strong support for a linear, no-threshold concentration-response relationship".[24] Consistent with this evidence, the Agency historically has estimated health impacts above and below the prevailing NAAQS.[25, 26,27,28,29, 30,31,32,33,34,35,36]

Following this approach, we report the estimated $PM_{2.5}$-related benefits (in terms of both health impacts and monetized values) calculated using a log-linear concentration-response function that quantifies risk from the full range of simulated $PM_{2.5}$ exposures.[37,38] When setting the 2020 PM NAAQS, the EPA noted that:

"…an important consideration in characterizing the potential for additional public health improvements associated with changes in $PM_{2.5}$ exposure is whether concentration- response relationships are linear across the range of concentrations or if nonlinear relationships exist along any part of this range. Several recent studies examine this issue, and continue to provide evidence of linear, no-threshold relationships between long-term $PM_{2.5}$ exposures and all-cause and cause-specific mortality.[39] However, interpreting the shapes of these relationships, particularly at $PM_{2.5}$ concentrations near the lower end of the air quality distribution, can be complicated by relatively low data density in the lower concentration range, the possible influence of exposure measurement error, and variability among individuals with respect to air pollution health effects [85 FR 82696]."[40]

Hence, we are most confident in the size of the risks estimated from simulated $PM_{2.5}$ concentrations that coincide with the bulk of the observed PM concentrations in the epidemiological studies that are used to estimate the benefits. Likewise, we are less confident in the risk we estimate from simulated $PM_{2.5}$ concentrations that fall below the bulk of the observed data in these studies.

To give readers insight to the level of uncertainty in the estimated $PM_{2.5}$ mortality benefits at lower ambient concentrations, we report the estimated PM benefits as a distribution, identifying points along this distribution corresponding to the Lowest Reported Levels (LRLs) of each long-term exposure mortality study and the PM NAAQS (see Figure 8-1 below). In addition to adult mortality discussed above, we use risk estimates from a multi-city study to estimate PM-related infant mortality.[41]

8.3 Economic Valuation Methodology for Health Benefits

We next quantify the economic value of the ozone and $PM_{2.5}$-related deaths and illnesses estimated above. Changes in ambient concentrations of air pollution generally yield small changes in the risk of future adverse health effects for many people. Therefore, the appropriate economic measure is "willingness to pay" (WTP) for changes in risk of a health effect. For some health effects, such as hospital admissions, WTP estimates are not generally available, so we use the cost of treating or mitigating the effect. These cost-of-illness (COI) estimates are typically a lower bound estimate of the true value of reducing the risk of a health effect because they reflect the direct expenditures related to treatment, but not the value of avoided pain and suffering. The unit values applied in this analysis are provided in Table 21 of the Estimating $PM_{2.5}$- and Ozone-Attributable Health Benefits TSD.

The estimated value of avoided premature deaths ($PM_{2.5}$ plus ozone) account for between 84 percent or 94 percent of total monetized benefits depending on the studies used. The value for the projected reduction in the risk of premature mortality is the subject of continuing discussion within the economics and public policy analysis community. Following the advice of the SAB's Environmental Economics Advisory Committee (SAB-EEAC), EPA currently uses the VSL approach in calculating estimates of mortality benefits, because we believe this calculation provides the most reasonable single estimate of an individual's willingness to trade off money for changes in the risk of death.[42] The VSL approach is a summary measure for the value of small changes in the risk of death experienced by a large number of people.

EPA continues work to update its guidance on valuing mortality risk reductions, and the Agency consulted several times with the SAB-EEAC on this issue. Until updated guidance is available, the Agency determined that a single, peer-reviewed estimate applied consistently, best reflects the SAB-EEAC advice it has received. Therefore, EPA applies the VSL that was vetted and endorsed by the SAB in the Guidelines for Preparing Economic Analyses while the Agency continues its efforts to update its guidance on this issue.[43] This approach calculates a mean value across VSL estimates derived from 26 labor market and contingent valuation studies published between 1974 and 1991. The mean VSL across these studies is $4.8 million (1990$). We then adjust this VSL to account for the currency year and to account for income growth from 1990 to the analysis year. Specifically, the VSL applied in this analysis in 2017$ after adjusting for income growth is $11 million for 2045.

The Agency is committed to using scientifically sound, appropriately reviewed evidence in valuing changes in the risk of premature death and continues to engage with the SAB to update its mortality risk valuation estimates. In 2016, the Agency proposed new meta-analytic approaches for updating its estimates, which were subsequently reviewed by the SAB-EEAC.[44] EPA is taking the SAB's formal recommendations under advisement.

In valuing $PM_{2.5}$-related premature mortality, we discount the value of premature mortality occurring in future years using rates of 3 percent and 7 percent.[45] We assume that there is a multi-year "cessation" lag between changes in PM exposures and the total realization of changes in health effects. Although the structure of the lag is uncertain, EPA follows the advice of the SAB-Health Effects Subcommittee (HES) to use a segmented lag structure that assumes 30 percent of premature deaths are reduced in the first year, 50 percent over years 2 to 5, and 20 percent over the years 6 to 20 after the reduction in $PM_{2.5}$.[46] Changes in the cessation lag assumptions do not change the total number of estimated deaths but rather the timing of those deaths.

Because short-term ozone-related premature mortality occurs within the analysis year, the estimated ozone-related benefits are identical for all discount rates. When valuing changes in ozone-attributable deaths using the Turner et al. (2016) study, we follow advice provided by the SAB-HES, which found that "…there is no evidence in the literature to support a different cessation lag between ozone and particulate matter. The HES therefore recommends using the same cessation lag structure and assumptions as for particulate matter when utilizing cohort mortality evidence for ozone."[47]

These estimated health benefits do not account for the influence of future changes in the climate on ambient concentrations of pollutants.[48] For example, recent research suggests that future changes to climate may create conditions more conducive to forming ozone; the influence of changes in the climate on $PM_{2.5}$ concentrations are less clear.[49] The estimated health benefits also do not consider the potential for climate-induced changes in temperature to modify the relationship between ozone and the risk of premature death.[50,51,52]

8.4 Characterizing Uncertainty in the Estimated Benefits

In any complex analysis using estimated parameters and inputs from numerous models, there are likely to be many sources of uncertainty. This analysis is no exception. The health benefits TSD that accompanied the final Revised Cross-State Air Pollution Rule (CSAPR) Update RIA details our approach to characterizing uncertainty in both quantitative and qualitative terms. That TSD describes the sources of uncertainty associated with key input parameters including emissions inventories, air quality data from models (with their associated parameters and inputs), population data, population estimates, health effect estimates from epidemiology studies, economic data for monetizing benefits, and assumptions regarding the future state of the country (i.e., regulations, technology, and human behavior). Each of these inputs is uncertain and affects the size and distribution of the estimated benefits. When the uncertainties from each stage of the analysis are compounded, even small uncertainties can have large effects on the total quantified benefits.

To characterize uncertainty and variability in this assessment, we incorporate three quantitative analyses, which are described in greater detail within the TSD (Sections 6.1 and 6.2):

- A Monte Carlo assessment that accounts for random sampling error and between study variability in the epidemiological and economic valuation studies;
- The quantification of PM- and ozone-related mortality using alternative mortality effect estimates drawn from different studies; and
- Presentation of 95th percentile confidence interval around each risk estimate.

Quantitative characterization of other sources of uncertainties are also discussed in Sections 6.1 and 6.2 of the TSD:

- For $PM_{2.5}$-related adult all-cause mortality:
  - The distributions of air quality concentrations experienced by the original cohort population (TSD Section 6.1.2.1);
  - Methods of estimating and assigning exposures in epidemiologic studies (TSD Section 6.1.2.2);
  - Confounding by ozone (TSD Section 6.1.2.3); and
  - The statistical technique used to generate hazard ratios in the epidemiologic study (TSD Section 6.1.2.4).
- For ozone-related mortality:
  - Confounding by $PM_{2.5}$ in the long-term ozone-attributable respiratory mortality risk estimate (TSD Section 6.2.2.1);
  - Potential threshold analysis in the short-term ozone-attributable respiratory mortality risk estimate (TSD Section 6.2.2.2.1); and
  - Confounding by $PM_{2.5}$ in the short-term ozone-attributable respiratory mortality risk estimate (TSD Section 6.2.2.2.2).
- Plausible alternative risk estimates for asthma onset in children (TSD Sections 6.1.3 and 6.2.4), cardiovascular hospital admissions (TSD Section 6.1.4,), and respiratory hospital admissions (TSD Section 6.1.5)
- Effect modification of $PM_{2.5}$- and ozone attributable health effects in at-risk populations (TSD Sections 6.1.6 and 6.2.5).

Quantitative consideration of baseline incidence rates and economic valuation estimates are provided in Sections 6.3 and 6.4 of the TSD, respectively. Qualitative discussions of various sources of uncertainty can be found in Section 6.5 of the TSD.

Below are key assumptions underlying the estimates for $PM_{2.5}$-related premature mortality, followed by key uncertainties associated with estimating the number and value of ozone-related premature deaths.

- We assume that all fine particles, regardless of their chemical composition, are equally potent in causing premature mortality. This is an important assumption because $PM_{2.5}$ varies considerably in composition across sources, but the scientific evidence is not yet sufficient to allow differentiation of effect estimates by particle type. The PM ISA, which was reviewed by CASAC, concluded that "across exposure durations and health effects categories … the evidence does not indicate that any one source or component is consistently more strongly related with health effects than $PM_{2.5}$ mass."[53]
- We assume that the health impact function for fine particles is log-linear down to the lowest air quality levels modeled in this analysis. Thus, the estimates include health benefits from reducing fine particles in areas with varied concentrations of $PM_{2.5,}$ including both regions that are in attainment with the fine particle standard and those that do not meet the standard down to the lowest modeled concentrations. The PM ISA concluded that "the majority of evidence continues to indicate a linear, no-threshold concentration-response relationship for long-term exposure to $PM_{2.5}$ and total (nonaccidental) mortality."[54]
- We assume that there is a "cessation" lag between the change in PM exposures and the total realization of changes in mortality effects. Specifically, we assume that some of the incidences of premature mortality related to $PM_{2.5}$ exposures occur in a distributed fashion over the 20 years following exposure based on the advice of the SAB-HES, which affects the valuation of mortality benefits at different discount rates. The above assumptions are subject to uncertainty.[55] Similarly, we assume there is a cessation lag between the change in PM exposures and both the development and diagnosis of lung cancer.
- We assume that there is no "cessation" lag between the change in ozone exposures and the total realization of changes in long-term mortality effects. The 20-year segmented lag for $PM_{2.5}$ accounts for the onset of cardiovascular-related mortality, an outcome which is not relevant to the long-term ozone respiratory mortality estimated here. There is no alternative empirical estimate of the cessation lag for long-term exposure to ozone.
- We use a log-linear impact function without a threshold in modeling short-term ozone-related mortality. However, we acknowledge reduced confidence in specifying the shape of the concentration-response relationship in the range of $\leq$ 40ppb and below. [56] Thus, the benefits estimates include health benefits from reducing ozone in areas with varied concentrations of ozone down to the lowest modeled concentrations.

In this analysis, we plot estimated PM-related deaths according to where they occur along the distribution of baseline $PM_{2.5}$ annual mean concentrations (Figure 8-1). Displaying the data in such a way allows readers to visualize the portion of population exposed to annual mean $PM_{2.5}$ levels at or above different concentrations, which provides some insight into the level of uncertainty in the estimated $PM_{2.5}$ mortality benefits. EPA does not view the level of the PM NAAQS or the lowest concentration levels reported in the mortality studies as concentration thresholds below which we would not quantify health benefits of air quality improvements.[G] Rather, the $PM_{2.5}$-attributable benefits estimates reported in this RIA are the most appropriate estimates because they reflect the full range of air quality concentrations associated with the emission reduction program being evaluated. The 2019 PM ISA concluded that the scientific evidence collectively is sufficient to conclude that there is a causal relationship between long-term $PM_{2.5}$ exposures and mortality and that overall, the studies support the use of a no-threshold log-linear model to estimate mortality attributed to long-term $PM_{2.5}$ exposure.

Figure 8-1 compares the percentage of the population and PM-related deaths to the annual mean $PM_{2.5}$ concentrations in the baseline for the year 2045. The figure identifies the LRL for each of the major cohort studies and the annual mean $PM_{2.5}$ NAAQS of 12 µg/m^3. For Turner et al. (2016), the LRL is 2.8 µg/m^3 and for Di et al. (2017), the LRL is 0.02 µg/m^3.[H] As PM-related mortality quantified using risk estimates from the Di et al. (2017) and Turner et al. (2016) are within 5 percent of one another, in the interest of clarity and simplicity, we present the results estimated using the risk estimate from Turner et al. (2016) alone in Figure 8-1. Additional information on low concentration exposures in Turner et al. (2016) and Di et al. (2017) can be found in section 6.1.2.1 of the Estimating $PM_{2.5}$- and Ozone-Attributable Health Benefits TSD. The air quality modeling predicts $PM_{2.5}$ concentrations to be at or below the level of the annual $PM_{2.5}$ NAAQS (12 µg/m^3) in most locations in 2045. As noted in RIA Chapter 6.4.2, we are more confident in the projected changes in annual mean $PM_{2.5}$ concentrations than we are in the projected absolute $PM_{2.5}$ concentrations in 2045.

[G] For a summary of the scientific review statements regarding the lack of a threshold in the $PM_{2.5}$-mortality relationship, see the TSD entitled *Summary of Expert Opinions on the Existence of a Threshold in the Concentration-Response Function for $PM_{2.5}$-related Mortality* (U.S. EPA, 2010).

[H] Turner et al. (2016) estimated $PM_{2.5}$ exposures using both a hierarchical Bayesian space–time model (HBM) and a land use regression model with Bayesian Maximum Entropy kriging of residuals (LURBME). As such, two LRLs are reported in the paper, 2.8 µg/m^3 and 1.4 µg/m^3. As the HBM risk estimate was used in the final 2021 CSAPR RIA, the HBM LRL is presented here.

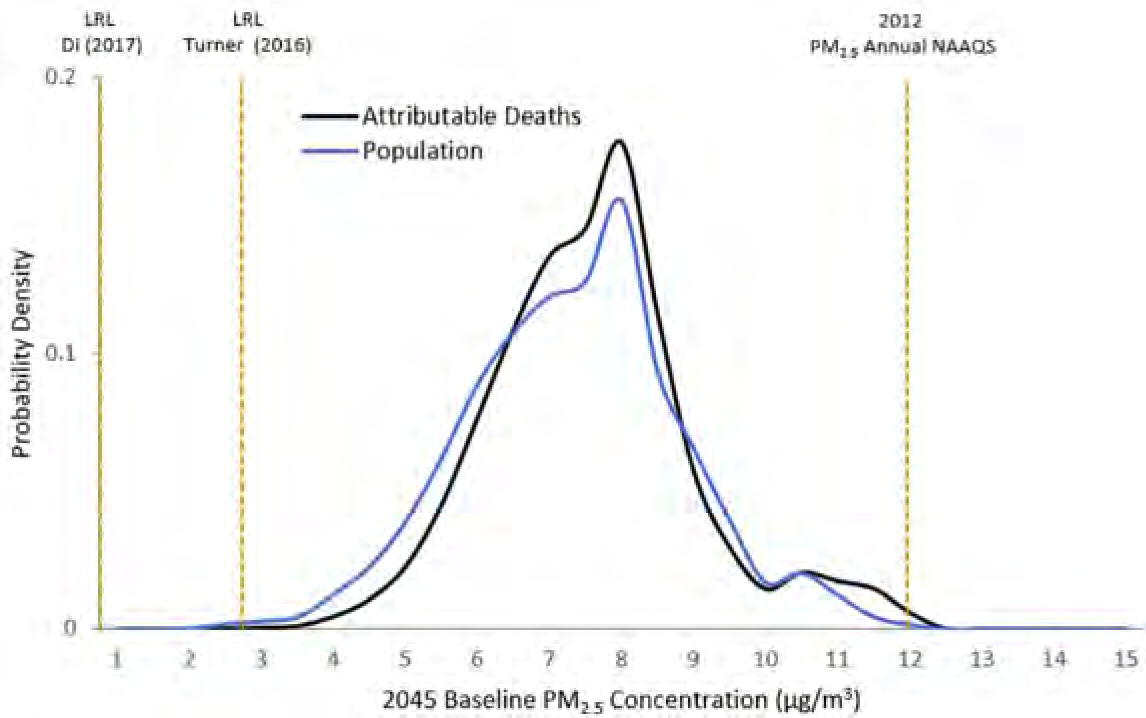

Figure 8-1: Estimated Percentage of $PM_{2.5}$-Related Deaths (Turner et al. 2016) and Number of Individuals Exposed (30+) by Annual Mean $PM_{2.5}$ Level in 2045

8.5 Estimated Number and Economic Value of Health Benefits

Below we report the estimated number of reduced premature deaths and illnesses in 2045 attributable to the standards along with the 95 percent confidence interval (Table 8-3 and Table 8-4). The number of reduced estimated deaths and illnesses are calculated from the sum of individual reduced mortality and illness risk across the population. Table 8-5 reports the estimated individual economic value of avoided premature deaths and illnesses relative to the baseline along with the 95 percent confidence interval. **Error! Reference source not found.**

Table 8-6 reports total benefits associated with the standards in 2045, reflecting alternative combinations of the economic value of $PM_{2.5}$- and ozone-related premature deaths summed with the economic value of illnesses for each discount rate.

Table 8-3: Estimated Avoided $PM_{2.5}$ Mortality and Illnesses in 2045 (95% Confidence Interval) [a,b]

		Avoided Health Incidence
Avoided premature mortality		
	Turner et al. (2016) – Ages 30+	740 (500 to 980)
	Di et al. (2017) – Ages 65+	800 (780 to 830)
	Woodruff et al. (2008) – Ages < 1	4.1 (-2.6 to 11)
Non-fatal heart attacks among adults		
Short-term exposure	Peters et al. (2001)	790 (180 to 1,400)
	Pooled estimate	85 (31 to 230)
Morbidity effects		
Long-term exposure	Asthma onset	1,600 (1,500 to 1,600)
	Allergic rhinitis symptoms	10,000 (2,500 to 18,000)
	Stroke	41 (11 to 70)
	Lung cancer	52 (16 to 86)
	Hospital Admissions - Alzheimer's disease	400 (300 to 500)
	Hospital Admissions - Parkinson's disease	43 (22 to 63)
Short-term exposure	Hospital admissions-cardiovascular	110 (76 to 130)
	ED visits- cardiovascular	210 (-82 to 500)
	Hospital admissions - respiratory	68 (23 to 110)
	ED visits - respiratory	400 (78 to 830)
	Asthma symptoms	210,000 (-100,000 to 520,000)
	Minor restricted-activity days	460,000 (370,000 to 550,000)
	Cardiac arrest	10 (-4.2 to 24)
	Lost work days	78,000 (66,000 to 90,000)

[a] Values rounded to two significant figures.
[b] $PM_{2.5}$ exposure metrics are not presented here because all PM health endpoints are based on studies that used daily 24-hour average concentrations. Annual exposures are estimated using daily 24-hour average concentrations.

Table 8-4: Estimated Avoided Ozone Mortality and Illnesses in 2045 (95% Confidence Interval)[a]

		Metric and Season[b]	Avoided Health Incidence
Avoided premature mortality			
Long-term exposure	Turner et al. (2016)	MDA8 April-September	2,100 (1,400 to 2,700)
Short-term exposure	Katsouyanni et al (2009)	MDA1 April-September	120 (-69 to 300)
Morbidity effects			
Long-term exposure	Asthma onset[c]	MDA8 June-August	16,000 (14,000 to 18,000)
Short-term exposure	Allergic rhinitis symptoms	MDA8 May-September	88,000 (47,000 to 130,000)
	Hospital admissions - respiratory	MDA1 April-September	350 (-91 to 770)
	ED visits - respiratory	MDA8 May-September	5,100 (1,400 to 11,000)
	Asthma symptoms - Cough[d]	MDA8 May-September	920,000 (-50,000 to 1,800,000)
	Asthma symptoms - Chest Tightness[d]	MDA8 May-September	770,000 (85,000 to 1,400,000)
	Asthma symptoms - Shortness of Breath[d]	MDA8 May-September	390,000 (-330,000 to 1,100,000)
	Asthma symptoms - Wheeze[d]	MDA8 May-September	730,000 (-57,000 to 1,500,000)
	Minor restricted-activity days[d]	MDA1 May-September	1,600,000 (650,000 to 2,600,000)
	School absence days	MDA8 May-September	1,100,000 (-150,000 to 2,200,000)

[a] Values rounded to two significant figures.

[b] MDA8 – maximum daily 8-hour average; MDA1 – maximum daily 1-hour average. Studies of ozone vary with regards to season, limiting analyses to various definitions of summer (e.g., April-September, May-September or June-August). These differences can reflect state-specific ozone seasons, EPA-defined seasons or another seasonal definition chosen by the study author. The paucity of ozone monitoring data in winter months complicates the development of full year projected ozone surfaces and limits our analysis to only warm seasons.

[c] The underlying metric associated with this risk estimate is daily 8-hour average from 10am – 6pm (AVG8); however, we ran the study with a risk estimate converted to MDA8.

[d] Applied risk estimate derived from full year exposures to estimates of ozone across a May-September ozone season. When risk estimates based on full-year, long-term ozone exposures are applied to warm season air quality projections, the resulting benefits assessment may underestimate impacts, due to a shorter timespan for impacts to accrue.

Table 8-5: Estimated Economic Value of $PM_{2.5}$- and Ozone-Attributable Premature Mortality and Illnesses in 2045 (95% Confidence Interval; millions of 2017$)[a]

<table>
<tr><th colspan="3"></th><th>3% Discount Rate</th><th>7% Discount Rate</th></tr>
<tr><td colspan="5">Avoided premature mortality</td></tr>
<tr><td rowspan="3">$PM_{2.5}$</td><td rowspan="2">Long-term exposure</td><td>Turner et al. (2016)</td><td>$8,100
($710 to $22,000)</td><td>$7,300
($640 to $20,000)</td></tr>
<tr><td>Di et al. (2017)</td><td>$8,800
($790 to $23,000)</td><td>$7,900
($710 to $21,000)</td></tr>
<tr><td>Short-term exposure</td><td>Woodruff et al (2008)</td><td colspan="2">$50
(-$28 to $200)</td></tr>
<tr><td rowspan="2">Ozone</td><td>Long-term exposure</td><td>Turner et al. (2016)</td><td>$23,000
($2,000 to $61,000)</td><td>$20,000
($1,800 to $55,000)</td></tr>
<tr><td>Short-term exposure</td><td>Katsouyanni et al (2009)</td><td colspan="2">$1,500
(-$720 to $5,700)</td></tr>
<tr><td colspan="5">$PM_{2.5}$- related non-fatal heart attacks among adults</td></tr>
<tr><td colspan="2" rowspan="2">Short-term exposure</td><td>Peters et al. (2001)</td><td>$62
($14 to $110)</td><td>$60
($14 to $100)</td></tr>
<tr><td>Pooled estimate</td><td>$6.7
($2.4 to $18)</td><td>$6.4
($2.3 to $17)</td></tr>
<tr><td colspan="5">Morbidity effects</td></tr>
<tr><td colspan="2" rowspan="6">Long-term exposure</td><td>Asthma onset
($PM_{2.5}$ & O_3)</td><td>$820
($720 to $940)</td><td>$520
($440 to $580)</td></tr>
<tr><td>Allergic rhinitis symptoms
($PM_{2.5}$ & O_3)</td><td colspan="2">$61
($31 to $91)</td></tr>
<tr><td>Stroke
($PM_{2.5}$)</td><td colspan="2">$1.4
($0.37 to $2.5)</td></tr>
<tr><td>Lung cancer
($PM_{2.5}$)</td><td>$1.4
($0.43 to $2.4)</td><td>$1.1
($0.33 to $1.8)</td></tr>
<tr><td>Hospital Admissions - Alzheimer’s disease ($PM_{2.5}$)</td><td colspan="2">$5.0
($3.8 to $6.3)</td></tr>
<tr><td>Hospital Admissions - Parkinson’s disease ($PM_{2.5}$)</td><td colspan="2">$0.57
($0.29 to $0.84)</td></tr>
<tr><td colspan="2" rowspan="9">Short-term exposure</td><td>Hospital admissions - cardiovascular
($PM_{2.5}$)</td><td colspan="2">$1.7
($1.2 to $2.1)</td></tr>
<tr><td>ED visits - cardiovascular
($PM_{2.5}$)</td><td colspan="2">$0.25
(-$0.098 to $0.59)</td></tr>
<tr><td>Hospital admissions - respiratory
($PM_{2.5}$ & O_3)</td><td colspan="2">$14
(-$3.2 to $30)</td></tr>
<tr><td>ED visits - respiratory
($PM_{2.5}$ & O_3)</td><td colspan="2">$5.0
($1.4 to $10)</td></tr>
<tr><td>Asthma symptoms
($PM_{2.5}$ & O_3)</td><td colspan="2">$650
(-$79 to $1,400)</td></tr>
<tr><td>Minor restricted-activity days
($PM_{2.5}$ & O_3)</td><td colspan="2">$170
($67 to $300)</td></tr>
<tr><td>Cardiac arrest
($PM_{2.5}$)</td><td>$0.38
(-$0.16 to $0.87)</td><td>$0.38
(-$0.15 to $0.86)</td></tr>
<tr><td>Lost work days
($PM_{2.5}$)</td><td colspan="2">$14
($12 to $16)</td></tr>
<tr><td>School absence days
(O_3)</td><td colspan="2">$120
(-$16 to $240)</td></tr>
</table>

[a] Values rounded to two significant figures.

Table 8-6: Total Ozone and $PM_{2.5}$-Attributable Benefits in 2045 (95% Confidence Interval; billions of 2017$)[a,b]

	Total Annual Benefits in 2045		
3% Discount Rate	$12 ($0.72 to $31)[c]	and	$33 ($3.5 to $87)[d]
7% Discount Rate	$10 ($0.37 to $28)[c]	and	$30 ($3.0 to $78)[d]

[a] The benefits associated with the standards presented here do not include the full complement of health and environmental benefits that, if quantified and monetized, would increase the total monetized benefits.
[b] Values rounded to two significant figures. The two benefits estimates separated by the word "and" signify that they are two separate estimates. The estimates do not represent lower- and upper-bound estimates though they do reflect a grouping of estimates that yield more and less conservative benefit totals. They should not be summed.
[c] Sum of benefits using the Katsouyanni et al. (2009) short-term exposure ozone respiratory mortality risk estimate and the Turner et al. (2016) long-term exposure $PM_{2.5}$ all-cause risk estimate.
[d] Sum of benefits using the Turner et al. (2016) long-term exposure ozone respiratory mortality risk estimate and the Di et al. (2017) long-term exposure $PM_{2.5}$ all-cause risk estimate.

8.6 Present Value of Total Benefits

The full-scale benefits analysis reflects spatially and temporally allocated emissions inventories generated using SMOKE/MOVES (see RIA Chapter 5), photochemical air quality modeling using CMAQ (see RIA Chapter 6), and $PM_{2.5}$ and ozone benefits generated using BenMAP-CE, all for conditions projected to occur in calendar year 2045. As we presented in RIA Chapter 5 and Chapter 7, national estimates of year-over-year emissions and program costs were generated from program implementation to a year when the program will be fully phased-in and the vehicle fleet will be approaching full turnover (2027-2045). The time and resources required to conduct air quality modeling to support a full-scale benefits analysis for all analysis years from 2027 to 2044 precluded the Agency from conducting benefits analyses comparable to the calendar year 2045 benefits analysis. Instead, we have used a reduced-form approach to scale total benefits in 2045 back to 2027 (including interim years) using projected reductions in year-over-year NO_X emissions so that we can estimate the present value of the stream of estimated benefits

This approach is similar to the Agency's method for estimating "benefits-per-ton" values over time.[57] For interim analysis years without air quality modeling, we input the program's 2045 air quality data into BenMAP-CE to generate benefits that occur in earlier analysis years. This approach allows us to calculate the benefits for interim years by adjusting for changes in population, baseline mortality incidence, and income growth over time. Table 8-7 displays the data used to generate benefits that reflect input data for years 2027, 2030, 2035, 2040, and 2045.[I]

[I] Interim analysis years chosen for computational efficiency at reasonable intervals.

Table 8-7: Benefits Inputs that Change Over Time used to Calculate Year-over-Year Estimates

Analysis Year	Air Quality Modeling & Emissions Year	Population Year	Baseline Mortality Incidence Year	Income Growth Year	Currency Year
2027	2045	2027	2025	2027	2017
2030		2030	2030	2030	
2035		2035	2035	2035	
2040		2040	2040	2040	
2045		2045	2045	2045	

We next calculate the total monetized benefits estimated for each of the analysis years and divide them by the estimated tons of NO_X emissions projected to be controlled by the rule in 2045 (see RIA Chapter 5, Table 5-29) to generate "benefit-per-ton" values that reflect benefits inputs consistent with the analysis year.[J] Because NO_X is the dominant pollutant controlled by the program, we make a simplifying assumption that total PM and ozone benefits can be scaled by NO_X emissions, even though emissions of other pollutants are controlled in smaller amounts by the rule (see RIA Chapter 5, Table 5-21). By using the 2045 air quality modeling surfaces for the earlier analysis years, we also assume that the spatial distribution of NO_X emissions reductions does not change over time. While there may be localized differences in the rate of fleet turnover due to state or local incentive programs, we do not currently have sufficient data to incorporate those differences into our analyses and believe that they would generally even out over time (as noted in RIA Chapter 5, we use MOVES default vehicle activity data, including data on age of the fleet or turnover).

To estimate total benefits for the interim years, we multiply the benefit-per-ton values estimated for each earlier analysis year by the NO_X emissions projected to be controlled in that same year (2027, 2030, 2035, and 2040; see RIA Chapter 5, Table 5-31). For intervening years between the analysis years, we linearly interpolate total benefits.

Table 8-8 and Table 8-9 present the undiscounted stream of scaled annual total benefits of the rule between 2027 and 2045. We also estimate the present value and annualized value of the stream of benefits in these tables. Table 8-8 presents total benefits as the sum of short-term ozone respiratory mortality benefits for all ages, long-term $PM_{2.5}$ all-cause mortality benefits for ages 30 and above, and all monetized avoided illnesses.[58,59] Table 8-9 presents total benefits as the sum of long-term ozone respiratory mortality benefits for ages 30 and above, long-term $PM_{2.5}$ all-cause mortality benefits for ages 65 and above, and all monetized avoided illnesses.[60,61] The present value of benefits in both tables is discounted back to year 2027 using both a 3 percent and 7 percent discount rate.

[J] Note that these "benefit-per-ton" values are internally consistent with the air quality modeling conducted for the proposal in 2045. They are appropriate for scaling benefits of the program but should not be used outside of the context of this rulemaking analysis.

Table 8-8: Undiscounted Stream and Present Value of Human Health Benefits from 2027 through 2045: Monetized Benefits Quantified as Sum of Short-Term Ozone Respiratory Mortality Ages 0-99, and Long-Term $PM_{2.5}$ All-Cause Mortality Ages 30+ (Discounted at 3% and 7%; billions of 2017$)[a,b]

	Monetized Benefits	
	3% Discount	**7% Discount**
2027	$0.66	$0.59
2028	$1.4	$1.2
2029	$2.1	$1.9
2030	$2.8	$2.6
2031	$3.8	$3.4
2032	$4.8	$4.3
2033	$5.5	$5.0
2034	$6.2	$5.6
2035	$6.9	$6.2
2036	$7.5	$6.7
2037	$8.0	$7.2
2038	$8.6	$7.7
2039	$9.1	$8.2
2040	$9.6	$8.7
2041	$10	$9.0
2042	$10	$9.4
2043	$11	$9.7
2044	$11	$10
2045[c]	$12	$10
Present Value	$91	$53
Annualized Value	$6.3	$5.1

[a] The benefits associated with the standards presented here do not include the full complement of health and environmental benefits that, if quantified and monetized, would increase the total monetized benefits.
[b] Benefits calculated as value of avoided: $PM_{2.5}$-attributable deaths (quantified using a concentration-response relationship from the Turner et al. 2016 study); Ozone-attributable deaths (quantified using a concentration-response relationship from the Katsouyanni et al. 2009 study); and $PM_{2.5}$ and ozone-related morbidity effects.
[c] Year in which $PM_{2.5}$ and ozone air quality was simulated (2045).

Table 8-9: Undiscounted Stream and Present Value of Human Health Benefits from 2027 through 2045: Monetized Benefits Quantified as Sum of Long-Term Ozone Respiratory Mortality Ages 30+, and Long-Term $PM_{2.5}$ All-Cause Mortality Ages 65+ (Discounted at 3% and 7%; billions of 2017$)[a,b]

	Monetized Benefits	
	3% Discount	**7% Discount**
2027	$1.8	$1.6
2028	$3.7	$3.3
2029	$5.7	$5.1
2030	$7.9	$7.1
2031	$11	$9.6
2032	$13	$12
2033	$16	$14
2034	$18	$16
2035	$19	$17
2036	$21	$19
2037	$23	$21
2038	$25	$22
2039	$26	$23
2040	$28	$25
2041	$29	$26
2042	$30	$27
2043	$31	$28
2044	$32	$29
2045[c]	$33	$30
Present Value	$260	$150
Annualized Value	$18	$14

[a] The benefits associated with the standards presented here do not include the full complement of health and environmental benefits that, if quantified and monetized, would increase the total monetized benefits.
[b] Benefits calculated as value of avoided: $PM_{2.5}$-attributable deaths (quantified using a concentration-response relationship from the Turner et al. 2016 study); Ozone-attributable deaths (quantified using a concentration-response relationship from the Katsouyanni et al. 2009 study); and $PM_{2.5}$ and ozone-related morbidity effects.
[c] Year in which $PM_{2.5}$ and ozone air quality was simulated (2045).

8.7 Unquantified Benefits

In addition to the $PM_{2.5}$ and ozone-related health impacts we are unable to quantify or monetize in Table 8-2, there are additional benefits associated with reductions in exposure to ambient concentrations of NO_2,[K] ecosystem benefits, and visibility improvement that EPA is not currently able to quantify due to data, resource, or methodological limitations. EPA continues to pursue data and methods to further improve our assessment of benefits that are currently unquantified. In particular, we are evaluating the feasibility of assessing impacts on ecosystem

[K] EPA is considering how to incorporate NO_2 health benefits into our rulemakings. The ISA states that a key uncertainty in understanding the relationship between non-respiratory health effects and short- or long-term exposure to NO_2 is co-pollutant confounding, particularly by other roadway pollutants. The Agency will utilize the same systematic process for selecting, quantifying, and monetizing NO_2-related health impacts as it has for $PM_{2.5}$ and ozone. This process includes: applying a criteria for identifying and selecting studies and risk estimates most appropriate to inform a benefits analysis for a RIA; identifying pollutant-attributable health effects for which the ISA reports strong evidence and that may be quantified in a benefits assessment; collecting baseline incidence and prevalence estimates and demographic information; developing appropriate economic unit values, and characterizing uncertainty with quantified benefits estimates.

services from reductions in nitrogen deposition and terrestrial acidification. RIA Chapter 4 provides a qualitative description of both the health and environmental effects of the criteria pollutants controlled by the program. These additional unquantified health and welfare benefit categories are listed in Table 8-10.

There will also be benefits associated with reductions in air toxic pollutant emissions that result from the program (see RIA Chapter 4.1.6 and RIA Chapter 5.3.1), but we did not attempt to monetize those impacts. This is because currently available tools and methods to assess air toxics risk from mobile sources at the national scale are not adequate for extrapolation to incidence estimation or benefits assessment. While EPA has worked to improve these tools, there remain critical limitations for estimating incidence and assessing benefits of reducing mobile source air toxics.

Table 8-10: Unquantified Criteria Pollutant Health and Welfare Benefits Categories

Category	Effect	Effect Quantified	Effect Monetized	More Information
Improved Human Health				
Reduced incidence of morbidity from exposure to NO_2	Asthma hospital admissions	—	—	NO_2 ISA[62,a]
	Chronic lung disease hospital admissions	—	—	NO_2 ISA[a]
	Respiratory emergency department visits	—	—	NO_2 ISA[a]
	Asthma exacerbation	—	—	NO_2 ISA[a]
	Acute respiratory symptoms	—	—	NO_2 ISA[a]
	Premature mortality	—	—	NO_2 ISA[a,b,c]
	Other respiratory effects (e.g., airway hyperresponsiveness and inflammation, lung function, other ages and populations)	—	—	NO_2 ISA[b,c]
Improved Environment				
Reduced visibility impairment	Visibility in Class 1 areas	—	—	PM ISA[a]
	Visibility in residential areas	—	—	PM ISA[a]
Reduced effects on materials	Household soiling	—	—	PM ISA[a,b]
	Materials damage (e.g., corrosion, increased wear)	—	—	PM ISA[b]
Reduced effects from PM deposition (metals and organics)	Effects on individual organisms and ecosystems	—	—	PM ISA[b]
Reduced vegetation and ecosystem effects from exposure to ozone	Visible foliar injury on vegetation	—	—	Ozone ISA[a]
	Reduced vegetation growth and reproduction	—	—	Ozone ISA[a]
	Yield and quality of commercial forest products and crops	—	—	Ozone ISA[a]
	Damage to urban ornamental plants	—	—	Ozone ISA[b]
	Carbon sequestration in terrestrial ecosystems	—	—	Ozone ISA[a]
	Recreational demand associated with forest aesthetics	—	—	Ozone ISA[b]
	Other non-use effects			Ozone ISA[b]
	Ecosystem functions (e.g., water cycling, biogeochemical cycles, net primary productivity, leaf-gas exchange, community composition)	—	—	Ozone ISA[b]
Reduced effects from acid deposition	Recreational fishing	—	—	NO_X SOx ISA[63,a]
	Tree mortality and decline	—	—	NO_X SOx ISA[b]
	Commercial fishing and forestry effects	—	—	NO_X SOx ISA[b]
	Recreational demand in terrestrial and aquatic ecosystems	—	—	NO_X SOx ISA[b]
	Other non-use effects			NO_X SOx ISA[b]
	Ecosystem functions (e.g., biogeochemical cycles)	—	—	NO_X SOx ISA[b]
Reduced effects from nutrient enrichment	Species composition and biodiversity in terrestrial and estuarine ecosystems	—	—	NO_X SOx ISA[b]
	Coastal eutrophication	—	—	NO_X SOx ISA[b]
	Recreational demand in terrestrial and estuarine ecosystems	—	—	NO_X SOx ISA[b]
	Other non-use effects			NO_X SOx ISA[b]
	Ecosystem functions (e.g., biogeochemical cycles, fire regulation)	—	—	NO_X SOx ISA[b]
Reduced vegetation effects from ambient exposure to SO_2 and NO_X	Injury to vegetation from SO_2 exposure	—	—	NO_X SOx ISA[b]
	Injury to vegetation from NO_X exposure	—	—	NO_X SOx ISA[b]

[a] We assess these benefits qualitatively due to data and resource limitations for this RIA.

[b] We assess these benefits qualitatively because we do not have sufficient confidence in available data or methods.

[c] We assess these benefits qualitatively because current evidence is only suggestive of causality or there are other significant concerns over the strength of the association.

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Chapter 9 Comparison of Benefits and Costs

This chapter compares the estimated range of total monetized health benefits to total costs associated with the final rule. This chapter also presents the range of monetized net benefits (benefits minus costs) associated with the final rule. Program costs are detailed and presented in Chapter 7 of this RIA. Those costs include costs for both the new technology and the operating costs associated with that new technology, as well as costs associated with the final rule's warranty and useful life provisions. Program benefits are presented in RIA Chapter 8.[A] Those benefits are the monetized economic value of the reduction in $PM_{2.5}$- and ozone-related premature deaths and illnesses that result from reductions in NO_X emissions and directly emitted $PM_{2.5}$ attributable to implementation of the final rule.

As noted elsewhere in this RIA, the estimated benefits, costs, and net benefits do not reflect all the anticipated impacts of the final rule.

9.1 Methods

EPA presents three different benefit-cost comparisons for the final rule:

1. A future-year snapshot comparison of annual benefits and costs in the year 2045, chosen to approximate the annual health benefits that will occur in a year when the program will be fully implemented and when most of the regulated fleet will have turned over. Benefits, costs and net benefits are presented in year 2017 dollars and are not discounted. However, 3 percent and 7 percent discount rates were applied in the valuation of avoided premature deaths from long-term pollution exposure to account for a twenty-year segmented cessation lag.

2. The present value (PV) of the stream of benefits, costs and net benefits calculated for the years 2027-2045, discounted back to the first year of implementation of the final rule (2027) using both a 3 percent and 7 percent discount rate, and presented in year 2017 dollars. Note that year-over-year costs are presented in RIA Chapter 7 and year-over-year benefits can be found in RIA Chapter 8.

3. The equivalent annualized value (EAV) of benefits, costs and net benefits, representing a flow of constant annual values that, had they occurred in each year from 2027 to 2045, will yield an equivalent present value to the present value estimated in method 2 (using either a 3 percent or 7 percent discount rate). Each EAV represents a typical benefit, cost or net benefit for each year of the analysis and is presented in year 2017 dollars.

The two estimates of benefits (and net benefits) in each of these benefit-cost comparisons reflect alternative combinations of the economic value of $PM_{2.5}$- and ozone-related premature deaths summed with the economic value of illnesses for each discount rate (see Chapter 8 for more detail).

[A] As detailed in RIA Chapter 8, estimates of health benefits are based on air quality modeling conducted for the proposal, and thus differences between the proposal and final rule are not reflected in the benefits analysis. We have concluded, however, that the health benefits estimated for the proposal are a fair characterization of the benefits that will be achieved due to the substantial improvements in air quality attributable to the final rule.

9.2 Results

Table ES presents the benefits, costs and net benefits of the final rule in annual terms for year 2045, in PV terms, and in EAV terms.

Annual benefits are larger than the annual costs in 2045, with annual net benefits of $5.8 and $25 billion using a 7 percent discount rate, and $6.9 and $29 billion using a 3 percent discount rate.[B] Benefits also outweigh the costs when expressed in PV terms (net benefits of $14 and $110 billion using a 7 percent discount rate, and $36 and $200 billion using a 3 percent discount rate) and EAV terms (net benefits of $1.3 and $11 billion using a 7 percent discount rate, and $2.5 and $14 billion using a 3 percent discount rate).

Given these results, implementation of the final rule will provide society with a substantial net gain in welfare, notwithstanding the health and other benefits we were unable to quantify (see RIA Chapter 8.8 for more information about unquantified benefits). EPA does not expect the omission of unquantified benefits to impact the Agency's evaluation of the costs and benefits of the final rule, though net benefits would be larger if unquantified benefits were monetized.

Table 9-1: 2045 Annual Value, Present Value and Equivalent Annualized Value of Costs, Benefits and Net Benefits of the Final Rule (billions, 2017$)[a,b]

		3% Discount	**7% Discount**
2045	Benefits	$12 - $33	$10 - $30
	Costs	$4.7	$4.7
	Net Benefits	$6.9 - $29	$5.8 - $25
Present Value	Benefits	$91 - $260	$53 - $150
	Costs	$55	$39
	Net Benefits	$36 - $200	$14 - $110
Equivalent Annualized Value	Benefits	$6.3 - $18	$5.1 - $14
	Costs	$3.8	$3.8
	Net Benefits	$2.5 - $14	$1.3 - $11

[a] All benefits estimates are rounded to two significant figures; numbers may not sum due to independent rounding. The range of benefits (and net benefits) in this table are two separate estimates and do not represent lower- and upper-bound estimates, though they do reflect a grouping of estimates that yield more and less conservative benefits totals. The costs and benefits in 2045 are presented in annual terms and are not discounted. However, all benefits in the table reflect a 3 percent and 7 percent discount rate used to account for cessation lag in the valuation of avoided premature deaths associated with long-term exposure.

[b] The benefits associated with the standards presented here do not include the full complement of health and environmental benefits that, if quantified and monetized, would increase the total monetized benefits.

[B] The range of benefits and net benefits presented in this section reflect a combination of assumed $PM_{2.5}$ and ozone mortality risk estimates and selected discount rate.

Chapter 10 Economic Impact Analysis

This rulemaking is considered economically significant, because it is expected to have an annual impact on the economy of $100 million or more, and thus an economic analysis has been completed as part of this RIA. This rule is not expected to have measurable inflationary or recessionary effects.

The benefits to human health and the environment are discussed in Chapter 8, and the costs of the final standards are discussed in Chapter 7. The benefit-cost analysis for this rule is presented in Chapter 9. This chapter provides an analysis of the impacts of the standards on vehicle sales and employment.

10.1 Impact on Sales, Fleet Turnover and Mode Shift

As explained in Chapter 7, this rule is expected to increase the cost of heavy-duty (HD) vehicles by requiring emissions control technologies capable of controlling NO_X at lower levels than are currently permitted, as well as longer emissions warranty periods for emissions control technology components. In addition, there is an expected increase in operating costs due to an increase in the use of diesel exhaust fluid (DEF) and emission-related repair costs beyond the new useful life periods.

Three sectors expected to be most immediately affected by this action are: 1) HD vehicle and engine manufacturers, 2) HD vehicle and engine buyers, and 3) HD engine equipment suppliers (e.g., suppliers of emissions control components). Effects on industries downstream of these sectors, such as HD vehicle dealerships or delivery industries, will be relatively smaller due to the limited role of the cost of a HD vehicle in pricing in those sectors. The three sectors will also be responding to the 2016 rulemaking, “Greenhouse Gas Emissions and Fuel Efficiency Standards for Medium- and Heavy-Duty Engines and Vehicles – Phase 2” (the Phase 2 rule). The final standards will be implemented during the same time frame as the final Phase 2 rule standards. Both this rulemaking and the Phase 2 rule will require HD engine manufacturers to develop and implement improvements in engine emissions controls.

As discussed in the Phase 2 rule RIA,[1] increases in costs of HD vehicles from improved emissions controls will be likely to lead to increases in final prices for HD vehicles; the magnitude of that effect will depend on how much of the cost is passed along to potential buyers. These price increases may affect HD vehicle sales in several ways. First, as basic economic supply and demand theory suggests, higher prices are expected to reduce HD vehicle sales. Second, HD vehicle buyers may strategically seek to avoid increased prices by “pre-buying,” increasing the purchases of new vehicles before the compliance deadline for the new requirements. This might lead to an associated period immediately afterward of “low-buying,” during which purchases decrease, and thereby impact the rate of fleet turnover. A third potential effect is transportation mode shift, changing from on-highway trucking to other modes of transportation (e.g., shipping via barge or rail instead of by truck). The magnitude of each of these three categories of effects (sales, fleet turnover, mode shift) will depend on the costs. This section discusses these impacts.[C]

10.1.1 **Sales**

The effects of the final standards on HD vehicle sales depend on the magnitude of the cost increase associated with implementing improved emissions controls to comply with the requirements, and on the degree to which the costs get passed through to vehicle buyers.

As discussed in Chapter 7, an increase in cost of HD vehicles will result from the standards requiring the use of emissions control technologies capable of controlling NO_X at lower levels, as well as imposing longer useful life and emissions warranty periods for emissions control technology components. While the final requirement for longer emissions warranty periods will likely increase the purchase price of new HD vehicles, the corresponding lengthened useful life periods are expected to make emissions control technology components more durable. More durable components coupled with manufacturers paying for repairs during a longer warranty period will in turn reduce repair costs, which, especially in the long term, may increase (or reduce the decrease in) sales of new HD vehicles due to fleets and independent owner-operators being inclined to purchase vehicles with lower repair costs.[D] The exact purchase behavior of fleet owners and independent owner-operators is challenging to predict, particularly in the time period immediately after new standards go into effect, when buyers may be waiting to see how the new vehicles perform relative to manufacturer claims.

If cost increases are small, either purchasers or sellers may absorb the cost increase without measurable changes in behavior. Significant cost increases passed through to buyers may lead potential buyers to purchase fewer vehicles than without the higher costs, or to buy vehicles sooner than they would have otherwise, in advance of the requirements. In a report on the HD Phase 2 rule, The National Academies of Sciences, Engineering, and Medicine stated that both pre-buy and low-buy are likely to be short-lived phenomena, and potentially unavoidable.[2] Allowing manufacturers to generate NO_X emissions credits before the final standards are required to be met may mitigate pre-buy; instead of early compliance imposing only net costs, early compliance now provides a benefit in terms of reduced compliance costs in the future, as well as early emissions reductions (see preamble Section III for details on the timing of the final standards).[3]

[C] We recognize that additional external factors, including the current global COVID-19 pandemic, might impact the heavy-duty vehicle market, however due to data limitations we are unable to include possible effects of such external factors in our analyses. However, though sales have not rebounded to levels seen directly before the COVID-19 pandemic, trends indicate they are heading that way. Seasonally adjusted data from The Bureau of Economic Analysis (published at https://fred.stlouisfed.org/series/HTRUCKSSA) show heavy weight truck retail sales (trucks weighing over 14,000 pounds) started increasing in mid-2009, through about May of 2019, and then fell dramatically until May of 2020, though sales never fell as low as they were in the mid-2009 time-frame. Sales increased again through March of 2021, before falling again through Sept 2021, and are currently increasing again. This indicates that possible shocks stemming from the global COVID-19 pandemic are likely short term, and the industry may return to historical levels by 2027.

[D] The reduced repair costs may counteract some of the sales effect of increased vehicle purchase cost. As a result, they may reduce incentives for pre- and low-buy, and mitigate adverse sales impacts.

Measuring the existence and magnitude of pre- and low-buy depends on separating those effects from other factors that affect HD vehicle sales. If, for example, the timing of the standards coincides with a decrease in HD vehicle sales due to an economic downturn, as likely happened with initial implementation of the 2007/2010 HD rule that went into effect during the eve of the Great Recession in 2007, then the estimated effects of the standard would somehow have to be disentangled from the effects of the economic slowdown. Researchers estimating pre- or low-buy may seek to control for underlying sales patterns like this by including other factors, such as diesel price and gross domestic product (GDP), that also influence new HD vehicle sales. They then look for deviations from these trends at the time that the standards go into effect.

Using this approach to control for other influential factors, Lam and Bausell found a pre-buy of around 18,000 to 21,000 HD trucks, totaling about 20 to 25 percent of total production, in the 6-month period before the October 2002 compliance deadline for HD engine manufacturers to reduce NOX emissions.[4] Similarly, Rittenhouse and Zaragoza-Watkins (RZW) looked for pre-buy in the seven months preceding compliance deadlines for EPA HD criteria pollutant standards in 1998, 2002, 2007 and 2010, as well as for low-buy in the seven months after those compliance deadlines.[5] For the 2007 standards, they found a sales increase of about 31,000 vehicles over the preceding seven months compared to the baseline, matched by an "approximately symmetric" drop in sales in the following months. For 2002, they found a similarly symmetric, though smaller, result of 14,000 – 18,000 Class 8 vehicles. These results suggest that the standards led to vehicles purchases being pulled forward that would have otherwise been purchased after the standards were promulgated in the absence of the standards. The resulting effect was slower adoption of lower-emissions vehicles, compared to the assumed rate of sales in the absence of new emissions standards, although there was essentially no net change in sales (the sum of pre-buy and low-buy was not statistically different from zero in 2007 or 2002). RZW did not find evidence of pre- or low-buy for the standards that went into effect in 1998 and 2010. They speculated that the standards in those years were less costly and involved use of less risky, already available technologies.

A limitation of the method used by the researchers discussed above is that they do not suggest a way to predict how future cost changes may influence sales. This is because they do not include price impacts in the approach to estimate pre- and low-buy impacts, yet vehicle price is expected to change when the standard goes into effect due to an increase in cost to the manufacturers. Both the change in price and the timing of the standard influence pre- and low-buy because they occur at the same time, and it is statistically difficult to separate the two effects. In addition, manufacturers of HD vehicles may affect either the magnitude or timing of price increases in response to cost increases, confusing the effect of price on sales. Thus, while these studies suggest that the current rule may lead to increases in sales through pre-buying behavior, and decreases in sales through low-buying, the estimation approaches used by the studies do not allow EPA to predict existence or magnitude of potential pre- and low-buy impacts from future standards. In an effort to improve our analyses, EPA has been working on a method to estimate these impacts. The approach and an example are explained in RIA Chapter 10.1.2, below.

An unpublished report attempted to develop a predictive model based on the impact of the 2007 standards. The authors assumed that a change in cost translates directly into a change in price, which was then converted into a change in sales (Harrison and LeBel).[6] The price change was based on asking manufacturers to estimate the costs of meeting the standards. The study then applied a price elasticity of demand of -1.9 (that is, a 1 percent increase in price will lead to a 1.9 percent decrease in sales) to estimate sales increases of 104,000 trucks during 2005-2006, and sales reductions of 149,000 trucks over 2007-2008. (The study did not provide details on the source of this elasticity estimate.) The study then reported "actual"[E] results of sales increases of 120,000 vehicles in 2005-6 and decreases of 183,000 vehicles in 2007-8, based on comparing estimated sales to an EPA estimate of baseline sales increased by a constant amount each year. Unlike the published studies reviewed above, Harrison and LeBel did not control for GDP, diesel prices, or other factors that might independently affect vehicle sales.[6] As a result, the EPA baseline used for the "actual" results is not likely to reflect actual sales in the absence of the standards, and the "actual" pre- and low-buy values likely do not reflect changes due only to the standards. For comparison, RZW's finding of pre-buy of about 31,000, based on controlling for other factors, is about one-third of Harrison and LeBel's prediction and one-fourth of their "actual" estimate.[5]

In sum, existing literature does not provide sufficient insight on the relationship between a change in vehicle cost due to a new standard and sales impacts. Neither Lam and Bausell nor RZW links a change in vehicle cost to sales impacts (a major interest of the EPA); instead, both papers focus on the magnitude of sales impacts in the periods surrounding compliance deadlines of HD emission regulations.[4,5] The method proposed by Harrison and LeBel links costs to sales via a demand elasticity, but omits controls for shifts in baseline conditions as well as omitting details on the source of the demand elasticity they used.[6]

For this rule, EPA acknowledges that these standards may lead to some pre-buy before the standards go into effect, and some low-buy after the standards are effective. The estimated increase in costs is not expected to have much effect on pre- or low-buy behavior because the increase is small relative to the cost of the vehicle. Based on the literature previously described, EPA is not able to quantify these effects for this rule. In the following subsection we outline an approach that could be used to quantify sales effects, and we illustrate how this method could be used to estimate pre- and low-buy as a function of the estimated costs outlined in Chapter 7.

10.1.2 **EPA's Research to Estimate Sales Effects**

In 2020 EPA contractors conducted a review of available peer reviewed literature on the effects of EPA's HD standards on HD sales (see RIA Chapter 10.1.1 for literature review results). The contractors then conducted an original analysis of the effects of previous EPA standards on pre- and low-buy for HD vehicles.[7]

The analysis uses monthly vehicle sales data from the twelve-month period before and after previous EPA HD standards went into effect (2002,[F] 2007, 2010, and 2014) to estimate pre- and

[E] Quotation marks around "actual" are included in Harrison and LeBel (2008).
[F] Due to a consent decree in 1998 requiring six major HD engine manufacturers in the U.S. to meet a 2.5 g/bhphr limit on NMHC+NOX by October 1, 2002, much of the regulatory implementation of the 2004 HD rule was pulled forward. Therefore, we will refer to the implementation of that regulation as the 2002 standards, instead of the 2004 standards, in order to keep the focus on compliance dates. More information on these consent decrees can be found on EPA's Civil Cases and Settlements by Statute webpage:
https://cfpub.epa.gov/compliance/cases/index.cfm?templatePage=12&ID=1&sortby=RELEASE_DATE,RELEASE_DATE&stat=Clean%20Air%20Act

low-buy due to each standard. The analysis examined controls for the effects of month of year, GDP, Brent Oil price, total imports and exports, and consumer sentiment, and then used binary indicator variables from 1 through 12 months pre- and post-regulation to identify deviations from trends in sales specifically around those regulations' implementation dates. All other variables (except for the binary variables of interest and the month of year) were transformed into log-differences to address statistical issues associated with time series data. Independent regressions were estimated for vehicle Classes 6 through 8, and for each of the four previous HD regulations' implementation dates. Additional details of this analysis are available in the contractors' report.

Results show no statistically significant sales effects for Class 6 vehicles. There were a few statistically significant results for Class 7, but the majority were of the opposite sign than expected (that is, reduced sales before the standards and increased sales afterwards). For Class 8 vehicles, there were statistically significant results in the expected directions, with evidence of short-lived pre-buy before the 2010 and 2014 standards, and evidence of short-lived low-buy after the 2002, 2007, and 2010 standards. The rest of this section focuses only on Class 8 vehicles. For more discussion on Classes 6 and 7, see Appendix 7.2 and Chapter 4.4.3 of the report.

The results provide estimates of the percent deviation in sales from trend for the combined months leading up to and following the start of new emissions standards. For pre-buy, statistically significant results range from no change persisting for the eleven months before the 2002 standards, to a 13.2 percent increase in the percent change in sales persisting for one month before the 2014 standards. Statistically significant effects persist for up to eleven months before the 2002 standards. For low-buy, statistically significant effects range from no change to a 14.9 percent decrease in the percent change in sales persisting for six months after the 2007 standards. Statistically significant effects persist for up to twelve months following the 2007 standards. Importantly, in addition to capturing the effects due to price changes associated with the regulations, the coefficients also capture unobserved factors, such as concerns over vehicle reliability and control technology uncertainty. Table 10-1 provides the results for the coefficients on the pre- and low-buy indicators, along with their length of persistence and the regulation to which the result is attributed. The significant coefficients are shown in bold face type.

Table 10-1: Pre and Low-Buy Sales Effects Coefficients

		2002	2007	2010	2014
Combined Months Pre-Regulation	12	0.024	0.004	0.009	0.000
	11	**-0.0****	-0.006	0.021	0.000
	10	**0.0****	-0.005	0.041	0.010
	9	0.032	-0.008	0.032	0.032
	8	0.041	-0.004	**0.057****	0.013
	7	0.044	-0.006	**0.059****	0.019
	6	0.037	-0.004	0.043	0.021
	5	0.029	0.003	**0.054***	0.019
	4	0.004	-0.011	**0.079*****	0.030
	3	0.047	-0.013	**0.071****	0.014
	2	-0.017	-0.012	**0.105*****	0.003
	1	-0.032	-0.01	**0.078*****	**0.132*****
Combined Months Post-Regulation	1	**0.065*****	**-0.07*****	**-0.144*****	-0.009
	2	-0.051	**-0.099*****	**-0.083***	-0.012
	3	-0.115	**-0.133*****	-0.051	-0.015
	4	-0.065	**-0.143*****	-0.052	0.003
	5	-0.066	**-0.144*****	**-0.075****	-0.009
	6	**-0.076***	**-0.149*****	-0.052	-0.006
	7	-0.017	**-0.121*****	-0.022	0.001
	8	-0.018	**-0.114*****	-0.034	0.000
	9	-0.018	**-0.099*****	-0.020	0.003
	10	-0.007	**-0.073****	-0.030	0.006
	11	-0.027	**-0.07****	-0.010	-0.013
	12	-0.014	**-0.065****	-0.005	0.000

*** $p < 0.01$; ** $p < 0.05$; * $p < 0.1$

As can be seen in Table 10-1, results vary by regulation. For the purposes of this discussion, we focus on results for the 2007 and 2010 standards.[G] For the 2007 standards, there is no statistically significant pre-buy. There is statistically significant low-buy for all the periods from the period of one month after the standard through the combined period of 12 months after the standard, with magnitude increasing up to, and falling after, the combined period of 6 months post-standard. For the 2010 standards, there is some evidence of both pre- and low-buy. Statistically significant pre-buy can be seen for the period of 1 month up to the combined period of 5 months, and again at the combined periods of 7 and 8 months pre-standard. There is significant low-buy for the periods of 1, 2 and 5 months post-regulation. Results indicate that the observed effects are short-lived, on the order of months rather than years.

[G] We do not consider the results of the 2002 compliance date to be generalizable for several reasons. Litigation may have affected purchase plans for many firms resulting from the pulling forward of compliance dates from 2004 to 2002. In addition, there may have been greater concerns over the reliability of new engines compared to other regulatory actions, which may have led to more low-buy. Also, the cost of compliance in 2002 was estimated to be lower than that of other regulations. We do not consider the 2014 standards to be generalizable either. This rule reduced greenhouse gas (GHG) emissions, which had lower technology costs and fuel savings relative to other rules. In addition, numerous pathways for compliance leads to difficulty estimating the price change in HD vehicles due to the regulation. More details and discussion on the 2002 and 2004 standards are available in the contractors' report.

10.1.2.1 Estimating Elasticities

To estimate a change in Class 8 vehicle sales due to future EPA emission standards, we transform the coefficients on the indicator variables, explained above, into demand elasticities. These elasticities (ε) measure the percent change in vehicle sales due to a percent change in vehicle prices:

Equation 10-1

$$\boldsymbol{\varepsilon} = \frac{\%\boldsymbol{\Delta Sales}}{\%\boldsymbol{\Delta Price}}$$

The percent change in sales for Class 8 vehicles comes from the coefficients on the indicator variables from Table 10-1. In estimating elasticities, we only use the significant coefficients, while noting that no response (an elasticity of 0) is also represented in the results. The percent change in price is estimated by dividing the estimated cost of compliance published in the EPA RIAs associated with the relevant standard (2007/2010 HD rule) by the estimated purchase price of a Class 8 vehicle in that year (adjusted to 2010 dollars).[H] Table 10-2 shows the regulatory cost, the HD vehicle prices and the resulting percent change in price we are using to estimate the elasticities from the 2007 and 2010 standards.

Table 10-2: Regulatory Costs and HD Vehicle Prices Used to Estimate Elasticities

Statutory Deadline	Regulatory Cost (2010$)	HD Vehicle Price	% Change in Price
2007	$9,741	$98,900	9.8%
2010	$7,662	$108,250	7.1%

From the statistically significant pre- and low-buy sales effects for Class 8 vehicles, we estimate a set of pre- and low-buy elasticities by dividing the percent change in sales, from Table 10-1, by the percent change in price, from Table 10-2.

Table 10-3 shows the estimated statistically significant coefficients (percent change in sales) from Table 10-1, their period of effect, and the associated estimated elasticity. We expect pre-buy elasticities to be positive (more sales before new emission standards) and low-buy elasticities to be negative (fewer sales after new emission standards). Because the smallest statistically significant sales effect is zero, and a number of other effects are not statistically different from zero, the smallest pre- and low-buy elasticities are zero – no effect due to the standards. It should also be noted that not only the magnitude of the elasticity matters, but also the time period over which the elasticity applies. A large elasticity for a short period of months may measure less effect than a small elasticity over a longer period.

[H] The estimated cost of compliance was based on EPA's cost of compliance in the RIA for the relevant standard. The price of a Class 8 HD vehicle for each year was calculated as an average of a high and low list price from an online source for HD vehicle sales (Commercial Truck Trader, a site that advertises new and used trucks for sale).

Table 10-3: Elasticity Estimates

	Statutory Deadline	Period of Effect (Months)	% Change in Sales (β)	Estimated Elasticity
Pre-Buy	All	Any	0	0
	2010	8	0.057	0.805
		7	0.059	0.834
		5	0.054	0.763
		4	0.079	1.116
		3	0.071	1.003
		2	0.105	1.483
		1	0.078	1.102
Low-Buy	All	Any	0	0
	2007	12	-0.065	-0.660
		11	-0.070	-0.711
		10	-0.073	-0.741
		9	-0.099	-1.005
		8	-0.114	-1.157
		7	-0.121	-1.229
		6	-0.149	-1.513
		5	-0.144	-1.462
		4	-0.143	-1.452
		3	-0.133	-1.350
		2	-0.099	-1.005
		1	-0.070	-0.711
	2010	5	-0.075	-1.060
		2	-0.083	-1.173
		1	-0.144	-2.034

There are several limitations to the results presented in Table 10-3. As noted in Chapter 10.1.2, the sales coefficients used to estimate the elasticities likely capture aspects of the final regulation not solely limited to changes in price (e.g., adverse fuel consumption effects, or concerns about the reliability of untested control technology). Similarly, the base vehicle prices and estimated regulatory costs discussed above are estimates and may not correspond with observed base prices or increased regulatory costs. A commenter on the proposed rule also noted that a limitation of this method is that it assumes 100% cost pass-through to consumers.[I] In addition, though we estimate a range of possible effects, including zero, this method assumes buyers will continue to respond to regulation similarly as they have in the past. This may change over time as market offerings change, for example if vehicles become more durable, or as the HD vehicle market includes more electrified or fuel cell HD vehicles.

[I] See Section 25 of the Response to Comments for our response to this comment.

Since these elasticities are based on monthly data, it is appropriate to apply the estimated elasticities to monthly series. Analysis of the coefficients over time indicates that the observed effects are short-lived, on the order of months rather than years. As noted above and described below, the time period is a critical factor for estimating the impacts.

10.1.2.2 Illustrative Example

This subsection outlines how we could apply the pre- and low-buy elasticities presented in Table 10-3 to this rulemaking. Though the methodology to develop the elasticities has been peer reviewed,[8] the application in a rulemaking is new, and thus, in this subsection, we are illustrating how we could use this method to estimate pre- and low-buy as a function of the estimated costs outlined in Chapter 7.

Expanding Equation 10-1, elasticity measures can be approximated as

Equation 10-2

$$\varepsilon = \frac{\%\Delta Sales}{\%\Delta Price} = \frac{\Delta Sales}{Sales} * \frac{Price}{\Delta Price}$$

In this application, we want to estimate how a change in price leads to a change in sales. Therefore, rearranging Equation 10-2, we get

Equation 10-3

$$\Delta Sales = \varepsilon * \frac{\Delta Price}{Price} * Sales$$

The elasticity measures come from the estimates explained above.

For this example, $\Delta Price$ is the estimated cost of compliance for a Class 8 HD vehicle to meet the final MY 2027 standards, $4,827 (see RIA, Chapter 7, Table 7-24). We assume implementation starts January 1 for the vast majority of the heavy-duty engine industry.[J]

$Price$ is the estimated price of a Class 8 truck, which we set to $130,000.[K]

$Sales$ are the estimated monthly Class 8 vehicle sales in 2026 and 2027.[L] Monthly sales are derived from Class 8 vehicle population data from projected sales volumes using EPA's MOVES

[J] This is an illustrative example, and thus may not fully represent the final program; see preamble Section III.A for additional discussion on implementation dates in the final rule.

[K] The price of HD vehicles varies greatly, and in part due to the features or options of the vehicle. The price we use here comes from the estimated price of a low-end, new, semi-truck from Truckers Bookkeeping Service from June, 2021; a pdf of the page "How much does a semi truck cost?" can be found in the docket for this rule, Docket ID EPA-HQ-OAR-2019-0055.

[L] We do not have the sales data for 2026, therefore we approximate 2026 vehicles sales in this illustrative example with 2027 vehicle sales.

model[M] and month-specific effects from the contractors' report.[N] To estimate pre-buy, we use the estimated monthly HD vehicle sales in the months before January 1, 2027. That is, pre-buy for the 2027 compliance date is estimated with the calculated monthly sales in 2026. To estimate low-buy, we use the estimated monthly HD vehicle sales in 2027.

To get the sales effects, the elasticity estimates from Table 10-3 are multiplied by the change in price divided by the base price. This value is then multiplied by the estimated Class 8 HD vehicle sales for each month over the period of effect for that elasticity measure. This results in a change in sales for each month over the period of effect. The monthly results are then summed to get a total affect for each elasticity estimate.

For pre-buy, for example, the elasticity measurement of 1.10 has a period of effect of one month, so we use the Class 8 sales from December, 2026 and make the pertinent multiplication for just December, 2026. For the elasticity measurement of 0.81, the period of effect is 8 months, so we use the Class 8 sales estimates from May, 2026 (8 months before January, 2027) through December, 2026 and the pertinent multiplication estimation is made for each affected month. Then, the changes in sales for each affected month are added together to get the total effect. The results for pre-buy are in Table 10-4 below. As discussed in Chapter 10.1.2.1, there are sales effects results that are statistically indistinguishable from zero, which means that zero impact on sales is the lower bound on effects. Total sales of Class 8 vehicles under this final rule are estimated to increase by between 0 and approximately 2 percent on an annual basis before the 2027 compliance deadline. In addition, the duration of the effects is a critical component in the calculation of sales impacts. For example, the elasticity of 0.83 for a duration of 7 months has a larger aggregate impact than the larger elasticity of 1.12 for a duration of 4 months. The result that produces the largest estimate for an aggregate increase in sales is the elasticity of 0.81 for 8 months.

Table 10-4: Illustrative Pre-Buy Results

Period of Effect (Months)	Elasticity	Aggregate Sales Change	Cumulative % Change in Sales
Any	0	0	0
8	0.81	4,701	2.01%
7	0.83	4,264	1.83%
5	0.76	2,815	1.21%
4	1.12	3,317	1.42%
3	1.00	2,255	0.97%
2	1.48	2,196	0.94%
1	1.10	936	0.40%

[M] See Population and Activity of Onroad Vehicles in MOVES3 (EPA-420-R-21-012) available online at https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P1011TF8.pdf for more information on how vehicle population data is estimated in MOVES.

[N] Because these populations are annual, and the elasticities are monthly, we have to distribute the annual sales throughout the year. To do so, we estimate the average monthly sales and then use the monthly sales effect (the percent change in sales by month) estimated in the contractors' report to better approximate the sales by month.

Low-buy is estimated the same way, though we use monthly sales estimates for the requisite number of months following the January 1, 2027 compliance date. Low-buy results for the final rule are show in Table 10-5. For an elasticity of -1.51 over the course of 6 months (an estimate from the 2007 standards), we use the monthly sales estimates for January, 2027 through June, 2027, make the pertinent multiplication estimations for each affected month, and add the monthly results to get the aggregate sales change. As discussed in Chapter 10.1.2.1, for all rules there are sales effects results that are statistically indistinguishable from zero, which means that zero impact on sales is the lower bound on effects. This example estimates sales of Class 8 vehicles in the months following the 2027 compliance date to fall by between 0 and just under 3 percent on an annual basis. As with pre-buy, both the magnitude of the elasticity and the duration of effect are important in estimating the total effect. For example, the elasticity of -1.01 for the duration of 9 months (from the 2007 standards) results in a larger aggregate effect than the larger elasticity of -1.23 for the duration of 7 months. The result that produces the largest estimate for an aggregate decrease in sales is the elasticity of -1.16 for a duration of 8 months.

Table 10-5: Illustrative Low-Buy Results[O]

Statutory Deadline	Period of Effect (Months)	Elasticity	Aggregate Sales Change	Cumulative % Change in Sales
All	Any	0	0	0
2007	12	-0.66	(5,717)	-2.45%
	11	-0.71	(5,552)	-2.38%
	10	-0.74	(5,323)	-2.28%
	9	-1.01	(6,447)	-2.76%
	8	-1.16	(6,586)	-2.82%
	7	-1.23	(6,108)	-2.62%
	6	-1.51	(6,507)	-2.79%
	5	-1.46	(5,186)	-2.22%
	4	-1.45	(4,102)	-1.76%
	3	-1.35	(2,832)	-1.21%
	2	-1.01	(1,252)	-0.54%
	1	-0.71	(364)	-0.16%
2010	5	-1.06	(3,759)	-1.61%
	2	-1.17	(1,461)	-0.63%
	1	-2.03	(1,041)	-0.45%

10.1.3 **Fleet Turnover and Emissions Impacts**

At the level of an individual HD vehicle, the emissions standards in this rule will result in a new vehicle emitting less than a legacy vehicle; these lower emissions impacts will occur immediately upon the new vehicle entering into service (e.g., 2027). In contrast, the total emissions impact of the final standards across the fleet will occur more gradually, because,

[O] The 2007 and 2010 Statutory Deadline column indicates the implementation date of the standards the low-buy sales effects estimates come from as seen in Table 10-1.

initially, vehicles meeting the standards will only be a small portion of the total fleet. For instance, in 2019 about 369,000 new medium-/heavy-duty trucks and buses were sold, compared to over 14 million total medium-/heavy-duty vehicle registrations.[9] Over time, as more vehicles subject to the standards enter the market and older vehicles leave the market, the emissions reductions due to the standards will increase. This relationship holds true even if new vehicle sales are unaffected by the standards.

If pre-buy and low-buy behaviors occur, they can shift emissions impacts in several ways. First, under low-buy, there is slower adoption of new vehicles, which implies that emissions reductions will be slower than under the assumption of no change in vehicle sales (RZW).[5] On the other hand, the pre-bought HD vehicles are likely to displace older, more polluting vehicles, which may provide an earlier reduction in emissions than would have occurred without the standards. However, although the pre-bought new HD vehicles are likely to have lower emissions than the older, displaced vehicles, the emissions reductions are likely to be smaller than the reductions that will be realized from the purchase and use of new vehicles subject to these standards, so that the net effect of pre-buy is to slow reductions in emissions.[5]

Another potential effect of the standards is a net reduction in new vehicle sales. This could result from either a smaller pre-buy than the post-standards low-buy,[P] or some potential buyers deciding not to purchase at all. In this case, the vehicle miles traveled (VMT) of older vehicles may increase to make up for the VMT that would otherwise have been expected of the newer ("missing") vehicles. To the extent that the older vehicles emit more than the missing vehicles, emissions may increase.[Q] However, because the VMT is likely to be shifted to the newer HD vehicles among the existing fleet, and most of those vehicles are expected to be in compliance with the existing HD vehicle standards, this effect is expected to be small.

Quantifying these effects requires a robust method to estimate the effects of the standards on pre-buy and low-buy, as well as a method to estimate shifts in VMT among vehicle vintages in the case of an expected change in the net sales of newer vehicles. In the absence of robust methods to estimate these effects, EPA is not quantifying the fleet turnover or emissions impacts of sales effects in this rule, though, as with pre-buy and low-buy, we acknowledge these potential impacts.

The estimated increase in operating costs due to an increase in the use of DEF may lead to a slower fleet turnover if VMT does not shift from older (lower DEF requirements) vehicles to the newer (higher DEF requirements) vehicles. This leads to slower emission reductions than if those newer vehicles were used. However, as this increase in operating costs is small, and may be offset in part by reduced repair costs, we expect minimal effects on fleet turnover due to the change in operating costs.

Another factor that may impact fleet turnover is the increase in HD vehicles' operational life. With an increase in operational life, vehicles compliant with this regulation may stay on the road

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[P] Though this is a possibility, it should be noted that the RZW (2018) study found that pre-buy approximately equaled low-buy. Harrison and LeBel (2008) found that low-buy exceeded pre-buy.

[Q] This effect is sometimes called the "Gruenspecht effect," based on the theory presented in Gruenspecht, Howard (1982), "Differentiated Regulation: The Case of Auto Emissions Standards," *American Economic Review* 72: 328-331.

longer, leading to reduced fleet turnover. As the vehicles that might be in operation longer due to this regulation are in compliance with this regulation, we do not expect emission impacts due to reduced fleet turnover as a function of increased operational life.

We do not have data to estimate the effect the final regulation might have on HD scrappage rates. If the regulation leads to HD vehicle owners holding onto used vehicles longer (a reduction in scrappage), this could result in slower fleet turnover, even if new vehicle purchases are unaffected by the regulation. If the regulation leads to vehicle owners preferring to shift ownership to newer vehicles, this could result in increased fleet ownership and associated increased scrappage. Modeling dynamic scrappage (scrappage that changes due to the regulation) relies on consumer choice modeling, which is especially difficult to estimate in the HD context, as there are many different kinds of consumers and products spanning Classes 4 through 8. Though EPA has included estimates of dynamic scrappage in previous light-duty rules, analyses based on dynamic scrappage were only included in the recent 2023-2026 model year light-duty rule, where we relied on the CCEMS model.[R] Though we do not have data to estimate dynamic scrappage, HD vehicle scrappage is accounted for in the MOVES model in the same way light-duty vehicle scrappage is estimated.[S] The MOVES scrappage algorithm uses historical vehicle survival rates to predict future year scrappage. Scrappage in the MOVES model is static and does not have a consumer choice component.

10.1.4 **Potential for Mode Shift**

Another possible response to the new emissions standards is shifting freight shipments to other transportation modes, such as rail or barge. This may happen, for example, if the new standards were to raise operating costs enough to make truck transportation more expensive than rail or marine alternatives.

EPA does not expect this rule to result in a transportation mode shift. Generally, shipping cargo via truck is more expensive per ton-mile than barge or rail, and less expensive than air.[10,11] This is due to many factors, not the least of which is labor costs (each truck has at least one driver). Even though trucking is more expensive than rail or marine on a ton-mile basis, it is a very attractive transportation alternative for several reasons: shipping via truck is generally faster and more convenient than rail or marine, trucks can reach more places, and trucks may be less constrained by available infrastructure than barge or rail. In addition, shipping via truck does not require trans-shipments (transferring from one mode to another, for example to deliver cargo to or from the port or rail yard), and it allows partial deliveries at many locations. This speed, infrastructure availability, and delivery flexibility make trucking the transportation solution of choice for many kinds of cargo across most distances. As a result, smaller shipments of higher-valued goods (e.g., consumer goods) tend to be transported by air or truck, while larger shipments of lower-valued goods (e.g., raw materials) tend to go via rail or barge.[10,12]

[R] Final Rule to Revise Existing National GHG Emissions Standards for Passenger Cars and Light Trucks Through Model Year 2026. Found online at https://www.epa.gov/regulations-emissions-vehicles-and-engines/final-rule-revise-existing-national-ghg-emissions. See Chapter 4 of the final Regulatory Impact Analysis for details regarding EPA's use of CCEMS for that rule.

[S] This is documented in Section 6.1 and Appendix C of <u>Population and Activity of Onroad Vehicles in MOVES3 (PDF)</u> (232 pp, 4.3 MB, April 2021, EPA-420-R-21-012).

Studies of intermodal freight shifts, such as Comer et al. or Bushnell and Hughes, focus on changes in cost per ton-mile as a potential source of transportation mode shift.[10,12] Comer et al. note, for instance, that fuel consumption "depend[s] on the type of freight being moved, route characteristics, transport speed, and locomotive/truck characteristics."[10] Bushnell and Hughes estimate that increased fuel prices for truck transportation lead to small substitutions between truck and rail for small or large shipments, and higher shifts for intermediate-sized shipments.[12] The findings from this study suggest that the variation (in kinds and values) of goods shipped by different means likely results in only a small amount of mode shift in response to a change in operating cost (e.g., fuel prices). However, due to data availability, this study approximates freight rates with fuel costs, assumes shipping distances using different modes are the same, and mostly does not consider transportation availability constraints affecting some modes in some regions. These limitations may distort the effects they estimate.

A mode shift study EPA carried out in 2012 in the context of new sulfur limits for fuel used in large ships operating on the Great Lakes may help address some of these limitations.[11] The methodology used a combination of geospatial modeling and freight rate analysis to examine the impact of an increase in ship operating costs. While the focus of the study was transportation mode shift away from marine and toward land, it noted that truck transportation is far more expensive than both rail and marine on a ton-mile basis.[T] It also shows that even a large percentage increase in marine operating costs did not raise freight rates by a similar percentage, because fuel costs are only part of total operating costs. In the case of truck transportation, operating costs are a much smaller portion of total costs. The results of this study combined with the others cited in this section indicate that changing the cost of truck transportation is unlikely to create mode shift.

The primary effect of the standards on operating costs is an increase in the use of DEF, which is expected to be partially offset by reductions in emission repair costs. The increase in total operating cost is a very small part of the total increase in cost impacts estimated in this final rule (see Chapter 7.3). Because the cost effect is expected to be small relative to the price of a HD vehicle, and substitution between trucks and other modes is limited by the nature of the goods and their routes, we do not expect significant effects on mode shift. Finally, given the higher costs of truck transportation, a relatively small increase in truck freight rates due to the small increase in operating costs are unlikely to affect the competitive dynamics of the transportation sector.

10.1.5 **Effects on Domestic and International Shares of Production**

The final standards are not expected to provide incentives for manufacturers to shift between domestic and foreign production. This is because the standards apply to any vehicles sold in the U.S. regardless of where they are produced. If foreign manufacturers already have increased expertise in satisfying the requirements of the standards, there may be some initial incentive for foreign production, but the opportunity for domestic manufacturers to sell in other markets might increase. To the extent that the requirements of these final rules might lead to application and use

[T] Figure 1-5 in U.S. EPA Office of Transportation and Air Quality. "Economic Impacts of the Category 3 Marine Rule on Great Lakes Shipping." EPA-420-R-12-005. 2012.

of technologies that other countries may seek now or in the future, developing this capacity for domestic producers now may provide some additional ability to serve those markets.

10.1.6 Summary of Sales, Turnover, and Mode Shift Impacts

As discussed in Chapter 7, EPA expects the cost of new HD vehicles to increase (see Chapter 7 for details). This increase may have some impact on new vehicle sales; in particular, it is possible that truck buyers increase purchases before the standards become effective (pre-buy), and reduce purchases afterwards (low-buy). Studies of pre-buy and low-buy suggest that these phenomena have occurred in the past, but those studies do not provide methodologies for estimating the impact of new rules on future vehicle sales. For that reason, EPA conducted an analysis to develop a relationship between estimated changes in vehicle price due to a new regulation and corresponding changes in vehicle sales (i.e., pre- and low-buy elasticities). We present the details of this new analysis and provide an illustrative example of applying pre- and low-buy elasticities to estimate potential sales impacts on Class 8 vehicles. For pre-buy and low-buy, the illustrative example shows that sales impacts on Class 8 vehicles are of limited duration and range from zero impact to approximately 2 percent for pre-buy and from zero to just under three percent for low-buy.

Whether shippers might choose a different mode for freight depends not only on the cost per ton-mile of the shipment, but also the value of the shipment, the time needed for shipment, and the availability of infrastructure. This rule is expected to affect the cost per ton-mile by only a small amount. For that reason, EPA expects little transportation mode shift to occur due to the final standards. EPA also does not expect changes in where production happens in response to these standards.

10.2 Employment Impacts

This section explains the methods and estimates of employment impacts due to this regulation. Though the rule primarily affects HD vehicles, the employment effects may be felt more broadly in the motor vehicle and parts sectors due to the effects of the standards on sales. Thus, we focus our assessment on the motor vehicle manufacturing and the motor vehicle parts manufacturing sectors, with some assessment of impacts on additional sectors likely to be most affected by the standards. Chapter 10.2.1 offers a brief, high-level explanation of employment impacts due to environmental regulation and discusses a selection of the peer-reviewed literature on this topic. Chapter 10.2.2 discusses EPA's qualitative and quantitative estimates of the partial employment impacts of this rule on regulated industries. Chapter 10.2.3 examines employment impacts in some closely related sectors, and Chapter 10.2.4 summarizes expected employment impacts.

10.2.1 Economic Framework for Employment Impact Assessment

Economic theory of labor demand indicates that employers affected by environmental regulation may increase their demand for some types of labor, decrease demand for other types of labor, or for still other types, not change it at all. A variety of conditions can affect employment impacts of environmental regulation, including baseline labor market conditions and employer and worker characteristics such as industry, and region. Isolating employment impacts of regulation is difficult as they are a challenge to disentangle from employment impacts caused by a wide variety of ongoing concurrent economic changes.

A growing literature has investigated employment effects of environmental regulation. Morgenstern et al. decompose the labor consequences in a regulated industry facing increased abatement costs. They identify three separate components.[13] First, there is a demand effect caused by higher production costs raising market prices. Higher prices reduce consumption (and production), reducing demand for labor within the regulated industry. Second, there is a cost effect where, as production costs increase (for example, due to pollution control activities that require additional labor to produce the same output quantity), plants use more of all inputs including labor to produce the same level of output. Third, there is a factor-shift effect where post-regulation production technologies may have different labor intensities. These three effects outlined by Morgenstern et al. provide the foundation for EPA's analysis of the impacts of the current regulation on labor.[13]

Additional papers approach employment effects through similar frameworks. Berman and Bui model two components that drive changes in firm-level labor demand: output effects and substitution effects.[14,U] The output effect happens when prices increase, leading to a decrease in quantity demanded, and results in a decrease in production. The substitution effect happens when regulation affects labor intensity of production (holding output constant). Deschênes describes environmental regulations as requiring additional capital equipment for pollution abatement that does not increase labor productivity.[15] These higher production costs induce regulated firms to reduce output and decrease labor demand (an output effect) while simultaneously shifting away from the use of more expensive capital towards increased labor demand (a substitution effect).[V] Ehrenberg and Smith describe how at the industry level, labor demand is more likely to be responsive to regulatory costs if: (1) the elasticity of labor demand is high relative to the elasticity of labor supply, and (2) labor costs are a large share of total production costs.[16] Labor demand might also respond to regulation if compliance activities change labor intensity in production.

Arrow, Cropper, at al. state that, in the long run, environmental regulation is expected to cause a shift of employment among employers rather than affect the general employment level.[17] Even if they are mitigated by long-run market adjustments to full employment, many regulatory actions have transitional effects in the short run.[18,19] These movements of workers in and out of jobs in response to environmental regulation are potentially important distributional impacts of interest to policy makers. Of particular concern are transitional job losses experienced by workers operating in declining industries, exhibiting low migration rates, or living in communities or regions where unemployment rates are high.

Workers affected by changes in labor demand due to regulation may experience a variety of impacts including job gains or involuntary job loss and unemployment. Compliance with environmental regulation can result in increased demand for the inputs or factors (including labor) used in the production of environmental protection. However, the regulated sector generally relies on revenues generated by their other market outputs to cover the costs of supplying increased environmental quality, which can lead to reduced demand for labor and

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[U] Berman and Bui also discuss a third component, the impact of regulation on factor prices, but conclude that this effect is unlikely to be important for large competitive factor markets, such as labor and capital.
[V] For an overview of the neoclassical theory of production and factor demand, see Chapter 9 of Layard and Walters (1978).

other factors of production used to produce the market output. Workforce adjustments in response to decreases in labor demand can be costly to firms as well as workers, so employers may choose to adjust their workforce over time through natural attrition or reduced hiring, rather than incur costs associated with job separations (see, for instance, Curtis and Hafstead and Williams).**Error! Bookmark not defined.**,[20]

Employment impacts, both positive and negative, in sectors upstream and downstream from the regulated sector, or in sectors producing substitute or complimentary products, may also occur.

10.2.2 Employment Impacts in the Motor Vehicle and Parts Manufacturing Sectors

In this section, EPA presents partial estimates of industry-level employment effects of the final rule. We use the labor intensity of production for motor vehicle manufacturing and motor vehicle parts manufacturing to provide a range of potential employment impacts.[W]

Our analysis follows the structure of Morgenstern et al., as described above, to estimate the impacts of this rule on the regulated sector.[13] We qualitatively describe the employment impacts due to the factor-shift and demand effects, provide an illustrative example of demand effects, and quantitatively estimate the employment impacts due to the cost effect. Due to a variety of reasons, including that our quantitative estimates of the demand effect are merely illustrative, and we do not estimate factor-shift effects, our estimates do not reflect the total effects on employment in the regulated industries.

10.2.2.1 The Factor-Shift Effect

The factor-shift effect reflects employment changes due to changes in labor intensity of production resulting from compliance activities. The labor intensity of manufacturing HD vehicle engines or HD vehicles might increase or decrease because of the rule. Due to a lack of information on expected changes in labor intensity, the estimated employment impacts in this chapter do not include the factor-shift effect. In addition, EPA is aware of HD market shifts toward battery electrification, and that there may be employment effects due to that shift at least in part due to different labor intensity needs. Our results do not reflect market transitions toward battery electrification, in large part because we do not have data on employment differences in traditional manufacturing sectors and battery electric manufacturing sectors, especially for future expected effects. In addition, as discussed in preamble Section III, battery-electric and fuel cell electric vehicles are subject to the final standards, but we did not evaluate these technologies in setting the level of the final standards. Further, battery-electric and fuel cell electric vehicles cannot participate in the final ABT program (see Section IV.G). The combination of not including battery-electric technology when setting the final standards, and not allowing manufacturers to generate NO_X emissions credits from these technologies, leads us to not expect this regulation to induce a significant shift toward battery-electric heavy-duty vehicle production.

[W] We do not identify impacts separately for these sectors because we do not have information on the division of costs between them.

10.2.2.2 The Cost Effect

The cost effect reflects the impact on employment due to increased costs from adopting technologies needed for vehicles to meet the standards. The analysis holds output constant, meaning that it does not include sales impacts. We estimate the cost effect using the historic share of labor in the cost of production to extrapolate future estimates of impacts on labor due to new compliance activities in response to this regulation. Specifically, we multiply the share of labor in production costs by the production cost increase estimated as an impact of this rule. This provides a sense of the magnitude of potential impacts on employment.

The use of the ratio of the share of labor in production costs to estimate "cost effect" employment has both advantages and limitations. It is often possible to estimate these ratios for specific sectors, for example, the average number of workers in the HD vehicle manufacturing sector per $1 million spent in that sector, rather than using ratios from more aggregated sectors, such as the motor vehicle manufacturing sector. This means that it is not necessary to extrapolate employment ratios from possibly unrelated sectors. On the other hand, these estimates are averages, covering all the activities in these sectors and may not be representative of the labor effects when expenditures are required for specific activities, or when manufacturing processes change due to compliance activities in such a way that labor intensity changes. For instance, the ratio of workers to production cost for the HD motor vehicles manufacturing sector represents this ratio for all HD vehicle manufacturing and not just for production processes related to emission reductions compliance activities. In addition, these estimates do not include changes in sectors that supply these sectors, such as steel or electronics producers. The effects estimated here can be viewed as effects on employment in the HD motor vehicle sector due to the changes in expenditure in that sector, rather than as an assessment of all employment changes due to the final standards. In addition, labor intensity is held constant in the face of increased expenditures; this approach does not take in account changes in labor intensity due to changes in the nature of production (the factor-shift effect), which could either increase or decrease the employment impacts estimated here.

Some vehicle parts are made in-house by HD vehicle manufacturers. Other parts are made by independent suppliers who are not directly regulated but will be affected by this rulemaking as well. Because EPA does not know whether abatement equipment to comply with the final standards will be produced by the original equipment manufacturers (OEMs) or by suppliers, we use labor ratios for both sectors (and their subsectors) to provide a range of estimates for the cost effect impacts.

We include estimates from two sectors that are broadly defined and two that are more narrowly defined. Specifically, we estimate labor impacts for the aggregated sectors 'motor vehicle manufacturing' and 'motor vehicle parts manufacturing', and for the more specific sectors 'light truck and utility vehicle manufacturing' and 'heavy-duty truck manufacturing.'[X]

We rely on three different public sources to get a range of estimates of employment per $1 million expenditures: the Annual Survey of Manufactures (ASM) and the Economic Census

[X] The 'light truck and utility vehicle manufacturing' sector is included because these estimates include results for trucks from class 2b/3 through 6. See Preamble Section I of this rule for more discussion on the HD engine classes included in this rulemaking.

(EC), both provided by the U.S. Census Bureau, and the Employment Requirements Matrix (ERM) provided by the U.S. Bureau of Labor Statistics (BLS). The EC is conducted every 5 years, most recently in 2017.[Y] The ASM is an annual subset of the EC and is based on a sample of establishments. The latest set of data from the ASM is from 2020. The EC and ASM have more sectoral detail than the ERM, providing estimates out to the 6-digit North American Industry Classification System (NAICS) code level. They provide separate estimates of the number of employees and the value of shipments, which we convert to a ratio in this employment analysis.[Z] The ERM provides direct estimates of employees per $1 million in expenditures for a total of 202 aggregated sectors that roughly correspond to the 4-digit NAICS code level, and provides data through 2020. Table 10-6 below shows the sector definition, the NAICS code, and the ERM sector number where appropriate that EPA uses to estimate employment effects in this analysis.

Table 10-6: Sectors Used in this Analysis

Sector Definition	NAICS	ERM Sector Number
Motor vehicle manufacturing	3361	80
Light truck and utility vehicle manufacturing	336112	
Heavy-duty truck manufacturing	33612	
Motor vehicle parts manufacturing	3363	82

Table 10-7 provides the estimates of employment per $1 million of expenditure for each sector for each data source, adjusted to 2017 dollars using the U.S. Bureau of Economic Analysis Gross Domestic Product Implicit Price Deflator retrieved from the Federal Reserve Bank of St. Louis. The values are adjusted to remove effects of imports through the use of a ratio of domestic production to domestic sales of 0.81.[AA] While the estimated labor ratios differ across data sources, they each exhibit a similar pattern across sectors. Within the 4-digit NAICS code level, motor vehicle parts manufacturing seems to be the most labor-intensive sector, followed by the motor vehicle manufacturing sector. Within motor vehicle manufacturing, heavy-duty truck manufacturing appears to be more labor-intensive than light truck and utility vehicle manufacturing.

[Y] Though the Economic Census was conducted in 2022, data from 2022 will not begin to be released until March 2024.

[Z] The total employment across the two 4-digit NAICS code sectors used in this analysis (see Table 10-6) as reported in the ASM and the EC ranges from 775,016 to 787,640 depending on which data source is used; as noted above the most recent ASM and EC were conducted in 2020 and 2017, respectively.

[AA] To estimate the proportion of domestic production affected by the change in sales, we use data from WardsAuto for total car and truck production in the U.S. compared to total car and truck sales in the U.S. Over the period 2009-2021, the proportion averages 83 percent. From 2016-2021, the proportion average is slightly lower, at 81 percent.

Table 10-7: Employment per $1 Million Expenditures (2017$) in the Motor Vehicle Manufacturing Sector [a]

Source	Sector (NAICS)	Ratio of Workers per $1 Million Expenditures	Ratio of Workers per $1 Million Expenditures, Adjusted for Domestic vs. Foreign Production
BLS ERM 2017	Motor vehicle manufacturing (3361)	0.504	0.408
ASM 2016	Motor vehicle manufacturing (3361)	0.754	0.611
EC 2017	Motor vehicle manufacturing (3361)	0.615	0.498
BLS ERM 2017	Motor vehicle parts manufacturing (3363)	1.846	1.496
ASM 2016	Motor vehicle parts manufacturing (3363)	2.642	2.141
EC 2017	Motor vehicle parts manufacturing (3363)	2.231	1.808
ASM 2016	Heavy-duty truck manufacturing (33612)	1.451	1.175
EC 2017	Heavy-duty truck manufacturing (33612)	0.988	0.800
ASM 2016	Light truck and utility vehicle manufacturing (336112)	0.640	0.519
EC 2017	Light truck and utility vehicle manufacturing (336112)	0.478	0.388

[a] BLS ERM refers to the U.S. Bureau of Labor Statistics' Employment Requirement Matrix, 2020 values. ASM refers to the U.S. Census Bureau's Annual Survey of Manufactures, 2020 values. EC refers to the U.S. Census Bureau's Economic Census, 2017 values. These are the most recent data available.

Over time, the amount of labor needed in the motor vehicle industry has changed: automation and improved methods have led to significant productivity increases. The BLS ERM, for instance, provides estimates that, in 1997, about 1.2 workers in the Motor Vehicle Manufacturing sector were needed per $1 million, but only 0.5 workers by 2020 (in 2017$).[BB] Because the ERM is available annually for 1997-2020, we use these data to estimate productivity improvements over time. We regress logged ERM values on a year trend for the Motor Vehicle Manufacturing and Motor Vehicle Parts Manufacturing sectors. We use this approach because the coefficient describing the relationship between time and productivity is a direct measure of the average percent change in productivity per year. The results of the regressions suggest a 4.4 percent per year productivity improvement in the Motor Vehicle Manufacturing Sector and a 4.0 percent per year improvement in the Motor Vehicle Parts Manufacturing Sector.

We then use those estimated percent improvements in productivity to project the number of workers per $1 million of production expenditures through 2031. Although the costs and benefits analyses in the preceding chapters go out to 2045, we chose to model the employment effect due to cost increases through 2031, for a total of five years. This is because our method is an approximate, partial employment analysis, as well as being dependent on future, uncertain, macro-economic conditions. The results provided below represent an order of magnitude effect, rather than definitive impacts. We calculate separate sets of projections (adjusted to 2017$) for each set of data, ERM, EC, and ASM, for all four sectors described above. The ERM projections

[BB] http://www.bls.gov/emp/ep_data_emp_requirements.htm; this analysis used data for sectors 80 (Motor Vehicle Manufacturing) and 82 (Motor Vehicle Parts Manufacturing) from "Chain-weighted (2009 dollars) real domestic employment requirements tables;" see "Cost Effect Employment Impacts calculation" in the docket.

are calculated directly from the fitted regression equations used to estimate the projected productivity growth, since the regressions themselves used ERM data. For the ASM and EC projections, we use the ERM's ratio of the projected productivity growth value in each future year to the projected production expenditure value in 2020 for the ASM and 2017 for the EC (the base years in our data) to determine how many workers are needed per $1 million of expenditures, in 2017$. In other words, we apply the projected productivity growth estimated using the ERM data to the ASM and EC numbers.

To simplify the results, we compare the projected employment across data sources and report only the maximum and minimum effects in each year across all sectors.[CC] We provide a range rather than a point estimate because of the inherent difficulties in estimating employment impacts as well as the uncertainty over how the costs are expended. The reported ranges provide an estimate of the expected magnitude of the cost effect. In Table 10-7, the Motor Vehicle Parts Manufacturing Sector value from the ASM provides the maximum employment estimates per $1 million; the Light Truck and Utility Vehicle Manufacturing Sector value from the EC provides the minimum estimates.

Cost estimates developed for this rule are provided in Chapter 7. We use the technology cost estimates from that chapter to estimate the employment impacts of a change in cost of manufacturing HD vehicles due to this rule. The technology cost estimates (in $ million) are multiplied by the estimates of workers per $1 million in costs. The projected estimates of technology costs and corresponding minimum and maximum estimated employment impacts for each year are shown in Table 10-8, below. The effects are shown in job-years, where a job-year is, for example, one year of full-time work for one person or two years of half-time work for two workers. Increased technology costs of vehicles and parts is, by itself and holding labor intensity and output constant, expected to increase employment by between 800 and 5,300 per year between 2027 and 2031 under this final rule. While we estimate employment impacts, measured in job-years, beginning with program implementation, some of these employment gains may occur earlier as vehicle manufacturers and parts suppliers hire staff in anticipation of compliance with the standards.

[CC] To see details, as well as results for all sources, see "Final Cost Effect Employment Impacts Calculation" in the docket.

Table 10-8: Estimated Employment Effects Due to Increased Costs of Vehicles and Parts (Cost Effect), in Job-Years

Year	Minimum Employment due to Cost Effect	Maximum Employment due to Cost Effect
2027	1,000	5,300
2028	900	5,000
2029	800	4,700
2030	800	4,400
2031	800	4,200

10.2.2.3 The Demand Effect

The demand effect reflects employment changes due to changes in new vehicle sales. If HD vehicle sales decrease, fewer people would be needed to assemble trucks and the components used to manufacturer them. On the other hand, if pre-buy occurs, HD vehicle sales may increase temporarily, leading to temporary increases in employment. RIA Chapter 10.1.1 and Chapter 10.1.2 discuss the factors influencing the effect of final requirements on demand for new HD vehicles, explains why that effect is difficult to quantify, outlines a new method to quantify the impacts and explains how we might use it to estimate pre- and low-buy sales effects in future rulemakings.

EPA received many comments on the proposed rule requesting that we expand our current employment analysis to include demand effects. We have responded to those requests with the example method laid out below to estimate illustrative demand effects due to a change in sales. Pre- and low-buy are short-term sales effects. As such, it should be noted that employment effects due to pre- and low-buy may be short term as well. Some of these effects may also be transitional as workers shift from one sector to another.

Using the illustrative results on pre- and low-buy sales effects as outlined in Chapter 10.1, combined with employment information from the ASM and EC as described above and domestic HD truck production from Wards Automotive Group, we estimate the increase in job-years due to pre-buy in the months before rule implementation, and the decrease in job-years due to low-buy in the months after rule implementation.

We sum the annual employment values from the Motor Vehicle Manufacturing sector (NAICS 3361) and the Motor Vehicle Parts Manufacturing sector (NAICS 3363) from the ASM and EC data sets.[DD] For the ASM, we use data from 2018, 2019 and 2020. For the EC, we only have data from 2017. We then divide the annual employment for each year by domestic truck

[DD] We only sum value from these four digit NAICS code sectors because NAICS codes are nested and summing the more detailed (longer) NAICS codes with the more general (shorter) NAICS codes will result in double counting. We do not use data from the ERM because it provides employment per million dollars in sales as opposed to employment in job-years.

production in that year to get a value for job-years per truck for each year. Employment, production and job-years per truck can be seen in Table 10-9.

Table 10-9: Annual HD Employment and Production

Data Source	Year	Annual Employment	Domestic Truck Production[EE]	Job-Years/Truck Produced
ASM	2020	775,000	6,890,000	0.112
ASM	2019	821,000	8,380,000	0.098
ASM	2018	810,000	8,510,000	0.095
EC	2017	788,000	8,150,000	0.097

It should be noted that year-over-year percentage change in domestic truck production is greater than the year-over-year percentage change in annual employment. In addition, production fell between 2018 and 2019, though employment increased in the directly affected sectors in our analysis. This indicates that employment changes in the sectors measured does not always follow changes in domestic HD truck production. This could be due to many factors, including workers transitioning between manufacturing jobs or sectors. There may also be lag factors, production changes happening ahead of employment changes, or future planning by manufacturers. For example, if manufacturers believe changes in production will be temporary, they may not want to change employment by much, with the understanding that they will need to revert back to previous levels of employment after production returns to previous levels. Additionally, employers may not want to face costs associated with layoffs, preferring to reduce employment through attrition.

Using the data in the table above, we estimate the average job-years per truck in the directly affected segments to be 0.101 job-years per HD truck. We apply this ratio to the estimated maximum total annual change in sales in 2026 due to pre-buy and in 2027 due to low-buy as shown in the illustrative sales effects example in Chapter 10.1.2. This results in an estimate of a change in job-years due to a change in demand. Table 10-10 shows the results.

Table 10-10: Estimated Maximum Total Change in Sales and Illustrative Change in Job-Years due to Demand Effect

Year	Max Total Annual Sales Effect	Change in Job-Years
2026 (pre-buy)	4,700	450
2027 (low-buy)	(6,600)	(640)

We assume these demand effects would be short-term, as they would be due to a short-term change in demand, with our illustrative pre-buy estimates ranging up to 8 months, and low-buy up to 12 months. As mentioned above, employment changes may lag production changes.

[EE] This data comes from Wards Automotive Group. U.S. Vehicle Production by Manufacturer, UsaPr05.xls

10.2.2.4 Summary of Employment Effects in the Motor Vehicle Sector

As explained above, the overall effect of the final standards on motor vehicle sector employment depends on the relative magnitude of the demand, cost, and factor-shift effects, as described by Morgenstern, et al.[13] We quantitatively estimate employment impacts of the standards due to the cost effect. To provide a sense of its magnitude, EPA provides a range of estimates of the cost effect. Due to a lack of data, we do not estimate factor-shift effects. And though we are not estimating demand employment effects for this rule, we do outline a method and illustrate how that method could be used to estimate these effects in Chapter 10.2.2.3. This proposed method relies on the illustrative results for pre- and low-buy outlined in Chapter 10.1.2.2.

For the regulated sector, the partial employment impact due to the effect of increased manufacturing costs due to compliance activities is estimated to range between 800 and 5,300 job-years between 2027 and 2031. We expect the demand effect to reduce these employment increases, as qualitatively discussed above. Finally, we are unable to predict the direction of the factor-shift effect.

10.2.3 <u>**Employment Impacts on Related Sectors**</u>

The rule may affect employment in several related sectors including downstream on purchasers and dealers.

10.2.3.1 Effects on Purchasers of Heavy-Duty Vehicles

Because of the diversity of HD vehicles, entities from a very wide range of transportation sectors will be purchasing vehicles subject to these standards. As discussed in Chapter 10.1, vehicles subject to these standards are likely to be more expensive to purchase compared to vehicles not subject to the standards. HD vehicles are typically commercial, and typically provide an "intermediate good:" that is, they are used to provide a commercial service, rather than being a final consumer good. As a result, the higher costs of the vehicles may result in higher prices for the services provided by these vehicles, and potentially reduced demand for those services. In turn, there might be less employment in the sectors providing those services.

One commenter on the ANPR provided data indicating that, between 2010 and 2018, the up-front costs of HD vehicles were approximately 10 to 16 percent of per-mile costs.[21] Given that we expect the potential impacts of the final standards on up-front costs of vehicles to be only a few percent of total vehicle cost, any increase in per-mile costs are likely to be less than 1 percent (even a 5 percent increase in vehicle costs would result in only a 0.8 percent increase in per-mile costs, at the high end of the range). Therefore, we expect only negligible to small impacts on transportation services demand, and related employment in transportation services sectors. Per-mile cost increases for some sectors will be higher than this average, while they will be lower in other sectors due to factors such as differences in how the vehicles are used, including average mileage accumulation of the vehicles in the sector. The actual effects on demand for the services and related employment will depend on cost pass-through, and responsiveness of demand to transportation cost increases.

Lengthening the warranty period may provide some positive impacts on employment for vehicle purchasers. Some commenters submitted comments on the proposed rule conveying concerns over the effects on productivity of downtime due to problems with emission control systems.[22] As discussed in Chapter 7.2.3, the extended warranty provisions finalized in this rule are expected to not only reduce repair costs for vehicle purchasers, but may also provide incentives to manufacturers to improve quality and thus reduce the need for repairs. These effects are expected to reduce costs, and thus mitigate adverse impacts on employment for vehicle purchasers.

10.2.3.2 Effects on Heavy-Duty Vehicle Dealers and Service Providers

If sales of HD vehicles decrease, then HD vehicle dealers will have fewer sales, and may employ fewer people as a result. At the same time, dealers, and other independent service repair shops, often provide repair and maintenance services. The extended warranty provisions are expected to facilitate repair and maintenance of emission control system components, which could result in increased demand for workers servicing vehicles. On the other hand, as discussed in Chapter 7.2.3, the extended useful life provisions are also be expected to provide incentives to OEMs to improve quality and may reduce the need for warranty claims. Thus, effects on employment for service providers, including dealers, may be positive or negative.

Similarly, as discussed in Preamble Section IV.B.3, this rule aims to improve access to serviceability information to improve owner experiences operating and maintaining HD engines and provide greater assurance of long-term in-use emission reductions by reducing likelihood of occurrences of tampering. One commenter on the ANPR noted that it is currently difficult for anyone other than dealers to service vehicles, and commented on the proposed rule that finalizing the proposed serviceability provisions will help drivers maintain the emissions equipment themselves. It is possible that improving serviceability will improve maintenance due to lower costs of conducting service.[23] It is also possible that improved serviceability may shift some of the work, and thus employment, from dealers to independent service repair shops or to the owners themselves, with an unclear impact on the overall level of employment.

10.2.4 Summary of Employment Impacts

Employment in the HD manufacturing sector depends on three effects: how the effects of the standards on vehicle prices affect demand for new vehicles (demand effect); the labor demand needed to meet the standards (cost effect); and any change in labor intensity of production due to complying with the standards (factor-shift effect). In Chapter 10.2.2, we outline a method to estimate demand effects on employment using the illustrative sales effects results discussed in Chapter 10.1.2. We are unable to quantify the factor-shift effect, and therefore we are unable to estimate the impact on net employment in the HD manufacturing sector. To give an estimate of the range of cost-effect-related employment changes due to the final regulation, the analysis estimates a range between 1,000 and 5,300 job-years in 2027, with impacts falling each year. By 2031, estimated impacts range from 800 to 4,200 job years. For comparison, in May 2021, the Bureau of Labor Statistics reports about 244,000 employees in Motor Vehicle Manufacturing.[24]

Other sectors that sell, purchase, or service HD vehicles might also experience employment impacts due to the standards. The effects on these sectors will depend on the degree of cost pass-through to prices for HD vehicles and the effects of useful life and warranty requirements on demand for vehicle repair and maintenance.

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Chapter 11 Small Business Analysis

This chapter presents our analysis of the economic impacts of this action on small entities that are subject to the highway heavy-duty engine and vehicle provisions of this rule. These are: heavy-duty vehicle manufacturers, heavy-duty secondary vehicle manufacturers, and heavy-duty alternative fuel engine converters. Other entities that are subject to the rule are either not small (e.g., engine and incomplete vehicle manufacturers) or are not expected to incur any additional burden from the rule (e.g., manufacturers in sectors other than highway heavy-duty engines and vehicles and that are subject only to the regulatory amendments contained in Section XII of the Preamble for this rule).

11.1 Definition and Description of Small Businesses

Under the Regulatory Flexibility Act (5 USC 601 et seq.), a small entity is defined as: (1) a business that meets the definition for small business based on the Small Business Administration's (SBA) size standards;[1,A] (2) a small governmental jurisdiction that is a government of a city, county, town, school district or special district with a population of less than 50,000; or (3) a small organization that is any not-for-profit enterprise which is independently owned and operated and is not dominant in its field.

This analysis considers only small business entities subject to the rule. Small governmental jurisdictions and small not-for-profit organizations are not subject to the rule as they have no certification or compliance requirements.

11.2 Overview of the Heavy-Duty Program and Type of Entities Covered

As described in the preamble and elsewhere in this RIA, this rulemaking sets out a comprehensive approach to reduce air pollution from highway heavy-duty engines. The key provisions can be grouped into three broad categories: 1) reducing emissions under a wider range of engine operating conditions than those covered in existing requirements, including refueling events (e.g., revised exhaust and refueling emission standards and updated test procedures); 2) maintaining emission control over a greater portion of an engine's operational life (e.g., lengthened useful life and regulatory emission warranty periods), and 3) providing manufacturers with flexibilities to meet the new emission standards while clarifying our regulations.

While the rule also includes regulatory amendments for sectors other than highway heavy-duty engines and vehicles, these amendments for other sectors correct, clarify, and streamline existing regulatory provisions, and they will impose no additional burden on small entities in these other sectors.

There are three categories of highway heavy-duty engine and vehicle entities that are subject to the rule:

[A] EPA relied on the 2019 SBA size standards that were current at the time of the analysis. We acknowledge that new size standards went into effect after we conducted the analysis, but those new standards would not change the analysis. See *https://www.sba.gov/article/2022/oct/03/sba-issues-final-rule-adopt-naics-2022-small-business-size-standards*. The October 2022 version of the size standards are available online: *https://www.sba.gov/document/support-table-size-standards*.

- Heavy-duty engine manufacturers
- Heavy-duty conventional vehicle manufacturers, including incomplete and secondary vehicle manufacturers
- Alternative fuel engine converters

Heavy-duty engine manufacturers have been developing, testing, and certifying engines for many years in compliance with EPA rulemakings adopted under the CAA. These companies will be required to produce engines that meet new emission standards and certify them using revised test procedures. The heavy-duty engine manufacturers that certify engines to EPA's program include no small entities.

Heavy-duty vehicle manufacturers are one of two types. The first type of company manufactures and certifies a complete or incomplete vehicle and its associated engine.[B] These companies are not small entities. The second type of company manufactures a vehicle of its own design using a certified engine or incomplete vehicle produced by a different company. Manufacturers that finish an incomplete vehicle produced and certified by a different company (i.e., "secondary manufacturers") complete the vehicle by adding the truck body and other equipment. While these secondary manufacturers are not required to certify with EPA (because they use an incomplete vehicle certified by another company), they may incur costs to accommodate any changes made to the certified incomplete vehicle to meet the new emission requirements. Several secondary manufacturers are small entities under the SBA definition, and the economic impacts of the rule on them are described in Section 11.3.

Alternative fuel engine converters are also subject to the rule. Two of these companies are small entities under the SBA definitions, and the impacts on them are described in Section 11.4. Finally, Section 11.5 contains a summary table of the expected economic impacts on small entities subject to the rule.

11.3 Impacts on Small Entities: Heavy-Duty Secondary Vehicle Manufacturers

A secondary vehicle manufacturer is defined as anyone that produces a vehicle by modifying a complete vehicle or completing the assembly of a partially complete vehicle, although a manufacturer controlled by the manufacturer of the base vehicle (or by an entity that also controls the manufacturer of the base vehicle) is not a secondary vehicle manufacturer; instead, both entities are considered to be one manufacturer (40 CFR 1037.801 definition of secondary vehicle manufacturer). EPA's heavy-duty vehicle program allows an engine manufacturer to introduce partially complete vehicles into U.S. commerce to be completed by a secondary vehicle manufacturer (see 40 CFR 1037.622). These incomplete vehicles will typically be certified by the engine manufacturer. The program also allows a manufacturer to introduce complete vehicles into U.S. commerce for modification by a small manufacturer (e.g., a recreational vehicle manufacturer); these also will typically be certified by the engine manufacturer. The provisions specify that a secondary vehicle manufacturer may finish assembly of partially complete vehicles if it obtains a vehicle that is not fully assembled with the intent to manufacture a complete vehicle in a certified configuration.

[B] See the definition of "vehicle" in 40 CFR 1037.801.

The impacts of this rule on secondary vehicle manufacturers are different depending on whether the vehicle is produced using a compression-ignition incomplete vehicle or a spark-ignition incomplete vehicle.

Secondary vehicle manufacturers that produce a heavy-duty vehicle using a compression-ignition incomplete vehicle are not expected to need to modify their manufacturing or other processes to comply with the rule. As discussed in Chapter 3 of this RIA, compression-ignition engine manufacturers are expected to achieve the new criteria pollutant engine emission standards by modifying the engine and aftertreatment system technologies already applied to these engines to meet the existing standards (e.g., selective catalytic reduction). As a result, the engine and aftertreatment systems needed to meet the new criteria pollutant emission standards are expected to be similar to the systems vehicle manufacturers install today and secondary vehicle manufacturers are not expected to need to redesign or modify their products or production processes to accommodate these compression-ignition engine-based certified systems. Therefore, we do not expect secondary vehicle manufacturers that use compression-ignition incomplete vehicles to experience adverse economic impacts as a result of this rule.

The analysis of impacts of the final rule on secondary vehicle manufacturers that produce a heavy-duty vehicle using a spark-ignition incomplete vehicle includes multiple steps. Similar to manufacturers of compression-ignition engines, Chapter 3 indicates that spark-ignition engine manufacturers are expected to achieve the new criteria pollutant engine exhaust emission standards by modifying the engine and aftertreatment system technologies already applied to these engines to meet the existing standards (e.g., three-way catalysts). As a result, the engine and aftertreatment systems are expected to be similar to the systems vehicle manufacturers install today and secondary vehicle manufacturers are not expected to need to redesign or modify their products or production processes to accommodate spark-ignition engine-based certified systems. Therefore, we do not expect secondary vehicle manufacturers that use spark-ignition incomplete vehicles to experience adverse economic impacts as a result of the new spark-ignition criteria pollutant exhaust emission standards.

However, these spark-ignition incomplete vehicles will also be required to comply with new refueling emission standards for vehicles fueled by gasoline, other volatile liquid fuels, and gaseous fuels. Compliance with these standards may require some secondary vehicle manufacturers to change their manufacturing or other processes to accommodate compliant refueling emission control systems. Historically, an incomplete vehicle that is sold to a secondary vehicle manufacturer includes the fuel system and its evaporative emission controls as part of the incomplete vehicle's certified configuration. When manufacturers of chassis-certified complete heavy-duty vehicles (i.e., Classes 2b and 3) adopted ORVR technology to meet refueling emission standards (59 FR 16262, April 6, 1994), the design changes were contained in the fuel system and did not require changes to the vehicle body to accommodate the new technology (i.e., filler door location and designs remained the same). To comply with the final refueling emission standards being adopted in this rule, manufacturers of incomplete heavy-duty vehicles fueled by volatile fuels may add to or replace existing components of the evaporative emission control systems currently being installed on incomplete vehicles. We expect incomplete vehicle manufacturers will strive to design compliant ORVR systems that maintain continuity from previous fuel system designs, minimizing the need for vehicle body redesign to accommodate any changes to emission control systems. Our expectation is reinforced by the comments we received on the proposal to this rule, in which ORVR suppliers expressed confidence in the relationship between engine OEMs and delegated assemblers (i.e., secondary vehicle manufacturers) to effectively implement refueling requirements for incomplete heavy-duty vehicles.[C] Secondary manufacturers that finish the vehicle bodies have many years of experience installing evaporative emission control systems as delegated assemblers, and the ORVR instructions are expected to add very few, if any, steps to the evaporative system instructions currently provided by the chassis manufacturers.

Secondary vehicle manufacturers are not required to certify with EPA, and so we do not have a list of secondary manufacturers that will be subject to the rule. Instead, we used the Hoovers D&B database to identify small companies engaged in the Motor Vehicle Body Manufacturing (NAICS Code 336211 with 1,000 employees or fewer) or Motor Home Manufacturing (336213 with 1,250 employees or fewer) sectors.[2] We limited our search to companies located in the United States. This approach is reasonable because it is unlikely that a foreign entity would purchase a certified incomplete vehicle from a manufacturer located in the United States, transport that vehicle to a location outside the United States, complete the vehicle, and then export that completed vehicle back into the United States with the associated transportation costs. If there were such a company, the cost of the additional transportation to and from the assembling country would likely exceed the expected costs of compliance with the rule ($2,528 per company per year; see below). Also, the additional transportation costs would likely make the completed vehicle uncompetitive in the U.S. market (with the exception of luxury trucks or recreational vehicles, in which case the company would likely have revenue that can accommodate the costs of complying with the requirements).

We adjusted the initial list of 1,190 companies to remove those that are subsidiaries of another company (they have a parent or ultimate parent company). For the Motor Vehicle Body Manufacturing sector, we further adjusted the list to reflect only companies engaged in truck and bus body manufacturing (as identified by their Standard Industrial Classification (SIC) code) and removed companies that do not make truck bodies or that make light-duty trucks (these companies are not subject to the rule).[D] For the Motor Home Manufacturing sector, we selected only companies engaged in motor home manufacturing (using their SIC code) and removed van conversions (those are light-duty vehicles not subject to this rule). Finally, we removed companies with four or fewer employees because it is not likely that a company with fewer than five total employees manufactures completed trucks. It should be noted that we also removed one company from the list that does not appear to have become operational (the $100 annual revenue reported in Hoovers for this company was likely a placeholder). This procedure yielded a list of 249 small entities engaged in the manufacture of secondary vehicles and thereby potentially subject to the refueling standards. It should be noted that the final list of companies does not distinguish between those that produce spark-ignition and compression-ignition vehicles.

[C] See comments from the Manufacturers of Emission Controls Association (EPA-HQ-OAR-2019-0055-1320) and Ingevity Corporation (EPA-HQ-OAR-2019-0055-1213).

[D] Removed: cranes, overhead traveling; dump truck lifting mechanism; fifth wheel, motor vehicle; truck beds; truck bodies (motor vehicles); truck bodies and parts; truck cabs, for motor vehicles; truck tops; van bodies.

We estimated the impacts of the rule on these small entities using the following information. We assume each company will have one-time costs associated with reviewing new instructions for ORVR (10 hours/family), possible vehicle design R&D if the change to the evaporative control cannister requires different mounting assemblies or the area where it is mounted must be adjusted (8 hours/family), and training associated with installing the new ORVR system (1 hour/family). We also assume a recurring production cost for installing the new ORVR system (1 hour/unit). Assuming 2 families with 20 unit per family for each company (40 vehicles produced per year) and $43.58/hour, the total cost of the program in the first year is expected to be $2,528 per company. We then compared this to the annual revenue reported in Hoovers for each of the small entities.[E]

For these secondary vehicle manufacturers, the expected costs of $2,528 is less than 1 percent of revenue for 201 of the companies and between 1 and 3 percent of revenue for 48 companies. Figure 11-1 contains a graphical representation of the revenue distribution of these companies. The impacts are summarized in Table 11-1 presented in Section 11.5.

[E] While EPA's guidance for Regulatory Flexibility Act analysis specifies that annual sales should be used in the analysis for small companies, it also indicates that "revenue or receipts (though technically different than sales) can usually serve as a reasonable proxy for sales." Footnote 19, page 21. EPA's Action Development Process, Final Guidance for EPA Rulewriters: Regulatory Flexibility Act as amended by the Small Business Regulatory Enforcement Fairness Act. November 2006.

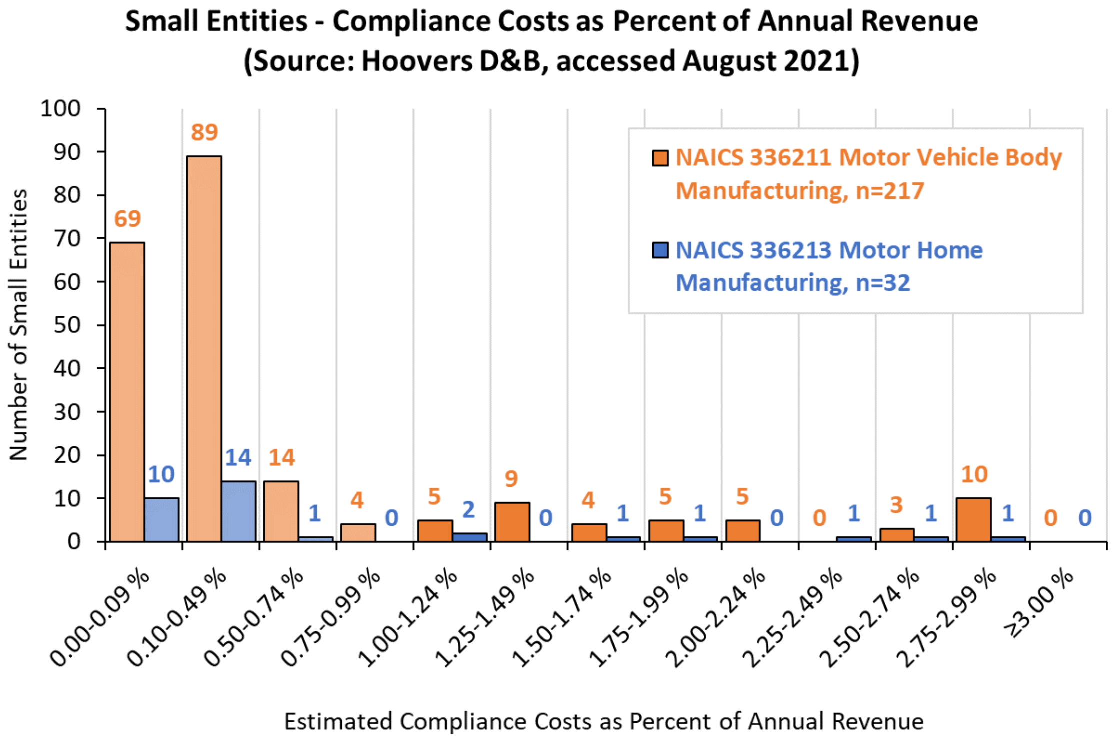

Figure 11-1: Secondary Vehicle Manufacturers, Estimated Impacts as a Percent of Annual Revenues

11.4 Impacts on Small Entities: Heavy-Duty Alternative Fuel Engine Converters

Companies that convert compression-ignition or spark-ignition heavy-duty engines to use alternative fuels are also subject to the final rule. Alternative fuel converters do not always need to certify the conversions to the emission standards. However, they may need to perform testing to show that modified engines continue to meet the applicable new standards as part of the process for meeting 40 CFR part 85, subpart F, to be exempt from the tampering prohibition. For this analysis, we conservatively assumed the process for these fuel converters will include emission testing using a new test procedure (SET if converting an SI engine or LLC if converting a CI engine) and preparing the data for submission to EPA to obtain an EPA Certificate of Compliance.

EPA identified 5 companies that convert heavy-duty engines to run on alternative fuel. These are companies that certified engines with EPA as of 2020. We obtained their employment and annual revenue numbers from Hoovers. Two of these companies are small entities under the SBA definitions based on annual receipts.

To estimate the impacts of the rule on these small entities we assume that each company will have one-time costs associated with developing and performing the emission test of 20 hours/family. There are no recurring costs per vehicle expected. Assuming 4 families and $43.58/hour, the total cost of the program in the first year is expected to be $3,486 per company.

Our examination of the annual revenues for the two small alternative fuel engine converters reveals that these costs, $3,486 per company per year, is not expected to impose a significant impact on either of them. Even a low significant impact threshold of 1 percent of revenue would correlate to an annual revenue of $348,640 or less. Each of the two small alternative fuel engine converters has annual revenues in excess of that amount and therefore will not experience a significant impact from the rule. These results are summarized in Table 11-1 presented in Section 11.5.

11.5 Summary Table of Impacts on Small Businesses Subject to the Rule

Table 11-1: Summary of Impacts on Small Businesses Subject to the Rule

NAICS Category	**Sector description**	**SBA Threshold**	**Number of small companies subject to the rule**	**Impact as percent of annual revenue, number of small companies**		
				≥3%	1-3%	<1%
336211	Secondary manufacturer: Motor vehicle body manufacturing	1,250 employees	217	0	41	176
336213	Secondary manufacturer: Motor home manufacturing	1,250 employees	32	0	7	25
Total secondary manufacturer			249	0	48	201
811198	Alternative fuel engine converters	$8.0 million annual receipts	2	0	0	2
TOTAL			*251*	*0*	*48*	*203*

Chapter 11 References

1 U.S. Small Business Administration. Table of Small Business Size Standards Matched to North American Industry Classification System Codes. 2019 Version. Available online: https://www.sba.gov/sites/default/files/2019-08/SBA%20Table%20of%20Size%20Standards_Effective%20Aug%2019%2C%202019_Rev.pdf

2 NAICS Association. NAICS & SIC Identification Tools. Available online: https://www.naics.com/search