STATE OF CALIFORNIA
AIR RESOURCES BOARD

Heavy-Duty Engine and Vehicle	**)**	
Omnibus Regulation and Associated	**)**	**Hearing Date:**
Amendments; Proposed Rulemaking;	**)**	**August 27, 2020**
Initial Statement of Reasons	**)**	

SUPPLEMENT TO THE COMMENTS OF THE TRUCK AND ENGINE MANUFACTURERS ASSOCIATION

August 21, 2020

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STATE OF CALIFORNIA
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1. Introduction

The Truck and Engine Manufacturers Association ("EMA") hereby submits this supplement to EMA's comments in opposition to the proposed Heavy-Duty Engine and Vehicle Omnibus Regulations and Associated Amendments (the "Omnibus Regulations" or "Low-NO_x Regulations") that the California Air Resources Board ("CARB") has proposed to adopt at a Board hearing scheduled for August 27, 2020. This supplement adds to the point, discussed at pages 29-30 of EMA's comments, that CARB has done nothing to assess the efficacy of its proposed Omnibus Low-NO_x Regulations.

2. CARB Has Done Nothing To Assess Or Establish The Efficacy Of Its Proposed Low-NO_x Regulations

EMA's comments point out that CARB has taken no steps and has provided no evidence in the rulemaking record to demonstrate that its proposed Low-NO_x Regulations will be effective at reducing ozone levels in the South Coast Air Basin (SoCAB). In that regard, EMA submitted a report prepared by the Ramboll Group (Exhibit "F" to EMA's comments) confirming that a "NO_x-disbenefit" phenomenon still exists in portions of the SoCAB. The NO_x-disbenefit phenomenon refers to the fact that NO_x reductions actually can lead to increases in ozone in "VOC-limited" regions of the SoCAB, such as the more heavily-populated areas near downtown Los Angeles. Ramboll's report (Exhibit "F") documents the continuing persistence of NO_x-disbenefits in the central and western portions of the SoCAB, including Los Angeles.

More recently, Ramboll has assessed whether the recent significant COVID-related reductions in ozone-precursor emissions, specifically NO_x, have led to actual corresponding reductions in ozone. As detailed in Ramboll's supplemental report (attached hereto as Exhibit "F.1",) notwithstanding NO_x reductions of approximately 20% when comparing June 2019 with June 2020 (months that had similar meteorology), ozone levels were similar at the key "design-value" monitoring in the SoCAB (and actually were slightly higher in downtown Los Angeles). Ramboll's supplemental analysis confirms that ozone levels in the SoCAB are, at best, currently unresponsive even to significant 20% reductions in ambient NO_x levels, reductions that are well beyond those that could be achieved through implementation of the proposed Low-NO_x Regulations.

Ramboll's analysis and findings confirm that the proposed Low-NO_x Regulations likely will not be effective in reducing ozone levels in the SoCAB. Just as important, CARB has done nothing to establish any different conclusion. The complete lack of evidence of the actual efficacy of CARB's proposed Low-NO_x Regulations is another factor establishing their invalidity.

Respectfully Submitted,

TRUCK AND ENGINE
MANUFACTURERS ASSOCIATION

Effects of 2020 COVID-19 NOx Reductions on Ozone in the SoCAB

Preliminary Analysis of 2019 & 2020 Met and Ozone Changes; Top Down NOx Emissions; Ozone Isopleth Analysis

Ralph Morris and Lynsey Parker

August 17, 2020

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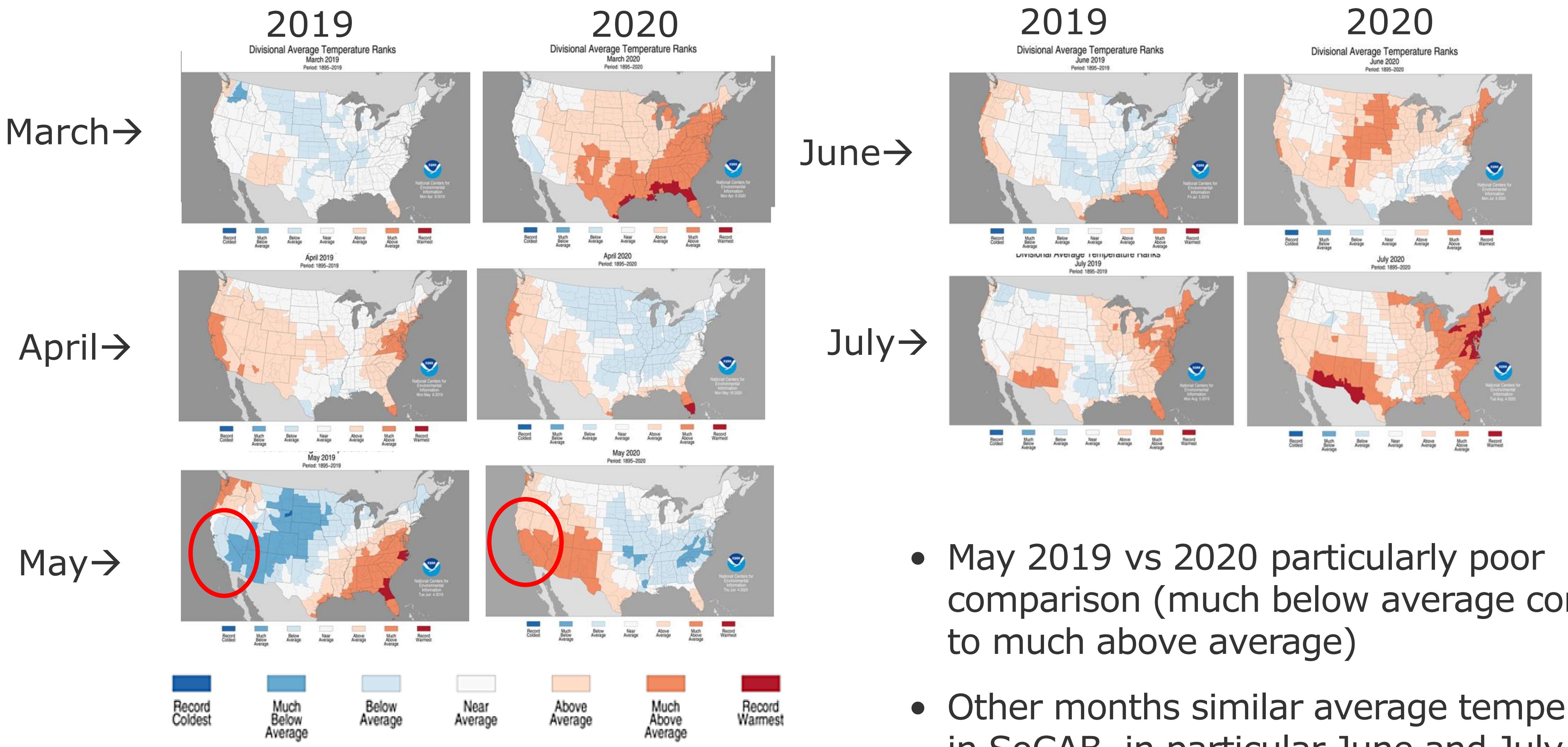
Outline

- Overview of Proposed Work
- Meteorological Characterization
- Top Down Emissions Characterization
- Observed Ozone Changes
- Isopleth Discussion
- Recommendations

Testing Hypothesis

- The current chemical regimes in some locations of the South Coast Air Basin (SoCAB) may be VOC-limited such that NOx emissions reductions due to the COVID Shelter-in-Place (SiP) orders may cause ozone increases or at least no changes in ozone.
- Do modeled ozone estimates respond in the same fashion as ozone observations in response to the COVID NOx emission reductions?
- Phase I Technical Approach
    - Analyze observed and modeled changes in ozone between 2019 and 2020
    - First determine whether 2019 and 2020 meteorological conditions have similar ozone formation conditions such that the signal of the COVID NOx emissions reduction can be detected through variations in meteorology.
    - Analyze changes in NOx concentrations to develop top-down adjustment factors for adjusting 2020 NOx emissions to account for COVID
    - Model 2019 and 2020 to see whether models respond to the changes in NOx emissions in the same fashion as observed ozone changes.
    - Develop plan for bottoms-up adjustment of emissions to account for COVID.

Meteorological Characterization: Average Temperatures



- May 2019 vs 2020 particularly poor comparison (much below average compared to much above average)
- Other months similar average temperatures in SoCAB, in particular June and July comparable

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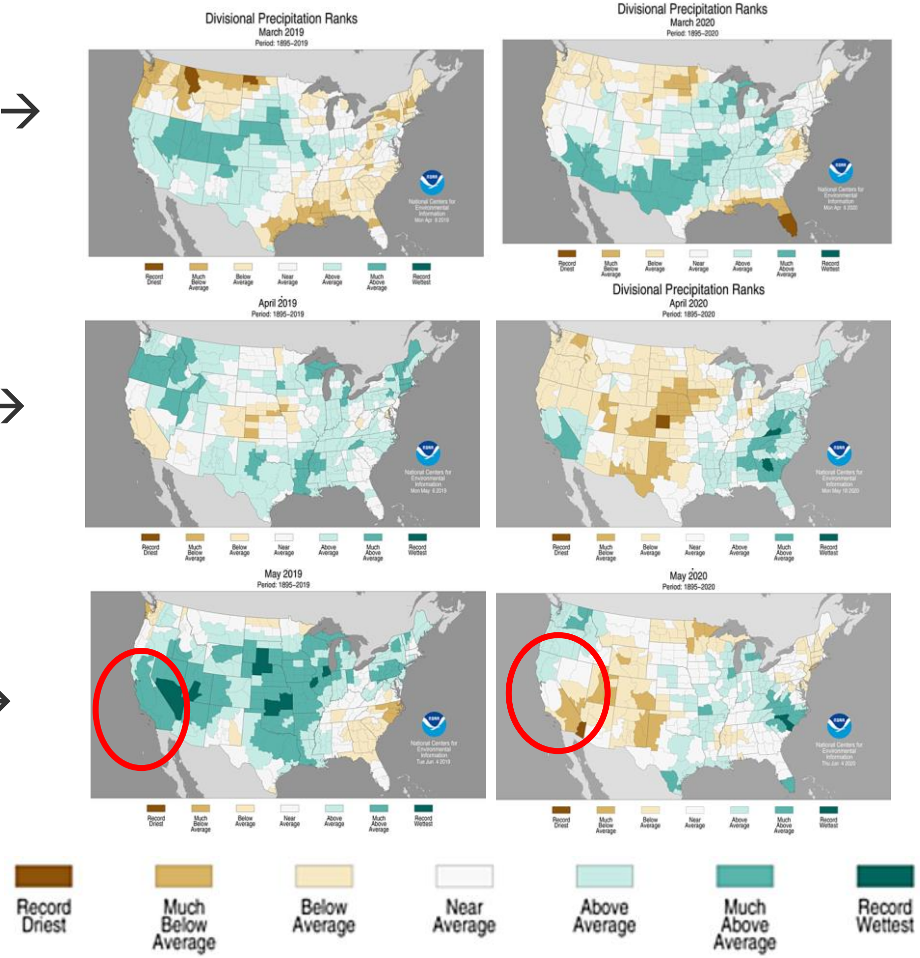
Period: 1895 – 20xx

https://www.ncdc.noaa.gov/temp-and-precip/us-maps/

Meteorological Characterization: Precipitation

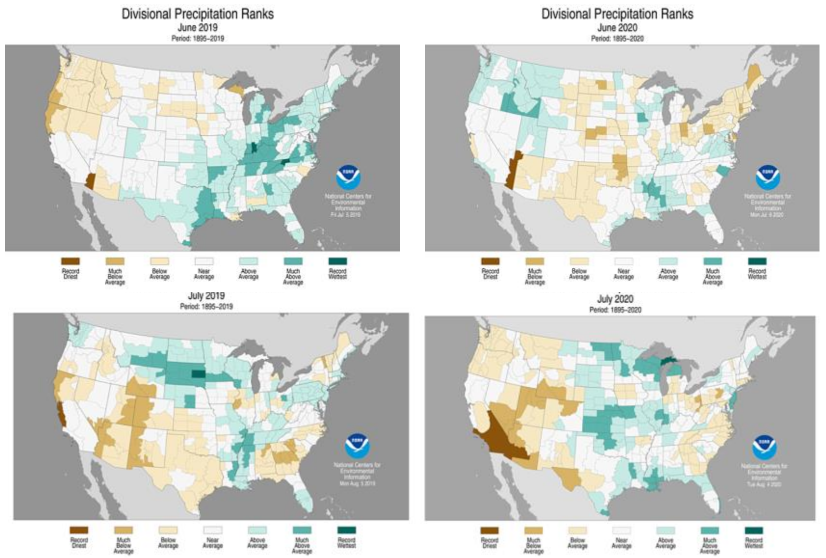
2019 2020

March→
April→
May→



2019 2020

June→
July→



- May 2019 vs 2020 particularly poor comparison, May 2019 much above average precipitation
- Other months reasonably similar average temperatures in SoCAB
  - July 2020 record driest

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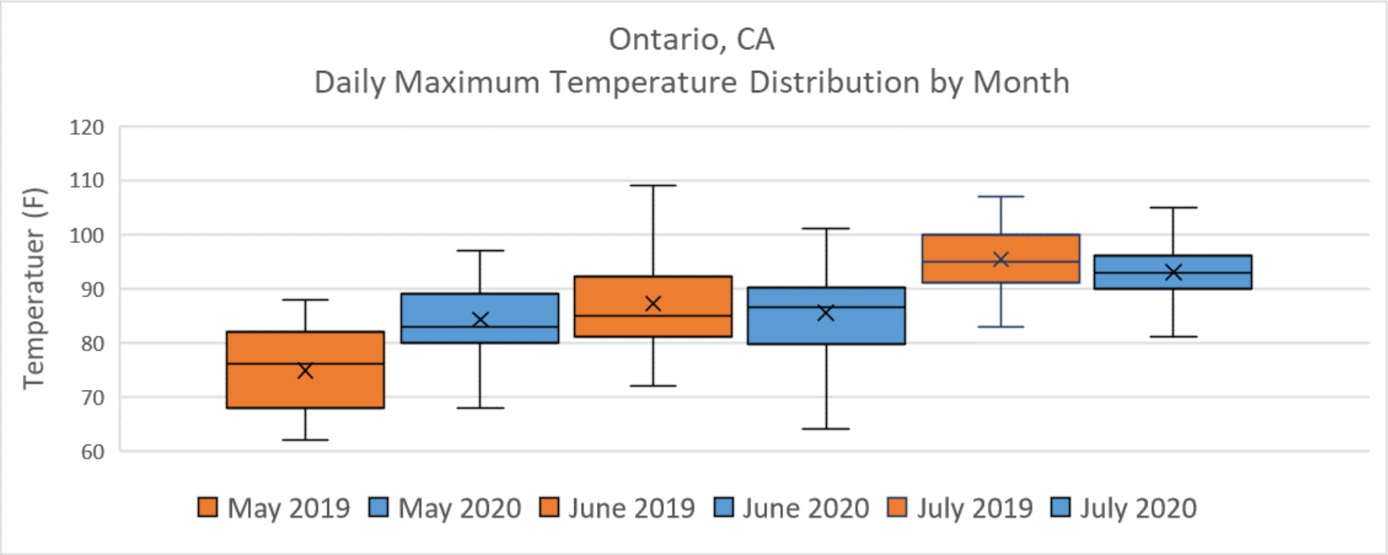
Period: 1895 – 20xx

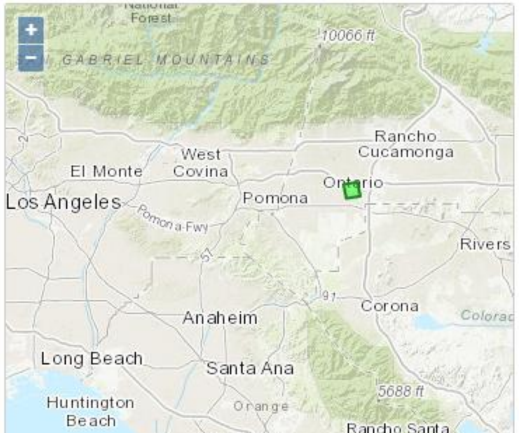
Meteorological Characterization: Local meteorological site



Non-coastal site: Ontario International Airport (KONT)

Temperature Distribution Plots:

- Whiskers = Max/min
- Boxes = 25 – 75 %
- Mid bars = medians
- Crosses = means



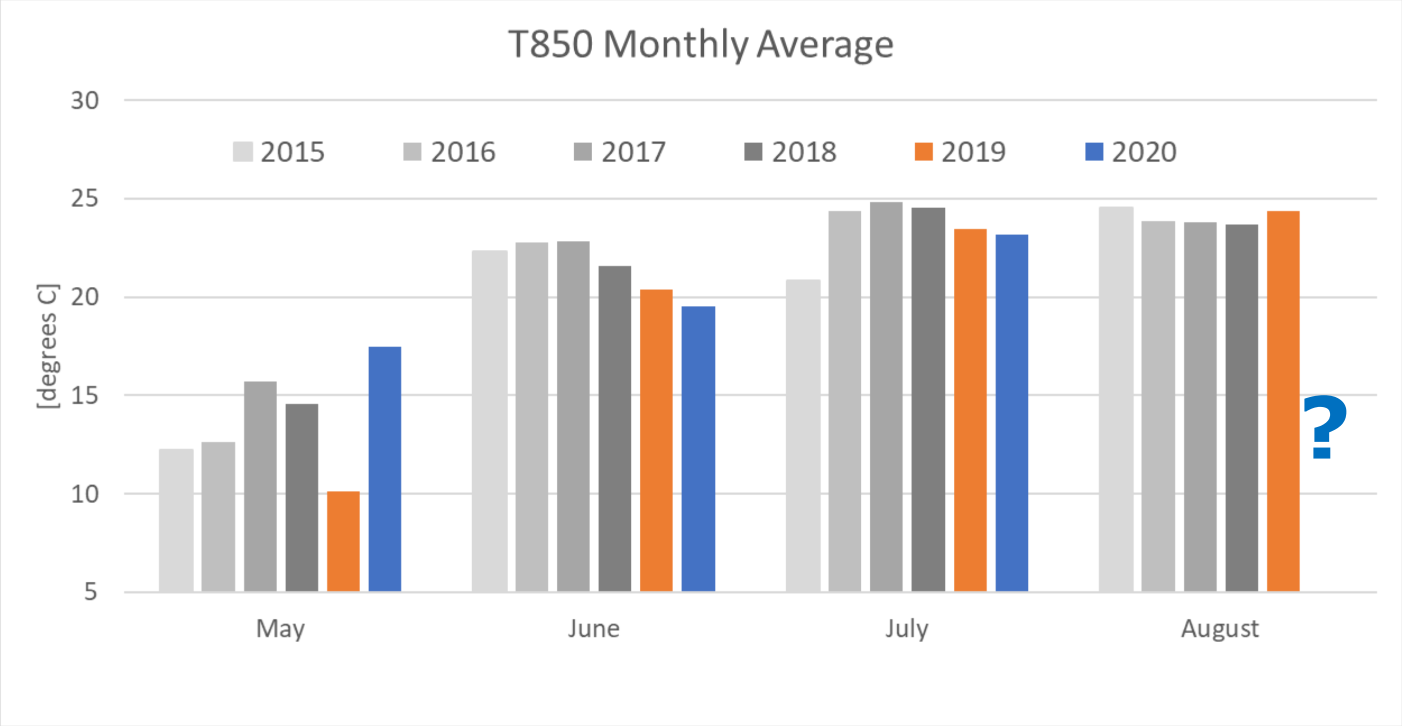
Ontario International Airport (KONT)

Lat: 34.05316°NLon: 117.57685°WElev: 906ft.

- May 2019 much cooler than May 2020
- June and July reasonably similar temps

Meteorological Characterization: "T850" metric

- T850 is the temperature at 850 mb (~ 1,500 meters) an indicator of inversion strength
  - Stronger inversion -> higher pollution in SoCAB
  - Correlates with surface temperatures
  - High ozone at Crestline (i.e. ~design value monitor) occurs when T850 (San Diego) is high
  - 2017 high ozone in SoCAB was due (in large part) to high T850



- Additional years (2015-2018) added for context:
  - May 2019/2020 anomalously low and high
  - June 2019 and 2020 similar, both low
  - July 2019 and 2020 similar and within range of other years

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https://ruc.noaa.gov/raobs/

Meteorological Characterization: Summary

- May 2019 and 2020 are anomalous compared to recent years, and very different from each other
  - ➢ May 2020 is much more conducive to ozone formation than May 2019
- June and July 2019 and 2020 are reasonably similar in terms of surface temperatures and T850 therefore similar in terms of ozone formation potential
  - ➢ Performing AQ modeling and comparing June/July 2019 and 2020 years is reasonable

Top Down Emissions Characterization: Overview

Goal:

Quantify COVID-19 response NOx emissions reductions and account for recent trends

Two Methods:

1. Surface NOx concentrations (7 sites in SoCAB, spanning basin)
    - Two near road sites (Long Beach and Anaheim)
    - Monthly average metric
    - Compare with 2015-2019 baseline period
2. Satellite NO_2 columns

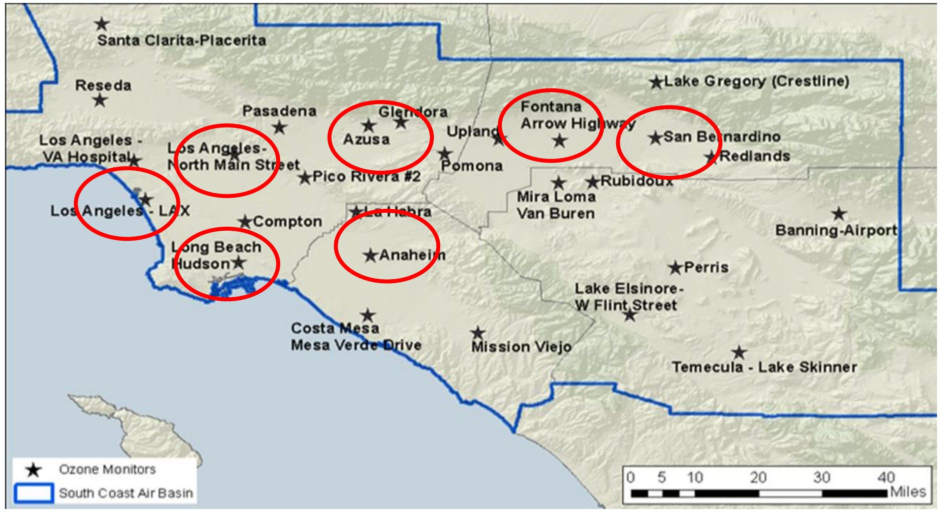

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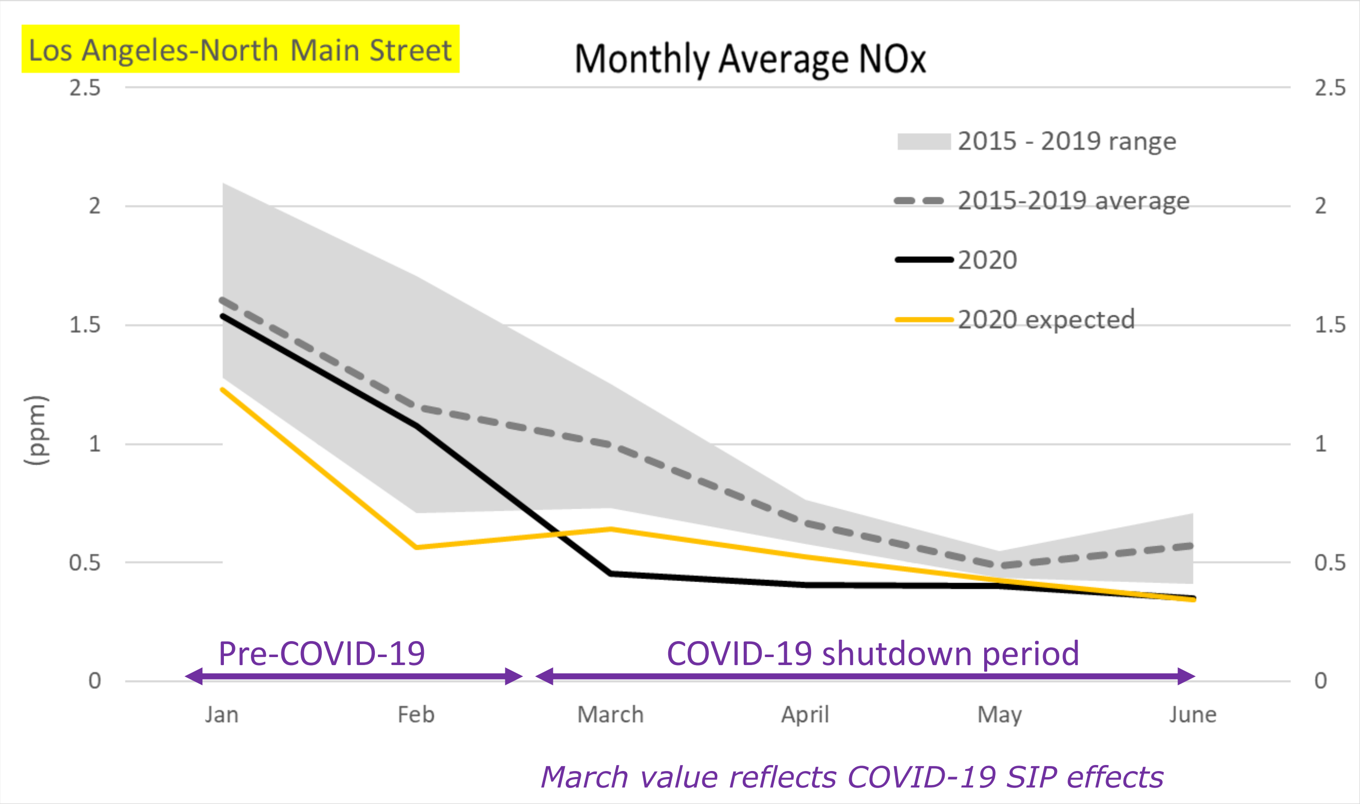
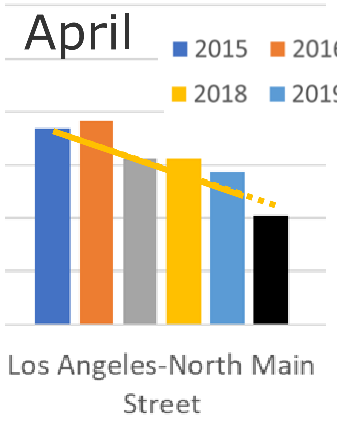
Exhibit F.1

Top Down Emissions Characterization: Surface NOx Methodology



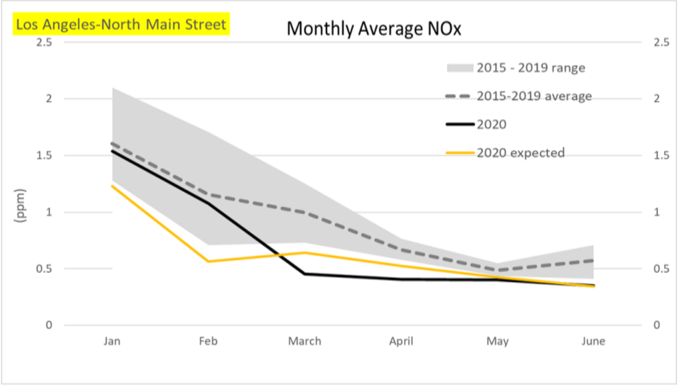
Example extrapolation to calculate "2020 expected"

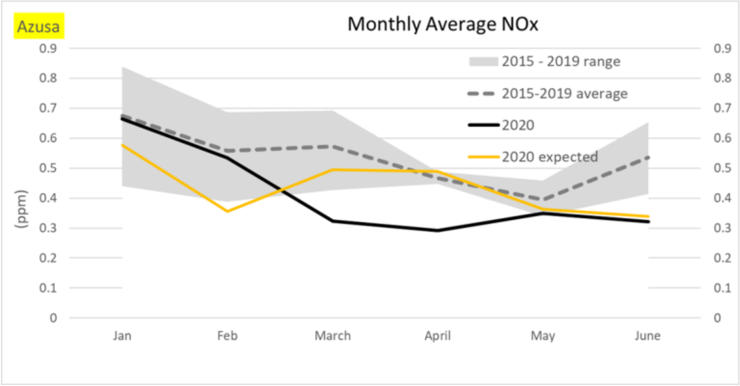


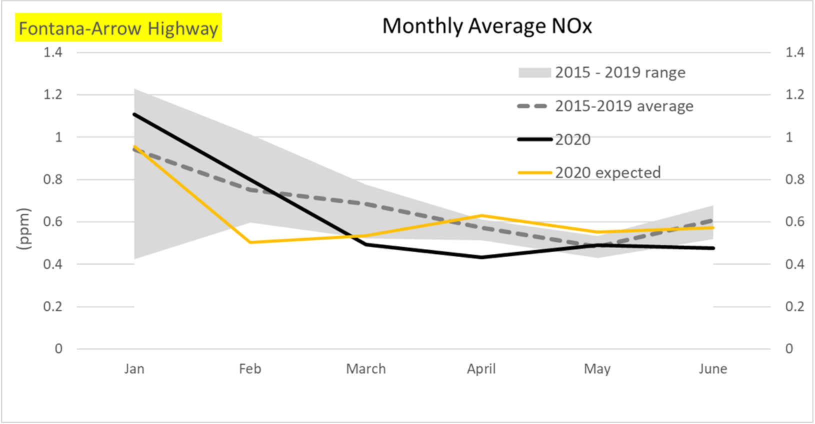
- "2020 expected" is based on monthly linear extrapolation of 2015-2019
- Shutdown occurred mid-March (impacts March concentrations)

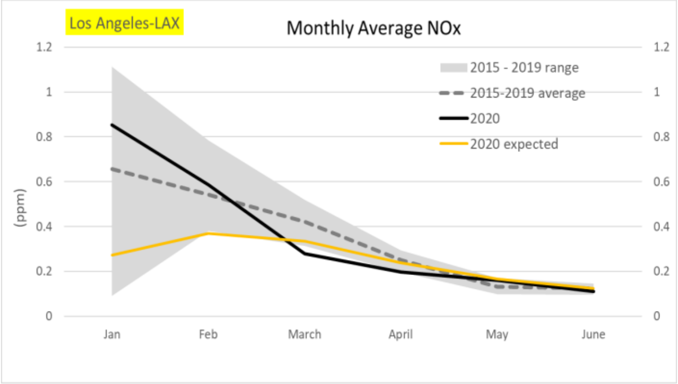
Exhibit F.1

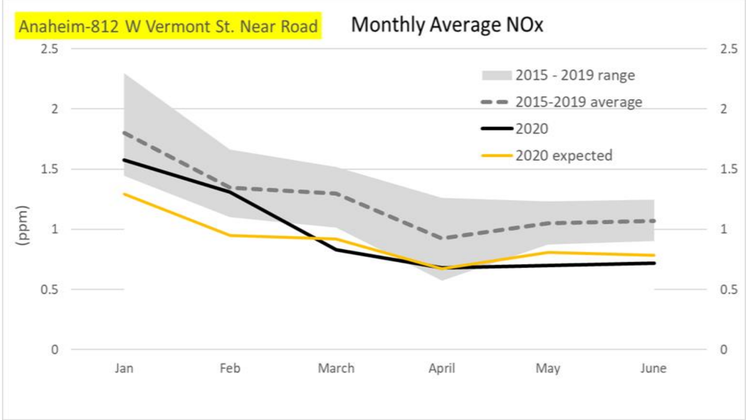
Top Down Emissions Characterization: NOx at Seven Sites

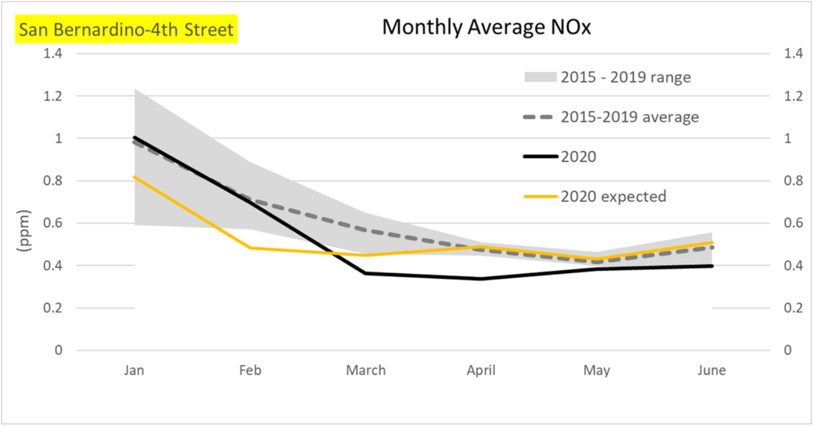


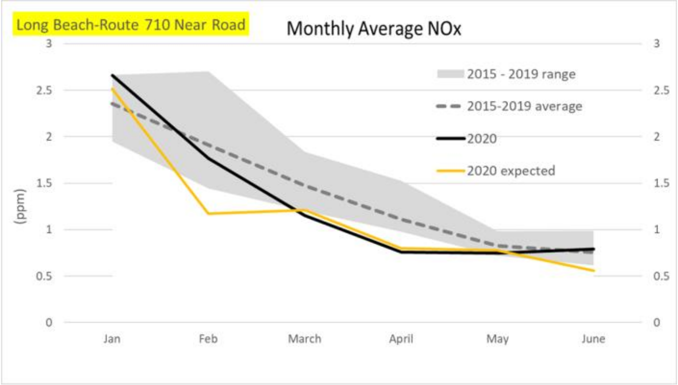


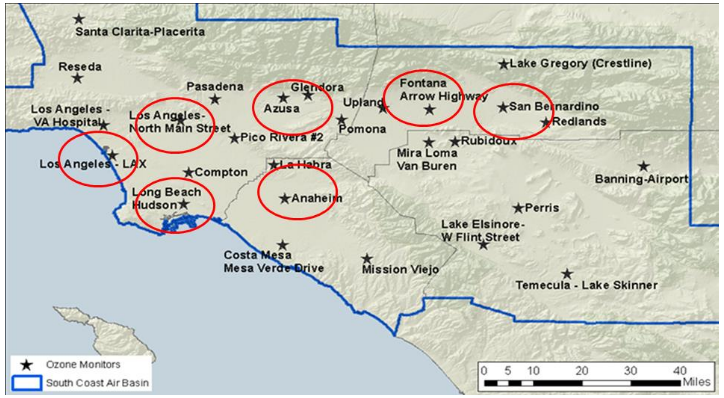








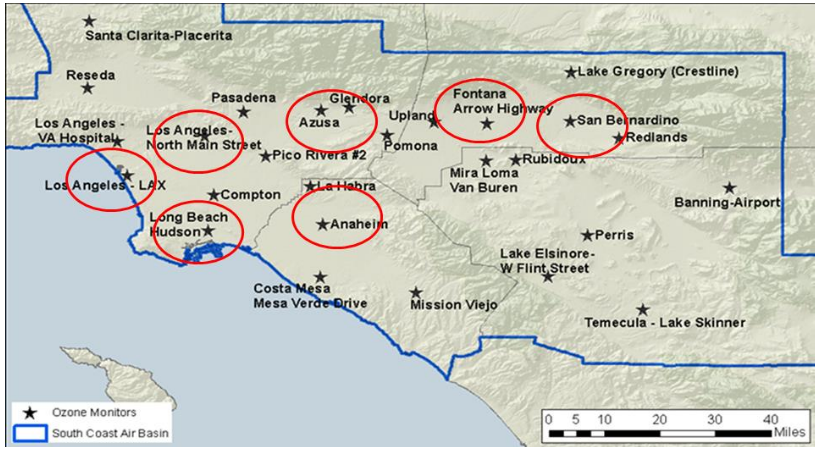




- 2020 Jan/Feb within 2015-2019 range
- 2020 March/April generally below range and below expected
- 2020 May/June below at interior sites

Exhibit F.1

Top Down Emissions Characterization: NOx Seven Sites Statistics



April-June difference from expected	
Los Angeles-LAX	-12%
Long Beach-Route 710 Near Road	8%
Los Angeles-North Main Street	-11%
Anaheim-812 W Vermont St. Near Road	-7%
Azusa	-19%
Fontana-Arrow Highway	-20%
San Bernardino-4th Street	-22%

- "2020 actual" compared against "2020 expected" April – June average show decreases at all sites except Long Beach
- "2020 actual" compared against 2015-2019 baseline both show substantial decreases since COVID-19 shelter in place orders in SoCAB during March and April and continuing in May and June at most sites

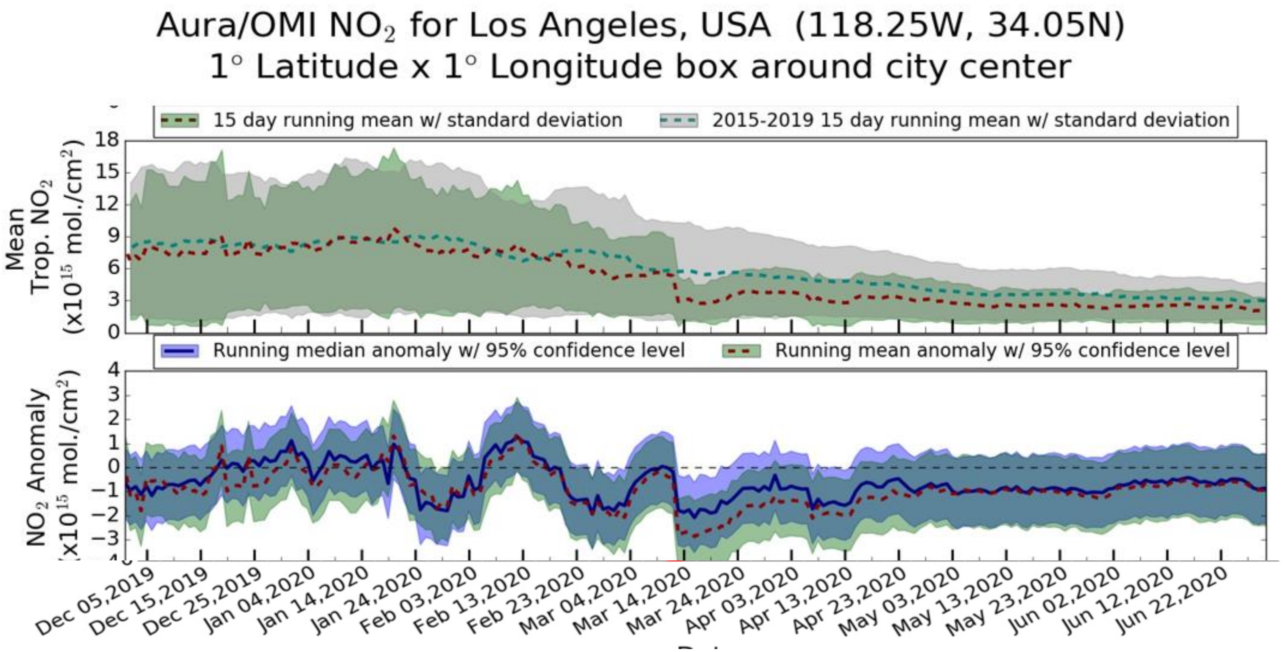
Reduction: (2020-base)/base						
	Jan	Feb	March	April	May	June
Los Angeles-LAX	30%	8%	-34%	-21%	22%	-10%
Long Beach-Route 710 Near Road	13%	-8%	-22%	-32%	-10%	5%
Los Angeles-North Main Street	-4%	-7%	-54%	-39%	-17%	-38%
Anaheim-812 W Vermont St. Near Road	-12%	-3%	-36%	-27%	-33%	-33%
Azusa	-1%	-4%	-43%	-38%	-12%	-40%
Fontana-Arrow Highway	18%	6%	-28%	-25%	1%	-22%
San Bernardino-4th Street	2%	-2%	-36%	-28%	-8%	-18%

Red = 15-30% increase
Grey = +/- 15%
Blue = 15-30% decrease
Green = 30+% decrease

Base = 2015-2019 average

Exhibit F.1

Top Down Emissions Characterization: Satellite NO_2 Columns



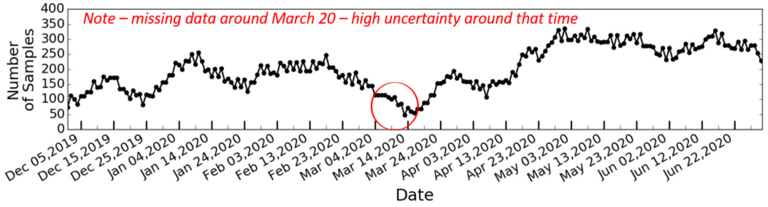
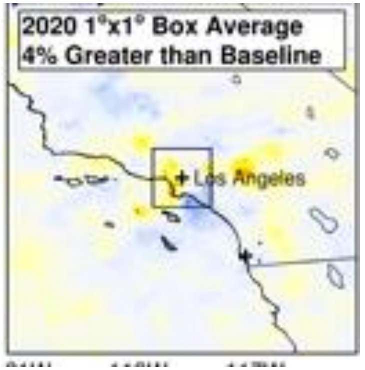


Image Credit: NASA



- NASA has generated plots for select U.S. cities, where they:
  - Compare 2020 to 2015-2019
  - Calculate a 15-day running mean to smooth out noisy satellite data
  - Compare 2020 mean and standard deviation (SD) to 2015-2019 mean and SD
  - Anomaly plot is the difference between 2015-2019 baseline and 2020
  - Zero anomaly line is dashed 0 line
- Consistently lower since March
- Does not separate COVID-19 response from longer term trends. (See next slide)

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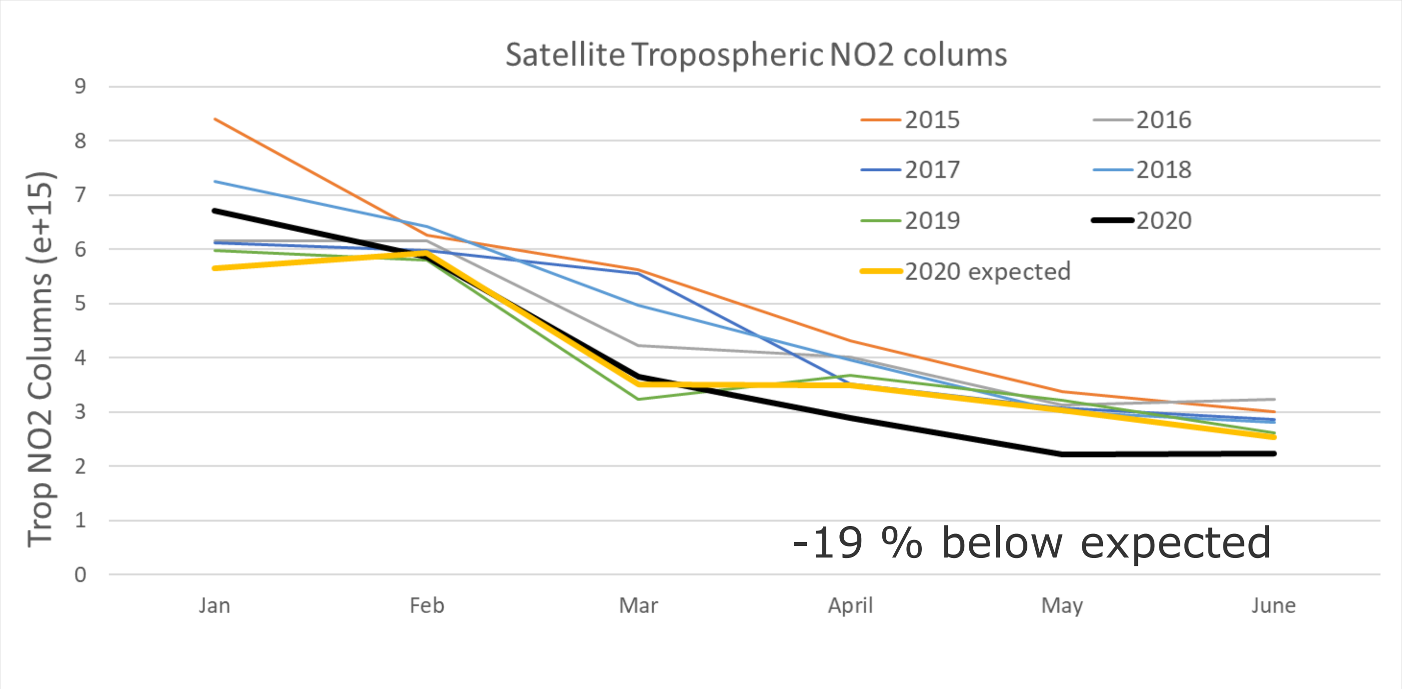
Top Down Emissions Characterization: Satellite NO2 Columns



- Download the NASA data for each year/day
- Average over months
- Perform similar “2020 expected” analysis
- Satellite estimated COVID-19 reduction in NO_2 adjusted for the 5-year trend over April – June is -19%

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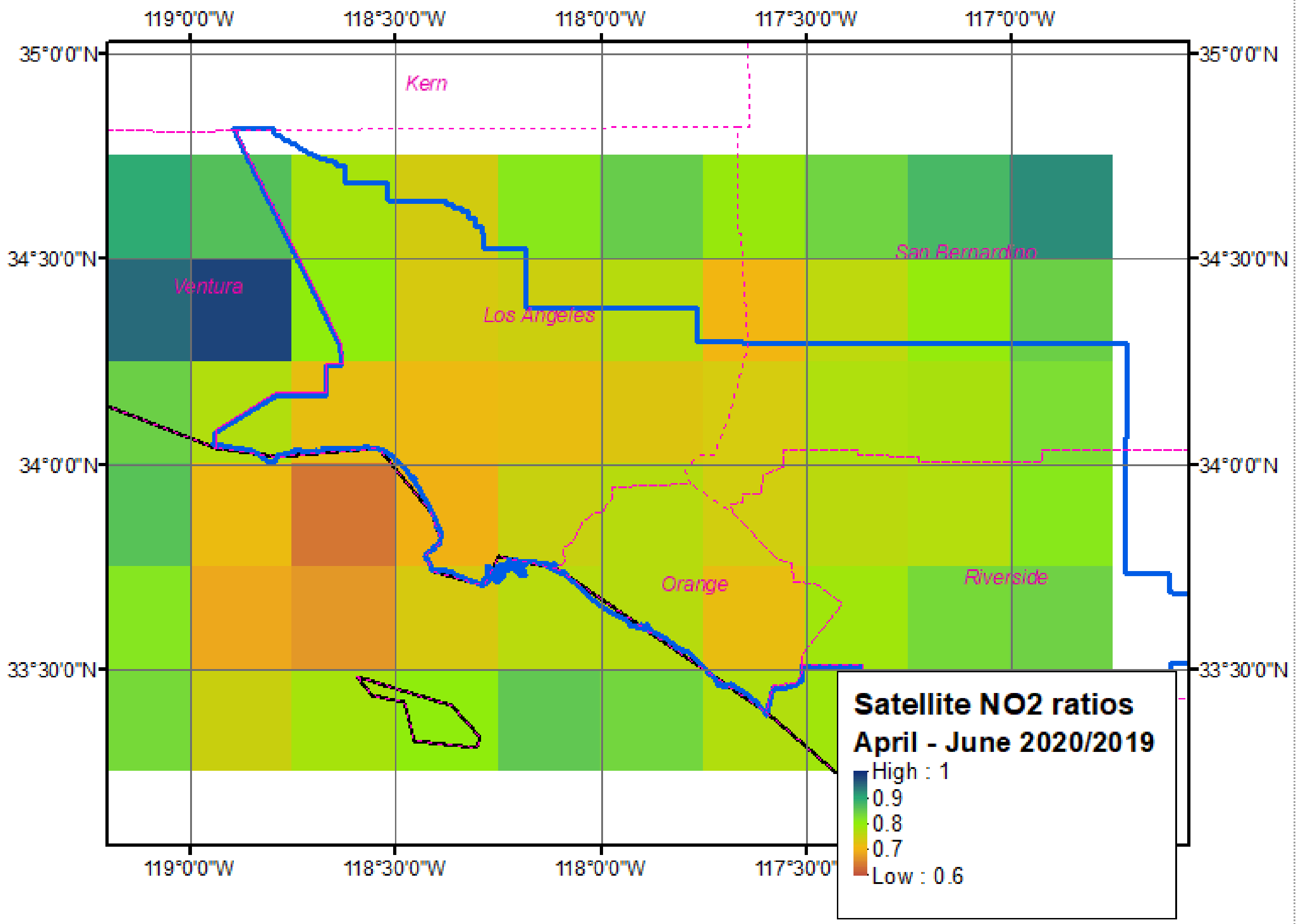
Exhibit F.1

Top Down Emissions Characterization: Satellite NO_2 Columns Spatial Variations

- Satellite data can potentially be used to inform NOx reductions that vary across the basin once the modeling period is determined
- Example plot shows OMI satellite data of ratio of 2020 over 2019 for tropospheric NO_2 columns
- 2020 is 70% - 80% of 2019 throughout much of the basin
- Considerable “devil in the details” when working with this data



https://giovanni.gsfc.nasa.gov/giovanni/#service=TmAvMp&starttime=2019-04-01T00:00:00Z&endtime=2019-06-30T23:59:59Z&bbox=-119.4562,33.1403,-116.6766,34.6674&data=OMNO2d_003_ColumnAmountNO2TropCloudScreened&variableFacets=dataProductPlatformInstrument%3AOMI%3B

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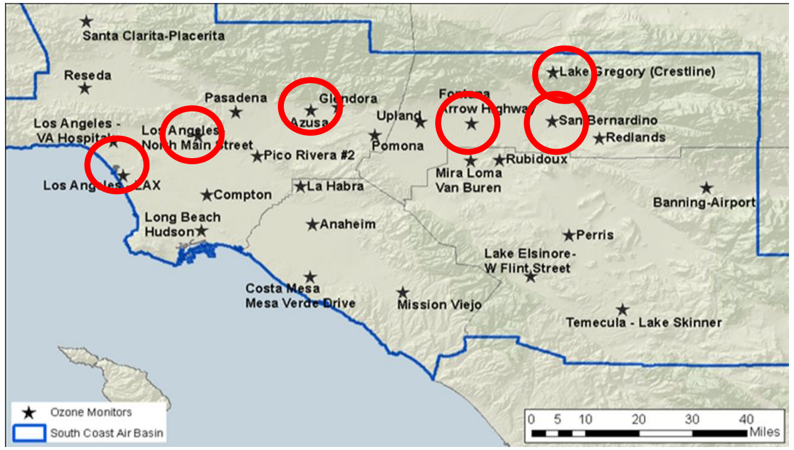
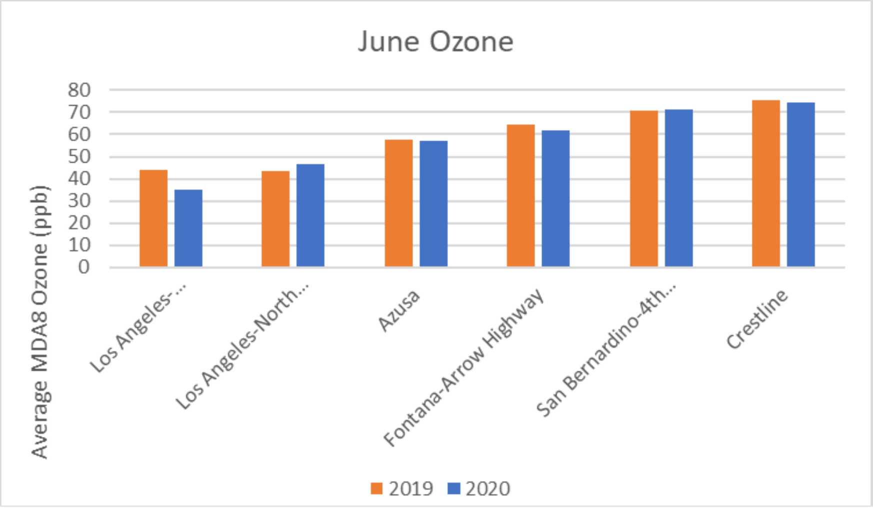
Top Down Emissions Characterization: Summary

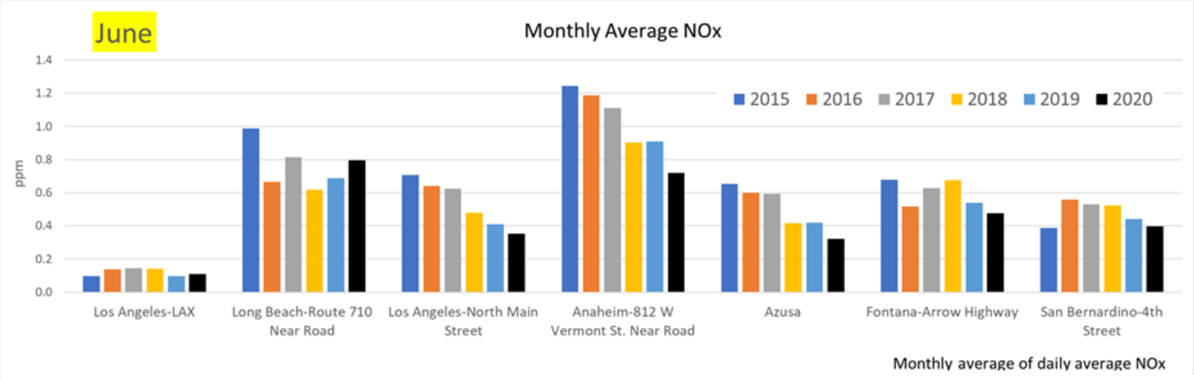
- NOx air concentrations measured by surface monitoring and tropospheric NO_2 columns detected by satellites both indicate a COVID-19 impact on NOx/NO_2 since March and continuing through the present after accounting for recent year declining trends
- Surface concentration NOx data and satellite NO2 data with temporal and spatial variations could be used to guide or validate NOx emissions estimates in the SoCAB due to the COVID-19 response

Exhibit F.1

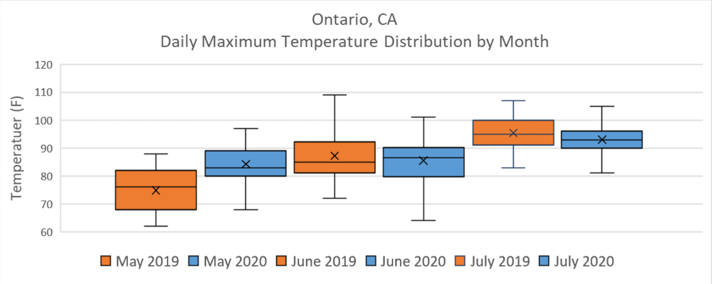
Observed Ozone Changes: Compare June 2019 and June 2020

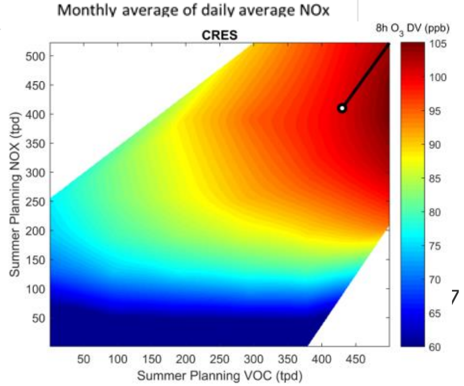






- Recall June 2019 and 2020 had comparable ozone conducive conditions. 2019 slightly more conducive on average
- NOx surface concentrations in 2020 < 2019 most locations
  - What about changes in VOCs?
- Ozone in non-coastal sites very similar in June 2019 and 2020 on average
  - What does this imply? Ozone unresponsive to change in emissions. Transitional regime, and/or following contour line for those sites





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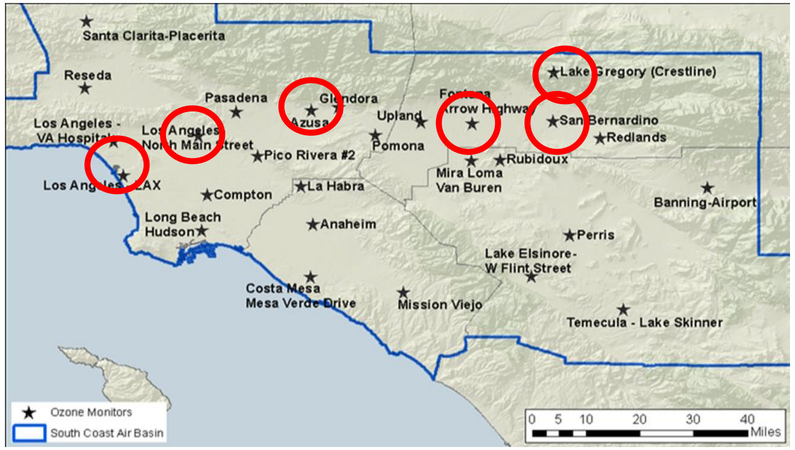
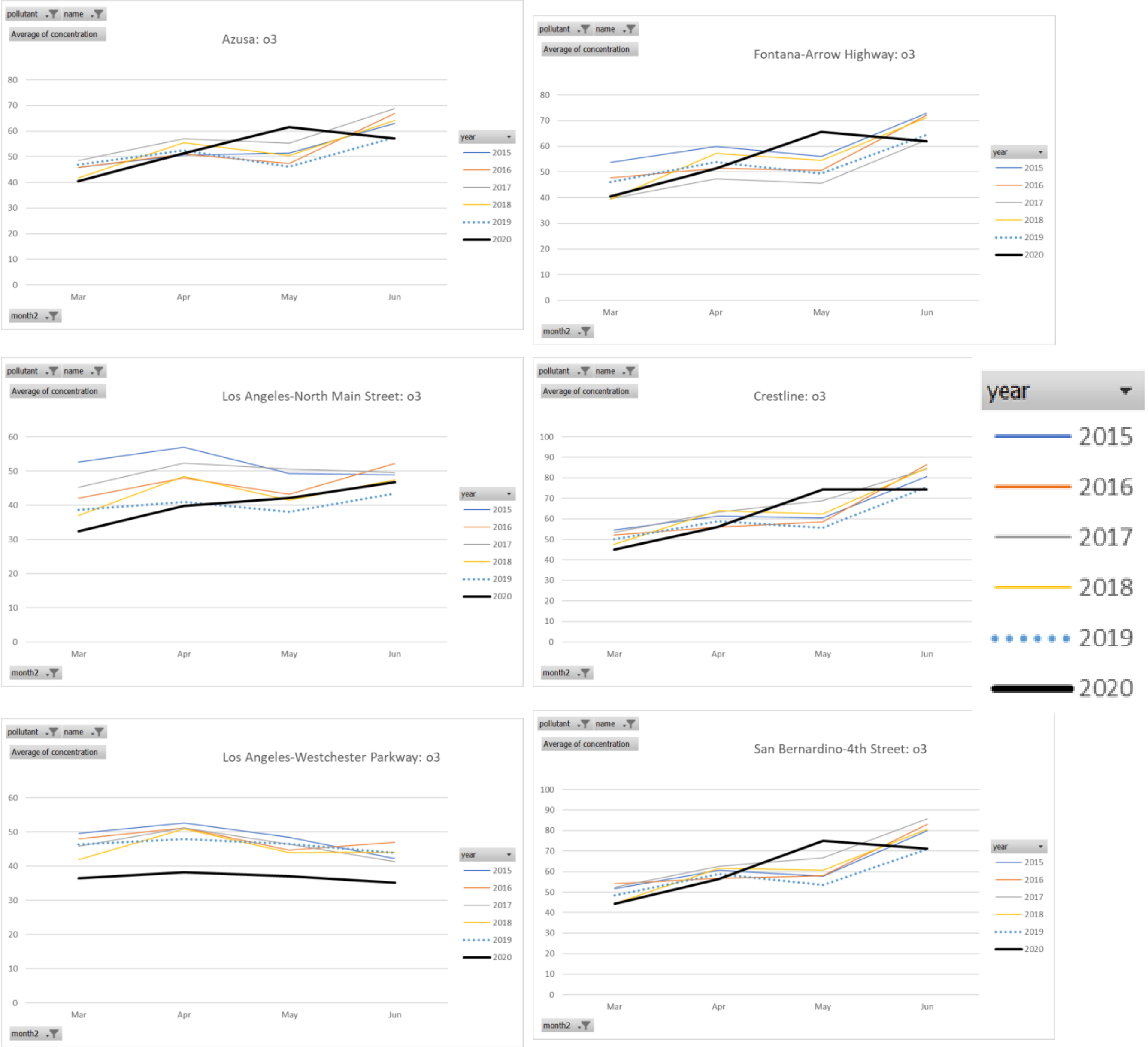
Observed Ozone Changes



- 2015 – 2020 March – June monthly averages of MDA8 ozone at 6 sites across the basin
    - March/April comparable to past 5 years (slightly lower)
    - Inland sites consistently higher O3 in May compare to past 5 years (likely due to Met; e.g., T850)
- June 2020 mostly similar to June 2019
- LAX substantially lower than last 5 years March – June, no traffic to airport? But NOx plot did not indicate that.

Exhibit F.1



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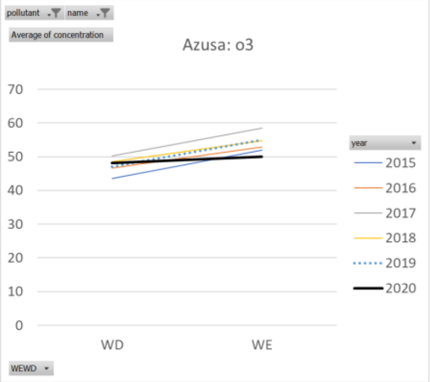
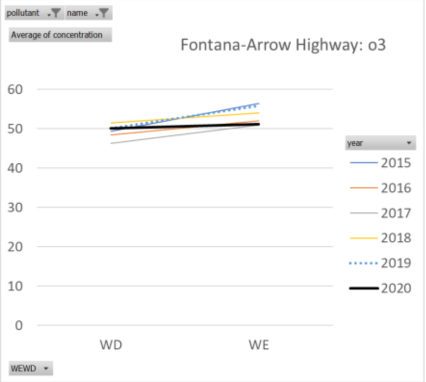
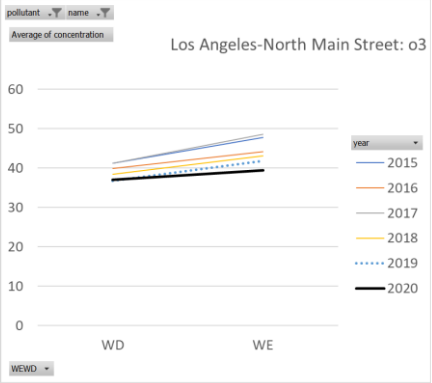
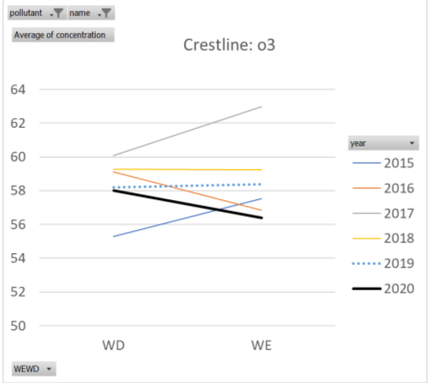
Observed Ozone Changes: Weekend Effect

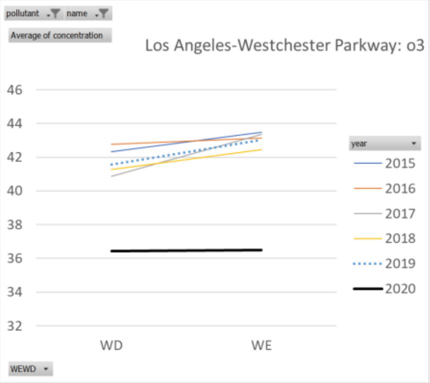
- The COVID-19 response (e.g. reduced commute driving) leads to a less pronounced difference in activity between weekends (WE) and weekdays (WD)
- Previous years indicate a weekend ozone effect (i.e. WD < WE) at all 6 sites for all years except for Crestline for some years (likely because it is transitioning from VOC to NOx-limited)
- Flat 2020 WE/WD ozone plots (i.e. no weekend effect in 2020) for sites with persistent weekend effect in prior years suggest 2020 weekend/weekday emissions are similar (or, lack the typical well-defined distinction)
- Further analysis of these plots is complicated since multiple confounding factors are relevant: (1) Total emissions reductions, (2) emissions reductions by WE/WD, (3) recent trends in emissions reductions over 2015-2019, (4) meteorological factors (e.g., 2017 ozone conducive year, 2020 ozone conducive May), (5) location of each site on the ozone isopleth plot. However, by reducing some parameters (e.g., restricting by meteorology to similar year/months such as omitting May, and focusing on 2019/2020 only) this type of analysis could be a key metric for evaluating the ability of air quality models to correctly simulate response of emissions reductions

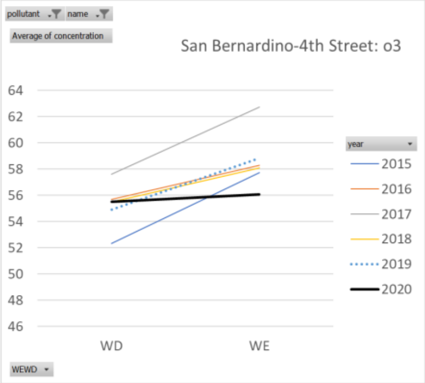


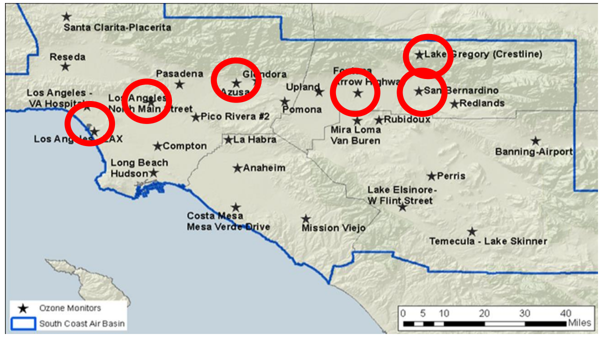












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March – June WE and WD averages

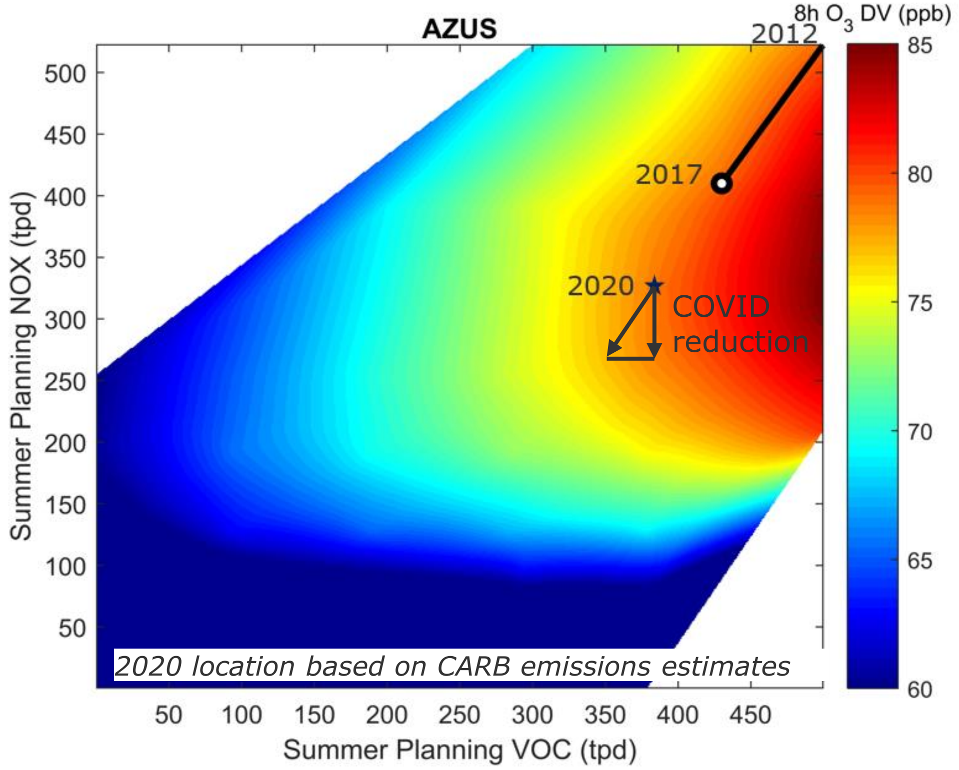
Observed Ozone Changes: Summary

- Some deviations from typical ozone have been observed since the COVID-19 shutdowns were implemented
- May 2020 had high ozone in SoCAB, which is likely attributable to meteorological factors
- June 2020 had met conditions and ozone very similar to June 2019, therefore was unresponsive to NOx reductions
- LAX had much lower ozone, the reason is not known. Note that NOx was not substantially reduced
- A lack of ozone weekend effect in 2020 suggests that weekday/weekend emission levels were similar in SoCAB
- A refined weekend/weekday effect analysis could be performed that could be a key metric in an air quality models evaluation

Exhibit F.1

Isopleth Discussion

1. Isopleths are for DVs based on peak ozone season days
2. Have not estimated VOC COVID-19 reductions
   - Expect the changes to be much less than NOx
   - If no change in VOC, then moving along the ridgeline of the ozone isopleth so no change in ozone with the 20% NOx reduction
3. Basin-wide NOx ~ 20% reduction from 2019 (satellite derived)
4. Close to transition region so would expect moderate to slight decrease in ozone, or no change, or slight increase depending on VOC reductions based on COVID-19 reductions at AZUSA location
5. More refined WE/WD analysis could potentially better inform location on the isopleth figure



Conclusions

- Surface measurements and satellite data indicate a COVID-19 impact on NOx/NO_2 in the SoCAB since March which continued through June (at a lower level) of about 10 – 30%, even after accounting for recent declining trends
- June 2019 and 2020 were meteorologically similar and therefore similar in terms of ozone formation conducive conditions
- June 2019 and June 2020 ozone levels were similar in the SoCAB at ozone design value monitors (i.e. Crestline, San Bernardino and Fontana locations)
- The analysis indicates that observed ozone levels were unresponsive to the reduction in NOx emissions (i.e. neither a NOx-disbenefit nor NOx-benefit was observed in June in the design value sites of the SoCAB)
- Do models respond in a similar fashion as the observed changes in ozone in response to the COVID NOX emissions reductions?

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AIR RESOURCES BOARD

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Initial Statement of Reasons	**)**	

COMMENTS OF THE
TRUCK AND ENGINE MANUFACTURERS ASSOCIATION

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STATE OF CALIFORNIA
AIR RESOURCES BOARD

Heavy-Duty Engine and Vehicle	**)**	
Omnibus Regulation and Associated	**)**	**Hearing Date:**
Amendments; Proposed Rulemaking;	**)**	**August 27, 2020**
Initial Statement of Reasons	**)**	

1. <u>Introduction</u>

The Truck and Engine Manufacturers Association ("EMA") hereby submits its comments in opposition to the proposed Heavy-Duty Engine and Vehicle Omnibus Regulations and Associated Amendments (the "Omnibus Regulations" or "Low-NO_x Regulations") that the California Air Resources Board ("CARB") has proposed to adopt at a Board hearing scheduled for August 27, 2020.

For several years, EMA and its members have actively attempted to engage with CARB staff regarding the development of CARB's rulemaking to achieve additional significant NO_x reductions from heavy-duty on-highway ("HDOH") commercial vehicles and engines. The scope of CARB's rulemaking effort, however, has expanded considerably over that time, and goes well beyond CARB's targeted 90% reduction in the tailpipe-NO_x HDOH engine-certification standard. The proposed "Omnibus Regulations" now also include a 50% reduction in the HDOH PM standard, a new "Low-Load" test-cycle and certification standard, a new in-use testing protocol with very significant modifications to the manufacturer-run in-use testing program, a new idle-NO_x standard and associated in-use test procedure, greatly extended "full useful life" and emissions warranty periods, a far more costly set of deterioration factor testing requirements (with multi-year impacts on product-development timelines), a "California-only" credit "averaging banking and trading" ("AB&T") program, and stricter recall and extended warranty liabilities associated with proposed "enhancements" to CARB's Emissions Warranty Information Reporting ("EWIR") program.

EMA and its members fully acknowledge the significant ozone air quality attainment problem that exists in the South Coast Air Basin, and we recognize why CARB is seeking additional HDOH NO_x emission reductions and regulatory improvements. To that end, our members have been and remain willing to develop and introduce cost-effective technology solutions to effect meaningful NO_x reductions. Indeed, EMA offered to voluntarily implement model year 2024-2026 product modifications to reduce NO_x emissions with expanded in-use compliance provisions *on a nationwide basis*. That program would have provided California with more NO_x reductions than it could achieve on its own, at a fraction of the cost of a California-only program, and could have set the stage for aligned national low-NO_x standards in 2027 and beyond. Regrettably, CARB has decided to press forward with its unique plans for an extremely aggressive California-only regulatory program for HDOH commercial vehicles and engines. As detailed below, that decision will lead to significant adverse results for all stakeholders and, more importantly, California's air quality.

The proposed Omnibus Regulations are cost-prohibitive, infeasible, unenforceable, and illegal, and, as confirmed by independent expert analyses, fall well short of any reasonable cost-benefit metrics. CARB has grossly underestimated the costs associated with nearly all aspects of the proposed far-reaching Omnibus Regulations, and has materially overestimated their potential benefits. CARB also has ignored the leadtime and stability provisions of the federal Clean Air Act ("CAA"), including through numerous requirements that would take effect with only two full years of leadtime – only half of the federally mandated leadtime period – and without regard to the timeframes required to fulfill the applicable federal preemption waiver requirements. That clear violation of the CAA renders the Omnibus Regulation ineligible for a federal preemption waiver, and therefore invalid and unenforceable. In addition, CARB's feasibility demonstrations are wholly inadequate, especially with respect to the 75% NO_x reductions targeted for model year ("MY") 2024 engines, where CARB's feasibility demonstration is basically non-existent.

Taken alone, the proposed Omnibus Regulations pose a serious threat to the California heavy-duty truck market. In that regard, the consequences of the Omnibus Regulations will go well beyond their significant "pre-buy/no-buy" impacts. There is the very real possibility, indeed, the likelihood, that several major HDOH diesel vehicle and engine producers will be compelled to exit the California market. Even if successful in developing potential technical solutions capable of meeting CARB's extremely stringent proposed low-NO_x standards and strict in-use compliance requirements — a premise which is far from certain given the minimal leadtimes provided — many manufacturers likely will determine that it would be impossible to recoup the tremendous research, development and production costs associated with the numerous onerous and unique provisions in CARB's proposed Low-NO_x Regulations.

Exacerbating that untenable situation is the fact that the Omnibus Regulations cannot be "taken alone." Contemporaneous with the Omnibus Low-NO_x Rule, CARB will be implementing its recently-adopted (and enhanced) Advanced Clean Truck ("ACT") Rule, pursuant to which CARB will be mandating the sale of increasing percentages of zero-emission battery-electric or hydrogen fuel-cell trucks, starting in 2024 and extending through 2035 and beyond. Accordingly, at the very same time that CARB will be mandating a cost-prohibitive reinvention of HDOH diesel trucks, CARB also will be mandating the increasing elimination of the market for HDOH diesel trucks. Obviously, that double-edged sword will devastate any sustainable market landscape for HD truck and engine manufacturers in California. And, to the extent that manufacturers are no longer able to sell HD diesel trucks in California, those same manufacturers will not be required to sell Zero Emission Vehicle ("ZEV") trucks in California, since CARB's ACT ZEV-truck sales mandates are formulated as a percentage of manufacturers' HDOH truck sales in California.

EMA's members have invested substantially in developing ZEV products, and we support efforts to help create a market for them. However, CARB's adoption of a ZEV sales mandate, without assurance of the requisite ZEV infrastructure and incentives, and with only a vague promise of future ZEV-purchase mandates in a separate second step, is not a pathway to success. That pathway becomes even more impossible given the diminishing likelihood of sufficient ZEV-purchase incentives and charging-infrastructure funding, which are critical to a robust rollout of

HDOH ZEV products, and which are at significant risk given the unprecedented continuing public health and economic crises.

The Omnibus Regulations and ACT Rule, separately and independently, will impose enormous costs and burdens on the HDOH engine and vehicle industry. They also will have significant adverse impacts on California's economy, and will cause fleet customers to keep their older trucks longer and defer buying new HDOH products. As a result, the projected environmental benefits of the Low-NO_x Regulations will be undermined and, likely, never achieved.

In addition, the accelerating COVID-19 pandemic will compound the inherent infeasibility of CARB's multiple regulatory proposals, and the resultant adverse impacts on the viability of the California heavy-duty truck market. As with every business sector, the COVID-19 crisis has had enormous and still-evolving impacts on EMA member-company operations and their research and development ("R&D") capabilities and activities. Mandated remote-work policies, coupled with reduced factory and test-cell operations, have seriously limited or completely halted manufacturers' capabilities to develop new low-NO_x products, both to meet current regulatory deadlines and to evaluate new regulatory programs such as those that CARB is proposing. In light of the cascading crisis, and given the fundamental uncertainty regarding how and when the situation will improve and return to some semblance of "normal," CARB must reconsider the scope and timing of the Omnibus Regulations.

In that regard, and contrary to one of CARB's core rationales, the scope and timing of the Omnibus Regulations will not serve as "a model of success" for U.S. EPA to follow, as CARB asserts in its Initial Statement of Reasons ("ISOR"). (See ISOR, p. ES-19.) First, EPA could never justify the costs of CARB's unique Omnibus program on a nationwide basis. (See NERA Report, discussed infra, which shows that any EPA rulemaking would face a per-truck cost cap less than $3,000.) Second, CARB's program is not calibrated for "success." Rather, CARB's Omnibus program is far more likely to lead to a significant pre-buy/no-buy response, and an absence of CARB-compliant new HDOH engines and vehicles in California, which will cause a number of material adverse consequences, including foregone opportunities for additional cost-effective reductions of NO_x emissions.

Consequently, the Board should not adopt the proposed Omnibus Regulations, but instead should direct CARB staff to work in good faith with all stakeholders to develop a cost-effective nationwide HDOH low-NO_x program to take effect in 2027.

2. Summary of EMA Comments

As detailed, below, the "Omnibus" package of new emission standards and test procedures that CARB has proposed for HDOH engines and vehicles is cost-prohibitive, infeasible, unenforceable, and illegal. One of the foreseeable consequences of the cost-prohibitive and infeasible Omnibus Regulations, especially when coupled with CARB's recently adopted ACT Rule, will be to drive HDOH engine and vehicle manufacturers out of the California market. And while CARB staff may not be focused on that outcome, those who need to perform work or move goods in the State surely will take note. Consequently, those businesses and laborers likely will be compelled to buy used vehicles (including out-of-state vehicles that have been driven for 7,500 miles so they are no longer "new" under CARB's "new motor vehicle" definition; see HSC

§43156), or to retain their current fleet vehicles longer than they otherwise would have. The result will be a significant slow-down in California's efforts to reach NAAQS attainment, and material adverse impacts to the California economy. Moreover, the pace of introducing heavy-duty ZEVs into California also will be slowed, since, as noted, the ACT regulations set their ZEV-truck sales mandates based on a percentage of manufacturers' new HDOH diesel truck sales in California starting in the 2024 model year (MY) — which sales will fall to near-zero given the cost-prohibitiveness of the Omnibus Regulations. As should be clear, 50% of very few (if any) HDOH diesel vehicle sales in California will equate to very few (if any) mandated HD ZEV sales under the ACT program.

All of this unnecessary regulatory turmoil could have been avoided if CARB staff had engaged in earnest with EMA over the last two years to develop a workable nationwide low-NO_x proposal – starting in 2024 and aligning with EPA regulations as of 2027 -- along the lines of the multiple proposals that EMA submitted for CARB's consideration in 2018 and 2019. CARB's failure to do so has led to this unfortunate cross-road of seemingly unsound public policy.

The Omnibus Regulations are **cost-prohibitive** because they will compel the development and installation of multiple expensive HDOH engine aftertreatment systems, along with significant engine hardware and software changes. Those direct costs will be added to the very high indirect costs that will result from CARB's Omnibus mandates for extended emissions warranties and full useful lives ("FULs"). Those elements of the Low-NO_x Regulations are likely to cause much higher warranty claims and recall liability, at least until the multiple new low-NO_x technology systems are proven-out over a multi-year period following the implementation of the 2024 and 2027 MY standards and compliance programs. Accordingly, and as confirmed by multiple independent expert analyses and reports (see infra), it is expected that the monetized costs of CARB's Omnibus program will exceed its monetized health-related benefits by at least a factor of eight (8). That type of inverse costs-to-benefits ratio renders the Omnibus Regulations cost-prohibitive (roughly, by an order of magnitude), and so invalid.

The Omnibus Regulations are **infeasible** because they require new engine technologies for the proposed 2024 MY standards that cannot be deployed in time and that will cause at least a 2% fuel-efficiency penalty, which is directly at odds with California's primary environmental objective to reduce greenhouse gas ("GHG") emissions as quickly as possible. The 2027 MY standards are infeasible because, among other things, CARB has not demonstrated the viability, durability, or packaging of the assumed low-NO_x technologies (e.g., cylinder deactivation and close-coupled multi-stream dual aftertreatment systems) in any prototype HDOH vehicle, and so has not even tried to demonstrate the feasibility of the new proposed "in-use" low-NO_x testing and compliance protocols. Moreover, the ultra-low level of CARB's proposed standards would leave no room for any compliance margin to account for emissions-testing variability or for expected deterioration over the proposed lengthened FULs of HDOH vehicles, which would extend out to 800,000 miles, or 12 years for low-annual-mileage vehicles, under the Omnibus Regulations. Infeasible regulations are invalid.

The in-use elements of CARB's Omnibus Regulations, including CARB's new three-bin "Moving Average Windows" ("3B-MAW")-based in-use standards, are **unenforceable** because those in-use standards are an order of magnitude below the emissions-detection capabilities of the latest portable emissions-measurement systems ("PEMS"), and an order-of-magnitude below the malfunction-detection capabilities of the latest on-board diagnostic ("OBD") sensors and systems. CARB's proposed Low-NO_x Regulations will rely on PEMS-based assessments and OBD sensor-based requirements to assure in-use compliance. But those systems cannot do so at the low-NO_x levels at issue.

Even in an emissions laboratory, PEMS NO_x-detection technologies (based on non-dispersive ultraviolet ("NDUV") detection methods for NO and NO2) have measurement "drift" that is roughly equivalent to the proposed in-use NO_x standard of 0.03 g/bhp-hr (starting in MY 2027), and have an error range of fully 50% down at the low-NO_x levels that CARB is proposing. And that is before the main sources of in-use PEMS-accuracy interferences are added in, which include imprecise exhaust and fuel-flow estimations, time-alignment issues, adverse ambient conditions and vibration, and PEMS-installation concerns. Indeed, UC-Riverside conducted a detailed study in 2016 which found that PEMS do not have the requisite level of accuracy as NO_x standards move below 0.10 g/kW-hr (0.075g/bhp-hr). That is why all of today's in-use NO_x standards include an in-use "measurement allowance" or compliance margin of at least 0.15 g/bhp-hr. While CARB's Omnibus Regulations would simply do away with that measurement allowance (without presenting <u>any</u> actual supporting data or technical justification), the underlying PEMS-measurement detection-limits, and accuracy and variability issues, remain the same.

Similarly, today's OBD NO_x-sensor-based capabilities are insufficiently precise to detect and "bin" in-use NO_x emission as CARB is proposing, or to assess in-use emissions compliance or potential emission-control malfunctions down at the low-NO_x levels that the Omnibus Regulations would mandate. To the contrary, the current OBD NO_x-malfunction threshold is no lower than 0.40 g/bhp-hr. Tellingly, CARB is proposing to retain, not lower, that OBD malfunction threshold under the new Low-NO_x Regulations, implicitly conceding that OBD NO_x sensors and related emission-detection systems are not accurate or robust enough to allow for the implementation of lower in-use OBD malfunction and enforcement thresholds. In fact, CARB expressly acknowledges that, "these higher OBD thresholds could allow emissions to exceed existing malfunction thresholds before detecting a fault, which could reduce the benefits of the proposed emission standards by allowing affected engines to operate without indication of the need for repair" (ISOR, III-10). In effect, then, CARB is proposing to maintain the NO_x-related OBD in-use compliance-assessment and enforcement criteria at a level that is an order of magnitude above the proposed applicable "3B-MAW"-based in-use NO_x emission standards.

The net result is that CARB is proposing in-use low-NO_x standards that cannot be accurately detected, measured or enforced through the PEMS and OBD systems that CARB is relying on as the tools of in-use compliance-assessment. In fact, given the current and near-term capabilities of PEMS and OBD systems, CARB is for all intents and purposes constrained to adopt in-use NO_x standards (be they "3B-MAW"-based or not) that reflect the measurement capabilities of the latest PEMS and OBD systems, which do not allow for in-use OBD NO_x-malfunction thresholds much below where they are now – 0.40 g/bhp-hr – and which still require the use of a PEMS-based NO_x measurement allowance of 0.15 g/bhp-hr. Adding that requisite measurement allowance to CARB's lowest proposed in-use NO_x standard yields a lowest feasible and enforceable in-use NO_x standard of 0.18 (0.03+0.15) g/bhp-hr, which still would need to be adjusted upward to match the current OBD NO_x threshold of 0.40 g/bhp-hr.

Since CARB is proposing in-use NO_x standards that are an order of magnitude lower than current in-use enforcement capabilities, CARB is proposing inherently unenforceable in-use NO_x standards, which renders the Omnibus Regulations invalid ab initio. Importantly, there is no disputing the unenforceability of CARB's in-use standards, about which CARB has said nothing. Emission standards that are an order of magnitude below their detectable and enforceable limit are necessarily unenforceable and invalid.

The Omnibus Regulations also are **illegal,** not only because they violate the requirements for adopting valid administrative regulations (including under the California Administrative Procedures Act), but also because they directly violate the controlling "leadtime" provisions of the federal Clean Air Act ("CAA"). CAA Section 202(a)(3)(c) mandates that new HDOH standards relating to the control of emissions cannot take effect unless the regulations afford four full-years of leadtime. CARB needs to demonstrate its compliance with CAA Section 202(a), including the four-year leadtime requirement, in order to obtain a waiver of federal preemption under CAA section 209(b). (See 42 U.S.C. § 7543(b)(1)(c).) Since the Omnibus Regulations are providing only two years of leadtime for all of the 2024 MY requirements, CARB's Regulations are violative of the controlling provisions of the CAA, are disqualified from receiving a waiver of federal preemption, and, as a result, are illegal.

EMA's comments will provide detailed data and analysis in support of each of the foregoing points, and will highlight multiple other unworkable, cost-prohibitive, and infeasible aspects of CARB's Omnibus proposal. In brief, the multiple points establishing the invalidity of the proposed Omnibus Regulations include the following:

(i) The Omnibus Regulations are cost-prohibitive, with costs exceeding monetized benefits by a factor of 8, as demonstrated through independent expert analyses prepared by ACT Research and NERA Economic Consulting. Cost-prohibitive rulemakings are invalid under California law, and cannot qualify for a federal preemption waiver under the federal CAA.

(ii) CARB is providing insufficient leadtime for the Omnibus Regulations, which is manifestly unreasonable, and which (again) will disqualify CARB from obtaining a federal CAA preemption waiver for the Omnibus Regulations.

(iii) The proposed low-NO_x emission standards and related requirements are inherently infeasible, especially since CARB is providing only two full-years of leadtime for the 2024-2026 MY standards and requirements.

(iv) CARB has failed to demonstrate the feasibility of the proposed 2024-2026 MY and 2027 MY and later low-NO_x emission standards and related requirements.

(v) The proposed Omnibus Regulations, coupled with the recently-adopted ACT Rule (and the 2023 new-truck purchase deadline under CARB's Truck and Bus Rule) will cause fleet operators in California to accelerate their purchases of new HD vehicles before the 2024 MY, and to refrain from purchasing new HD vehicles after the 2024 MY (a "pre-buy/no-buy" response), which will result in significant adverse air quality impacts in California, and which will significantly diminish the assumed benefits of the Omnibus Regulations.

(vi) The Omnibus Regulations likely will compel HDOH engine and vehicle manufacturers to exit the California market starting in advance of the 2024 MY, which will result in a lack of CARB-compliant HDOH products in the State, and material adverse impacts on California's economy.

(vii) Forcing HDOH diesel vehicles out of the California market will frustrate CARB's implementation of the ACT Rule, since the HD ZEV-sales mandates under that Rule are calculated as a percentage of new HD diesel truck sales, which will be significantly reduced, if not eliminated, due to the Omnibus Regulations.

(viii) CARB has not sufficiently assessed or validated the new proposed in-use low-NO_x standards and "3B-MAW" testing protocols under the Omnibus Regulations, which, in effect, amount to an untested and arbitrary approach for "binning" comingled sets of emission data that are not reasonably suited to the application of separate in-use low-NO_x emission standards.

(ix) CARB's proposed in-use low-NO_x standards are unenforceable, and so invalid, since in-use emission-measurement systems cannot detect or measure NO_x emissions down at the levels that CARB is mandating under the Omnibus Regulations.

(x) As confirmed even by CARB's own consultants (the National Renewable Energy Laboratory ("NREL")), CARB's Cost Assessment, Standardized Regulatory Impact Analysis, and Environmental Analyses are unreasonable and insufficient, and cannot meet CARB's administrative rulemaking requirements, including under the California Administrative Procedures Act, the California Government Code, and the California Environmental Quality Act.

(xi) There was a far more cost-effective nationwide alternative to the Omnibus Regulations, which is an additional factor establishing the unreasonable nature of the Omnibus Regulations.

(xii) The Omnibus Regulations will result in adverse fuel-efficiency penalties, which could threaten the feasibility and implementation of the HDOH Phase 2 GHG standards.

(xiii) The proposed extended warranty, FUL, and durability demonstration testing requirements, coupled with the increased strict-liability recall obligations imposed under the Omnibus Regulations, will result in unworkable regulations that, again, will drive HDOH engine and vehicle manufacturers out of the California market by 2024.

(xiv) The continuing COVID-19 pandemic and the related economic fallout are exacerbating the infeasibility and unreasonableness of the Omnibus Regulations, and warrant CARB's thorough reassessment of the multiple elements and timing of those proposed Regulations.

Cost-prohibitive, infeasible, unenforceable, and federally-preempted regulations do not reflect sound public policy, cannot be sustained, and should not be approved by the California Air Resources Board.

3. <u>The Proposed Omnibus Low-NO_x Regulations Are Cost-Prohibitive</u>

Independent experts at Americas Commercial Transportation Research Company ("ACT Research") have conducted a comprehensive cost study regarding the Omnibus Regulations, and have determined that they will result in an approximate $58,000 per-vehicle cost increase for heavy heavy-duty (HHD) vehicles sold in California as of the 2031 MY, using a 7% discount rate, and an approximate $51,000 per-vehicle cost increase for medium heavy-duty (MHD) vehicles.

On the benefits side, independent experts at NERA Economic Consulting ("NERA") have determined that the range of potential monetized health benefits from the implementation of the Omnibus Regulations in California, when focusing on potential reductions in secondary $PM_{2.5}$ as CARB is doing in its benefits analysis, could be as high as approximately $9,400 per-vehicle, or as low as approximately $3,800 per-vehicle (depending on which of the two most-cited epidemiologic studies is used to derive the operative risk factors for secondary $PM_{2.5}$). (Copies of the ACT Research and NERA Reports, which are discussed in greater detail below, are attached hereto as Exhibits "A" and "B".)

Using the averages of the foregoing numbers, on a per-vehicle basis, the Omnibus Regulations likely will have a costs-to-benefits ratio (or a negative benefits-to-costs ratio) of approximately 8:1, which makes those regulations cost-prohibitive. Cost-prohibitive regulations are invalid under California law, and cannot qualify for a preemption waiver under the operative provisions of the federal CAA.

a. CARB's ISOR and Appendices Significantly Understate the Costs of the Low-NO_x Regulations

CARB's Cost Assessment for the Omnibus Regulations (<u>see</u> ISOR, section IX) is understated by an order of magnitude. (<u>See</u> ISOR, pp. ES-15 and 16.) CARB's estimated average per-vehicle cost increase of $6,410 (including CARB's estimated HHD per-vehicle increased cost of $8,478, and its estimated MHD per-vehicle cost increase of $6,923) are not "all-in" costs, are unreasonably low, and are belied by the ACT Research study that EMA commissioned, as well as by the independent expert report that CARB commissioned from NREL. Moreover, the ACT ZEV-truck Rule — which impacts the same HDOH vehicles and manufacturers over the same time period (ISOR, p. I-36) — will exacerbate the per-vehicle cost increases at issue by reducing the HDOH diesel vehicle market in California year-over-year, thereby driving up the marginal cost of each CARB-compliant diesel vehicle as the market over which to allocate the increased Omnibus-compliance costs continues to shrink each year starting in 2024. CARB's Cost Assessment completely fails to account for that reality. All in, CARB estimates the total costs to manufacturers at $4.07 billion through 2050. As noted, that is based on an average per-vehicle cost increases of only $6,410 (Notice, pp. 13-14), which is understated by an order of magnitude.

Indicative of CARB's unreasonable lack of rigor in preparing its Cost Assesment is the fact that CARB has ignored all of the costs associated with the new proposed 50%-lower PM standard. On that issue, the only thing that CARB states is the following:

> NO_x and PM emissions in diesel engines are closely tied together, and calibration to optimize NOx emissions would also involve calibration to optimize PM emissions. CARB staff therefore assumes that the cost for reducing PM emissions would be absorbed by the engineering cost required to optimize NO_x emissions (included in Table IX-4) and that there would be no additional cost to meet the proposed PM standard. (ISOR, p. IX-15.)

While it is true that "NO_x and PM emissions in diesel engines are closely tied together," they are inversely so. Thus, manufacturers cannot simply "absorb" the cost of reducing PM emissions in their efforts to reduce NO_x as part of a "calibration" exercise. Indeed, reducing NO_x makes the effort to reduce PM all the more challenging and expensive. CARB's unsupported assumption to the contrary exemplifies the inherently deficient nature of its Cost Assessment.

In a similarly dismissive manner, CARB acknowledges the impacts that its previously-adopted Truck and Bus Rule (along with the ACT Rule) will have on shrinking the market for the purchase and sale of new HDOH diesel vehicles from and after the 2024 MY, but completely fails to account for that fact in its Cost Assessment. Specifically, CARB notes as follows: "Small business fleets throughout California will likely comply with the Truck and Bus Regulation via accelerated turnover (<u>i.e.</u>, by purchasing new trucks or newer used trucks). Because such business fleets would have just recently purchased trucks to comply with the Truck and Bus Regulation, they would not likely immediately purchase trucks with new 2024 or subsequent MY engines." (ISOR, p. IX-53.) CARB also recognizes that the Omnibus Regulations "could encourage California and out-of-state fleets operating in California to hold onto their existing vehicles longer or to consider purchasing used vehicles in-state or out-of-state in lieu of new vehicles in California." (ISOR, pp. IX-67 and 68.) Nonetheless, CARB makes <u>no</u> <u>efforts</u> whatsoever in its Cost Assessment (or in its benefits assessment) to quantify the likely impacts of the anticipated pre-buy/no-buy response to the Omnibus Regulations. That is a fundamental shortcoming of CARB's cost-benefit analysis.

CARB also never lists the sales volumes of new California-certified HDOH engines and vehicles that CARB is projecting will occur starting in the 2024 MY and continuing out year-over-year through 2050, the end date for CARB's cost projections. Perhaps that is because CARB realizes that any such projections are likely to be overstated due to the anticipated pre-buy/no-buy impacts of it Omnibus Regulations (coupled with the equivalent pre-buy impacts stemming from the 2023 vehicle-purchase deadline established under CARB's Truck and Bus Rule), and the progressively shrinking market for HDOH diesel engines and vehicles that simultaneously will result from the increasing HDOH ZEV-sales mandates under the recently-adopted ACT Rule. That rule will cut the HDOH diesel truck market roughly in half by 2032, if not sooner. Thus, CARB's omission of the HDOH sales projections on which it is relying in preparing its Cost Assessment is both telling and significant. On that point, all that CARB asserts is that its Cost Assessment is based on "the EMFAC future vehicle sales projections," without specifying what those projections are. (See ISOR, pp. IX-12, 13, 24, and 29.) That is not enough to make a sustainable record for this rulemaking.

Under the ACT Rulemaking, HHD and MHD vehicle manufacturers must convert a portion of their sales in California to ZEVs beginning in 2024, with increasing percentages though 2035. The following table sets forth the increasing percentages in the sales mandates for zero-emission trucks that CARB approved on June 25, 2020:

Table A-1. ZEV Sales Percentage Schedule

Model Year	Class 2b-3 Group	Class 4-8 Group	Class 7-8 Tractors Group
2024	5%	9%	5%
2025	7%	11%	7%
2026	10%	13%	10%
2027	15%	20%	15%
2028	20%	30%	20%
2029	25%	40%	25%
2030	30%	50%	30%
2031	35%	55%	35%
2032	40%	60%	40%
2033	45%	65%	40%
2034	50%	70%	40%
2035 and beyond	55%	75%	40%

CARB's contemporaneous mandate for increasing percentage sales of ZEV trucks will progressively shrink the market and sales volumes for low-NO_x diesel trucks built to comply with the Omnibus Regulations, which in turn will increase their marginal costs since there will be fewer trucks among which manufacturers' increased regulatory-compliance costs can be spread and allocated. That will further suppress the demand for new low-NO_x trucks in California, which will add further impetus to the regulatory forces (including pre-buy/no-buy impacts) that could limit HDOH vehicle offerings in California, or drive MHD and HHD vehicle manufacturers out of the California market altogether. In addition, CARB's overlapping HDOH regulations will more than double manufacturers' necessary R&D investments, which also will need to be spread over a smaller and smaller percentage of sales in each HDOH vehicle category, making it impractical if not impossible for manufacturers to recoup those multiplicative R&D investments. The overall results for the HDOH market in California will be untenable.

Another unreasonable aspect of CARB's Cost Assessment methodology is that it relies on the warranty claims rates, emissions-component failure rates, repair rates, and engine/aftertreatment-part recall rates that were associated with 2013 MY engines, and then "extrapolates those 2013 rates" to assess the likely defect, repair and recall rates anticipated for the envisioned and highly-complex 2024 MY and 2027 MY engine and aftertreatment systems, as represented by the low-NO_x prototype engines being developed at Southwest Research Institute ("SwRI") under its research contract with CARB. (See ISOR, pp. IX-19, 26, 28, and 32.) That is not a reasonable "extrapolation" methodology given the significant differences between 2013 MY engine and aftertreatment technologies, and the anticipated 2024/2027 MY engine and aftertreatment technologies. Using that same unreasonable 2013 benchmark, CARB also makes the blanket and wholly unsupported assumption that, extending out from the 2024 MY and beyond, fully 70% of all emission-related engine recalls will be addressed through a simple "software reflash" that will never cost more than $400, notwithstanding all of the new emissions-related engine and aftertreatment hardware that CARB's low-NO_x regulations will require. (See ISOR, pp. IX-27 and 32.) That cost assumption, like the others CARB has relied on, is simply not reasonable.

CARB concedes in its Cost Assessment that "the direct and indirect costs" of the Omnibus Regulations "would likely be passed on to engine/vehicle operators." (ISOR, p. IX-46.) CARB also notes that "the elements contributing to increased costs include establishing more stringent emission standards over existing regulatory cycles, amendments to in-use test procedures, modifications to the durability demonstration procedure for certification, lengthened warranty periods, lengthened useful life periods, amendments to EWIR reporting and corrective action procedures, and requiring NO_x data-collection and reporting." (Id.) Notwithstanding CARB's recognition of the anticipated aggregate impacts on the costs of new HDOH engines and vehicles in California, CARB fails to calculate or disclose the "all-in" estimated cost impacts of its Omnibus Proposal on a per-vehicle basis. That failure to provide any clear "all-in" per-vehicle cost metric – – coupled with CARB's failure even to specify the number of projected HDOH vehicle/engine sales that CARB is assuming will occur from and after the 2024 MY, which CARB is relying on in making its cost-benefit calculations — are additional fundamental shortcomings of CARB's Cost Assessment. CARB is unfairly masking the real-world impacts of its Omnibus Rulemaking.

CARB attempts to buttress its fundamentally unreasonable Cost Assessment by claiming that HDOH vehicle purchasers would "experience savings" resulting from the additional vehicle repairs that would be covered under the mandated lengthened emission warranties. (ISOR, p. V-11.) That claim is incorrect and completely undercut by CARB's admission that "the added costs associated with longer warranty periods would ultimately be passed on to consumers in the form of an increased purchase price for the trucks." (Id.)

Nonetheless, and "for simplicity," CARB just assumes that vehicle purchasers would start to realize repair savings "beginning in the sixth year of vehicle ownership," (ISOR, p. V-12), apparently because vehicle manufacturers uniformly would underestimate the real-world costs of CARB's lengthened warranties, and so would not include sufficient increases to the purchase prices of their new HDOH vehicles. CARB offers no evidence whatsoever in support of that assumption, which CARB admittedly made "for simplicity." And, of course, there is no such evidence that manufacturers will be unable to sufficiently cost-out the monetary impacts of CARB's extended warranties, and fully recapture those costs through increased purchase prices for new HDOH vehicles.

CARB similarly assumes that the longer mandated emissions warranties "will ensure that manufacturers, not vehicle owners, will pay for problems caused by poor design and durability [of emissions-related components] that CARB's HD I/M program detects," and that the extended warranties". . . would also protect heavy-duty vehicle owners from paying out-of-pocket expenses to replace emission-related components that are supposed to remain durable throughout the useful life of the engine." ". . . The lengthened warranties will shift some of those repair costs to the manufacturers." (ISOR, pp. ES-14, 11-17.) Again, those are manifestly incorrect and unjustified assumptions. Manufacturers will be highly motivated to ensure that all costs associated with the CARB-mandated extended warranties are thoroughly assessed and built-in to the initial purchase price of the new HDOH vehicles and engines that are covered by CARB's new extended mandates. Accordingly, the full "all-in" costs of those longer warranties almost certainly will be passed through to vehicle owners, not simply absorbed by manufacturers as CARB incorrectly assumes.

In spite of CARB's failure to address in a transparent manner the full per-vehicle cost increases that its Omnibus Regulations will cause, there is a way to begin to assess what CARB's assumptions reveal about that key cost-effectiveness metric. More specifically, CARB does provide per-vehicle cost impact estimates for two of the many elements of the Omnibus Rule – the per-vehicle "technology costs," and the per-vehicle extended warranty costs. (See ISOR, pp. IX-10 and 22.) For HHD vehicles, those total incremental costs through 2031 add up to $14,728 per-vehicle ($2,466 + $5,173 + $6,159 + $930). Significantly, that calculation still leaves out all per-vehicle costs associated with the Omnibus Program's new in-use testing requirements, new durability and useful life requirements, new EWIR and recall requirements, and new data-collection and reporting requirements. Thus, it is a very low and unrealistic per-vehicle cost value. Nonetheless, even though it is a fractional estimate of the aggregate "all-in" costs at issue, it is still a higher per-vehicle cost factor than the cost estimates CARB includes in the up-front sections of its ISOR. As noted, CARB's Notice of Hearing (at p. 14) and CARB's Executive Summary (at p. ES-16) posit a per-vehicle cost increase number for HHD vehicles of just $8,478. (See also P. IX-52.)

As explained more fully below, and as confirmed by the expert reports submitted by ACT Research and NREL (CARB's own cost-assessment contractor), CARB's aggregate cost estimates are flawed, inconsistent and significantly understated. Consequently, CARB's Cost Assessment is wholly insufficient to support the proposed major rulemaking.

b. The NREL Report that CARB Commissioned Confirms that CARB's Cost Assessment is Significantly Understated

Tellingly, in preparing its Cost Assessment, CARB staff have attempted to distance themselves from the very detailed cost assessment that CARB's retained expert consultant, NREL, developed and delivered to CARB for use in evaluating the cost-effectiveness of the Omnibus Regulations. In addition, CARB makes it more difficult than necessary to evaluate its cost-assessment methodology by highlighting increased per-vehicle costs for certain Omnibus Program elements, "Statewide" costs for others, aggregate manufacturer costs in still other instances, and purchaser costs in other cases. That amalgam of different cost metrics makes a comparison of CARB's methodology and results to those obtained by independent experts, including NREL, more challenging, but no less revealing.

In its ISOR (at p. IX-74), CARB states that while it did use certain of NREL's findings "to estimate costs associated with the technology packages needed to meet the Low-NO_x Regulations, CARB staff did not use NREL's survey responses related to lengthened warranties, which were very high, over $23,000 per-vehicle for the largest diesel trucks." CARB's efforts to discount NREL's findings (including through CARB staff's inconsistent application of "average useful life miles" and "average warranty miles"(see ISOR, p. IX-74)) are symptomatic of the understated Cost Assessment that CARB has constructed to try to support the Omnibus Rulemaking.

There is no reasonable basis for CARB's post hoc disavowal of its own designated experts. To the contrary, NREL's findings are generally consistent with the other expert report developed by ACT Research relating to the likely per-vehicle cost impacts of the Omnibus Proposals. That ACT Research report, like the NREL report, confirms that the anticipated lengthened warranty and full useful life costs are and will be the most important and largest factors in assessing the aggregate per-vehicle cost impacts of CARB's Low-NO_x Regulations. To ensure that CARB does not unfairly gloss over the importance and relevance of the NREL report, a copy of that report is attached as Exhibit "C" to EMA's comments.

The NREL cost study that CARB commissioned is very instructive. As an initial matter, it confirms that accurate projections about future HDOH vehicle production and sales volumes in response to the Omnibus Rulemaking are paramount considerations. It also clearly recognizes that, when attempting to assess indirect costs, such as the potential impacts of expanded warranty and EWIR requirements, OEMs are the entities best positioned to estimate those costs, which implicitly confirms that CARB's indirect cost-assessment method – "linearly extrapolating data from the 2013 model year" (ISOR, Append. C-3, p. 49) – is not a reasonable approach. NREL's conclusions on those points are as follows: "Engine OEM participation was crucial, as only they could provide estimates for indirect costs that represented a significant portion of the total cost. Incremental costs are largely driven by indirect costs associated with engineering research and development costs and warranty costs. Indirect costs are highly dependent on production volumes over which to amortize research and development costs. Indirect costs due to warranty are high, reflecting high uncertainty with new technologies and the introduction timeframes." (NREL Report, p. vii.)

The NREL Report is most telling, of course, in the bottom-line results it presents, results that are based on far more reasonable cost-estimation approaches than CARB's. Specifically, the NREL Report concludes that for HHD vehicles, the per-vehicle cost for compliance with CARB's Omnibus Regulations will range from $28,868 to $47,042, with the higher range being the more likely outcome. It is important to understand in assessing the likely invalidity of this Omnibus rulemaking that NRELs' high-range cost estimate is fully five and a-half times higher (550% higher) than CARB's HHD per-vehicle estimate of $8,478. It also is important to note that NREL's conclusions regarding the all-in per-vehicle costs of the Omnibus Regulations are much more in line with ACT Research's conclusions than with CARB's.

The higher-end range of NREL's per-vehicle cost estimates for HHD vehicles – a per-vehicle cost estimate of $49,318 (see NREL Report, p. 29, Table 18) -- is the more likely and realistic estimate given the high level of complexity associated with the SwRI "Stage 3" prototype engine and aftertreatment system that CARB is relying on for its assertions regarding technological feasibility. As detailed more fully below, that complex and expensive prototype system includes cylinder deactivation, five SCR catalyst beds, three ammonia-slip catalysts, two DEF-dosing systems, and multiple NO_x, NH_3 and temperature sensors. Both higher direct and indirect costs will be associated with that complex system, which makes the high-end range of NREL's cost assessment far more realistic. Even so, it is likely that NREL's higher-end per-vehicle cost number is still too low because NREL, like CARB, did not attempt to quantify the expected negative impacts that pre-buys and no-buys (what NREL calls "potential increased pre-purchases" (NREL

Report, p. 32)) will have on 2024 MY and later HDOH vehicle production volumes, which necessarily will drive up per-vehicle costs. Nor did NREL assess the additional reduction in production volumes that will result from the simultaneous implementation of the ACT Rule, or from the coinciding 2023 new-vehicle purchase deadline established under CARB's Truck and Bus Rule.

Consequently, while NREL's $49,318 per-vehicle cost assessment for HHD vehicles is far more reasonable and probable than CARB's significantly understated per-vehicle value of $8,478 (and more in line with ACT Research's findings and conclusions), the NREL cost values still do not capture the full adverse cost impacts that CARB's Omnibus Regulations would generate. That more complete assessment is reflected in the cost study that ACT Research has submitted regarding the Omnibus Low-NO_x regulations, as discussed below.

c. The ACT Research Study Demonstrates that CARB's Cost Estimates are an Order-of-Magnitude Too Low

EMA retained ACT Research to conduct a comprehensive assessment of the direct and indirect costs, as assessed on a per-vehicle basis, that likely will result from CARB's implementation of the Low-NO_x Regulations. The ACT Research Study (Exhibit "A" hereto) is based on a detailed survey and analysis of the HDOH vehicle and engine manufacturing industry, which ACT Research completed in the first quarter of 2020. "The study's focus is on the costs (including per-vehicle costs) that the truck and engine manufacturing industry likely will incur to comply with the proposed Omnibus Regulations." (ACT Report, p. 3.) Table 3 of the ACT Research report summarizes ACT's findings regarding the aggregated per-vehicle cost impacts of the Omnibus Regulations on a nationwide basis, and in California (highlighted in yellow):

Table 3: Cost Estimates to Meet Proposed Combined MY 2027 and MY2031 Vehicle Standards

	National				California			
	MY2027 + MY2031 from MY2018 base				MY2027 + MY2031 from MY2018 base			
Discount Rate	7%	3%	7%	3%	7%	3%	7%	3%
	MDD	MDD	HDD	HDD	MDD	MDD	HDD	HDD
per unit								
Total Direct Costs	$3,688	$5,002	$5,533	$7,540	$9,058	$12,286	$7,888	$10,732
Total Indirect Costs	$11,753	$16,765	$20,430	$29,540	$42,307	$59,591	$50,017	$70,089
Cost Increase per Unit ($)	**$15,441**	**$21,767**	**$25,963**	**$37,079**	**$51,365**	**$71,878**	**$57,905**	**$80,821**
$ in millions								
Total Direct Costs	$562	$762	$1,422	$1,938	$72	$98	$124	$169
Total Indirect Costs	$1,813	$2,590	$5,330	$7,718	$312	$435	$783	$1,096
Total Cost Increase ($M)	**$2,375**	**$3,352**	**$6,752**	**$9,656**	**$384**	**$533**	**$907**	**$1,265**

Source: ACT Research Co., LLC: Copyright 2020

Applying a 7% discount rate to the estimated costs, the per-vehicle costs in California will range from $57,905 for HHD vehicles to $51,365 for MHD vehicles, for an average per-HD-vehicle cost of $54,635. When that per-vehicle cost number is compared against CARB's ISOR-estimated per-vehicle cost number ($8,478 for HHD vehicles and $6,923 for MHD vehicles, for an average per-HD-vehicle cost of $7,700) it is clear that CARB has understated the per-vehicle costs of its Omnibus rulemaking by at least a factor of 7. That is even higher than the factor of 5.5 derived from a comparison of CARB's cost estimates with those that NREL derived. Either way, CARB has grossly mischaracterized the costs of this rulemaking.

Importantly, ACT Research also conducted a detailed analysis of the pre-buy/no-buy vehicle-purchasing practices that California-based HD vehicle fleet-operators will engage in to try to avoid the adverse cost and other impacts of the Omnibus Regulations. As set forth in Table 8 of the ACT Report, CARB's Omnibus Regulations will result in an initial two-year "pre-buy" equivalent to 39% of the California HHD vehicle market, followed by a second two-year pre-buy in advance of 2031 that will be equivalent to 14% of the HHD vehicle market — a total of approximately 133,000 "pre-bought" HHD vehicles. A pre-buy of that magnitude would eliminate a correspondingly large percentage of CARB's assumed emission-reduction benefits of the Omnibus Regulations, and would cause an approximate 36% (31% plus 5%) additional increase in the per-vehicle costs of the proposed Regulations. (See ACT Report, pp 20-21.)

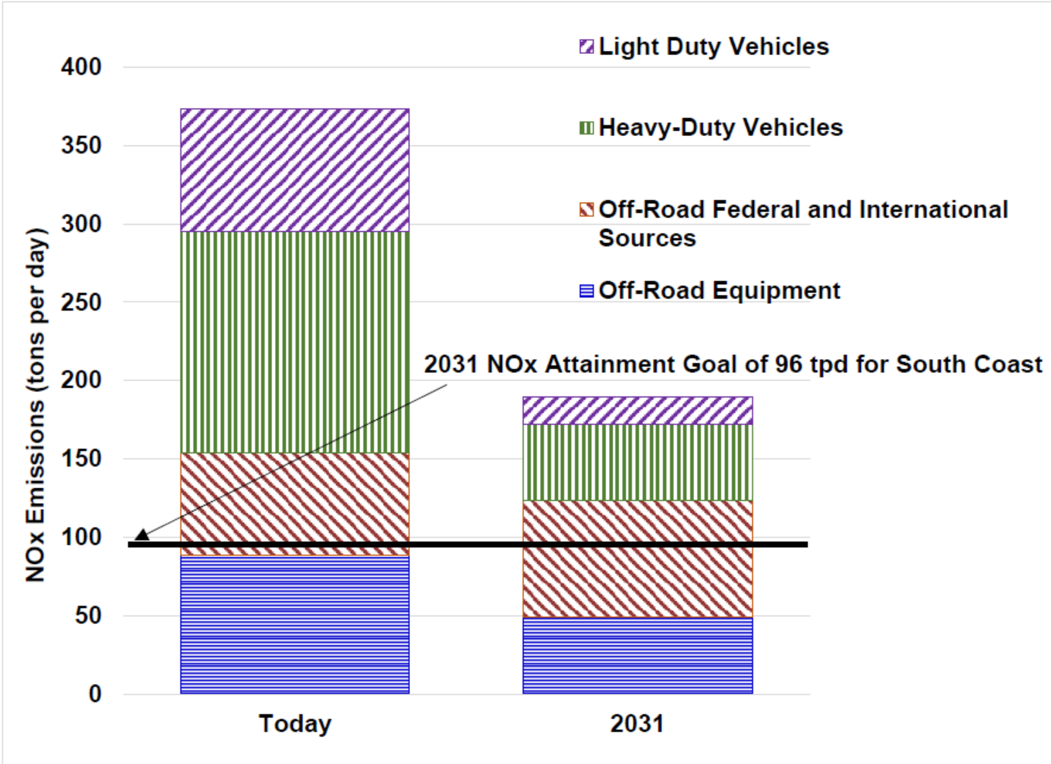
Thus, when factoring-in the likely pre-buy/no-buy impacts, it is clear that CARB has understated the likely per-vehicle cost impacts of its proposed Low-NO_x Regulations by nearly an order-of-magnitude.

d. CARB's Standardized Regulatory Impact Assessment ("SRIA") for the Proposed Omnibus Low-NO_x Regulations is Substantially Understated and Deficient as Support for a Rulemaking

Cost-effectiveness determinations also must take any corresponding benefits into account. In that regard, and as detailed below, CARB's Standardized Regulatory Impacts Analysis ("SRIA"), like the ISOR, also fails to account in a reasonable manner for the benefits (and costs) of the Proposed Omnibus Regulations.

As an initial matter, since the proposed low-NO_x standards and other Omnibus requirements will not take effect until the 2024 and 2027 model years, they will not help to avoid the upcoming ozone-nonattainment determination for the South Coast Air Basin (SoCAB) as of 2023 (additional ozone reductions of 108 tons-per-day (tpd) are still necessary by 2023 to reach attainment with the 80 ppb ozone standard in the SoCAB). Moreover, as depicted in the Figure below from CARB's ISOR, in order to meet the 75 ppb ozone standard in 2031, the SoCAB will require additional NO_x reductions of 154 tpd, since the NO_x "carrying capacity" in the SoCAB will drop from 141 tpd in 2023 to approximately 96 tpd in 2031. When that drop (45 tpd) is added to the 2023 shortfall of 108 tpd, the net result is that the SoCAB will need total additional NO_x reductions of 153 tpd as of 2031. By comparison, the ISOR asserts that the projected NO_x benefits from the Omnibus Regulations will be 23.2 tpd Statewide, and 7.0 tpd in the South Coast as of 2031. (Notice, p.12).

Figure ES-2. South Coast Mobile Source NOx Emissions With Existing Programs (CARB, 2018f)



Note: This figure is based on the 2016 State Strategy for the State Implementation Plan, which used CARB's EMFAC2014 inventory model to estimate on-road emissions. The more updated estimates in this document are based on EMFAC2017.

i. CARB's health benefits analysis is insufficient and unsupported

CARB's SRIA includes considerations of regulatory benefits and costs. On the benefits side, Appendix "E" to the ISOR contains the "details" of CARB's health benefits analysis of the NO_x reductions at issue. It is three pages long. Appendix "E" explains that CARB has opted to use a simplified (and unspecified) "incident-per-ton (IPT)" method to calculate avoided incidences of cardiopulmonary mortality due to exposure to secondary $PM_{2.5}$ (ammonium nitrate), and that the IPT method is premised on the core assumption (unproven) that "changes in [secondary $PM_{2.5}$] emissions are approximately proportional to changes in health outcomes," even at the current statewide ambient levels of $PM_{2.5}$. (ISOR, Append. E, p.2.) CARB also assumes that it is appropriate to utilize 95th-percentile confidence intervals.

CARB offers no support in the ISOR or the SRIA for those core assumptions. Nor does CARB specify, among other things: (i) the amount of assumed year-by-year reductions in secondary $PM_{2.5}$ that will result from the implementation of the Omnibus Regulations; (ii) whether those assumed year-by-year reductions in secondary $PM_{2.5}$ take into account the impacts of any pre-buy/no-buy response to the Omnibus Regulations, or the impacts of the Truck and Bus Rule and the ACT Rule; (iii) the specific epidemiological studies on which CARB is relying to calculate a concentration-response ("C-R") function or relative risk ("R-R") function for secondary $PM_{2.5}$, and why those specific studies were selected; (iv) the quantitative risk factors ("QRF") derived from the C-R or R-R functions, and how those QRFs were derived; (v) whether any adjustments were made to the QRF- or R-R-derived incidences of cardiopulmonary mortality to account for any differences in the $PM_{2.5}$ exposure levels experienced by the epi-study populations, on the one hand, and the prevailing and projected levels of ambient secondary $PM_{2.5}$ in California from and after 2024, on the other; and (vi) the range of uncertainties that relate to any derived QRFs or R-Rs, and to any derived mortality estimates, and how those uncertainties were accounted for. CARB provides none of those necessary assumptions and background information in the ISOR or in the SRIA, which makes it impossible to conduct any reasonable review of the validity of CARB's "simplified" health-benefits methods and calculations.

Notwithstanding those substantive omissions, CARB does concede that it did not take into account most of the key uncertainties that impact the scaling and quantification of health benefits (including the interpolation and estimation of exposures to secondary $PM_{2.5}$, socioeconomic status, and smoking rates), such that "the reported uncertainty ranges in [the reported] health impacts understate the true uncertainty." (ISOR, Append. "E," p. 2.) CARB's health benefits analysis is therefore inherently unsupported and suspect, and wholly insufficient to support this rulemaking, as further revealed by the comprehensive analysis that NERA has conducted regarding the likely range of quantified health benefits that could result from the type of HDOH low-NO_x regulations at issue.

ii. NERA's analysis clearly establishes the relevant health benefits benchmark for the Omnibus Rulemaking

NERA's expert report (Exhibit "B" hereto) estimates and quantifies the potential health benefits from the types of low-NO_x standards at issue, and includes two parts: a conceptual summary of methods and results; and a more detailed technical analysis. As explained in its conceptual summary, NERA conducted a comprehensive "scoping" analysis to estimate, on a per-vehicle basis, the likely maximum range of monetized health benefits that could result over time from the implementation of the envisioned low-NO_x standards. The relevant findings and conclusions from NERA's report as they relate to the monetized benefits potentially attributable to reductions in NO_x-related secondary $PM_{2.5}$ (the potential benefits CARB is relying on exclusively in its cursory health-benefits analysis) are described below.

NERA focused its benefits calculations on the value of projected health-risk reductions from the projected reductions in ambient ozone and secondary $PM_{2.5}$ that could result from reduced HDOH truck NO_x emissions due to the implementation of substantially tighter HDOH NO_x standards. Based on a long history of such benefits calculations (by EPA and many other entities), NERA assumed that approximately 98% of the estimated health benefits from reductions in ozone and $PM_{2.5}$ would be due to reductions in mortality risks. Thus, NERA focused its benefit-per-truck estimates by estimating only mortality risk benefits, having confidence that this method would have no meaningful impact on any quantified conclusions.

In order to obtain per-truck benefit estimates, NERA first calculated the tons of NO_x emissions reductions from an average new truck that would be purchased in 2027 meeting the tighter low-NO_x standard, accounting for a potential truck-life of up to 30 years. NERA made that calculation for each of the 8 truck types covered by the assumed low-NO_x standards. That computation was carried forward for each year of a truck's operational life. NERA also assessed the average truck's continued operation in each future year based on truck survival rates over time. The emissions reductions in each future year were then translated into a dollar estimate of each year's health benefits using a "reduced form" method in which the precursor emissions changes were multiplied by a "benefit per ton" value.

NERA's methodology generated a time-line from 2027 through 2057 of annual benefits per-truck in each year of the average 2027-vintage truck's operating life, varying across time (generally declining) as the truck ages. NERA discounted that stream of benefits to obtain the present value of benefits per-truck for each of the 8 truck types. Those 8 values were then combined into a single sales-weighted average benefit-per-truck estimate.

The most important input to NERA's benefit-per-ton estimates, and hence the benefit-per-truck estimates, is the assumption about the increase in mortality risk per unit change (reduction) in ozone and secondary $PM_{2.5}$ concentrations. That assumption is usually based on a statistically-derived association between mortality risk and observed pollutant concentrations or exposures, called a concentration-response (C-R) coefficient. The assumed C-R coefficient typically is derived from one or more of many existing epidemiological studies and associated peer-reviewed papers. EPA tends to change the mortality risk assumption as new epidemiology papers are published and as each NAAQS-review cycle is conducted. NERA reviewed statements in EPA's recent Policy Assessments for $PM_{2.5}$ and ozone (EPA, 2020 and 2019b) to attempt to anticipate which assumptions EPA might adopt in future Regulatory Impact Analyses ("RIAs"). Without commenting on the appropriateness of any such studies, NERA decided it would be reasonable to provide a range of estimates for the secondary $PM_{2.5}$ benefits-per-ton at issue. The lower end of the range is based on a C-R coefficient for all-cause mortality risk derived from the Krewski et al. (2009) study, and the higher end of the range is based on a C-R coefficient estimate for all-cause mortality risk from the Di et al. (2017) study.

There are significant scientific uncertainties when using statistical associations from epidemiological studies to predict risks for different populations and under different air quality concentrations and conditions in the future. At the same time, there are methods for identifying how the uncertainties may be reduced or scaled to derive benefits estimates that have a higher degree of confidence.

More specifically, any use of the derived unit risk estimate from an epidemiology study to predict changes in risks in different locations and under different levels of ambient pollution exposure necessarily involves extrapolation outside of the original range of the study's data. Extrapolation always introduces uncertainties that are not included in any of the original study's statistical measures of confidence. The more extreme is the extrapolation that a risk analysis requires with respect to exposure and population conditions not representative of the original study, the less qualitative confidence one would have in the derived risk estimate.

Such extrapolation can be a particular problem when using studies of associations between ambient air pollutant and health outcomes, even from the relatively recent past, to predict risk in a future year because of the steady declines in ambient pollutant concentrations that have taken place, especially with respect to $PM_{2.5}$, and that are projected to continue in the future. For example, the average concentrations of $PM_{2.5}$ experienced by the individuals studied in Krewski et al. (2009) fell by 30% during the period from 1980 to 2000, over which their mortality risk levels were being observed. Furthermore, the EPA dataset that NERA used to project average $PM_{2.5}$ levels in 2035 are another 50% lower (before any reductions due to a tightened HDOH low-NO_x standard) than the average exposures occurring at the end of the Krewski et al. study period (i.e., in 2000). Thus, the uncertainties due to extrapolation issues in this case are significant. Yet CARB did not take them into account at all.

It is possible to adjust the calculated risk estimates from the relevant epidemiology studies to exclude the portions of the estimates that involve the most extreme amounts of extrapolation from the exposure levels at issue in the original studies. As the amount of extrapolation from the original exposure and health-benefits estimates is reduced, confidence in the resulting estimate is qualitatively improved. This creates a "sliding-scale" of benefits estimates from least confident to most confident.

EPA introduced such a sliding confidence scale for its $PM_{2.5}$ co-benefits estimates in a recent RIA (EPA, 2019a), which employed a health risk estimate for all-cause mortality from the Krewski et al. (2009) epidemiology study. On that sliding scale, the "more confident" end of the spectrum of mortality risk estimates was calculated by excluding those portions of the underlying exposure and risk calculations that applied the original study's risk factor to $PM_{2.5}$ pollutant exposures below the 25th percentile of the originally-observed range of $PM_{2.5}$ exposures. The 25th percentile of a data set is generally viewed as the point where sparseness of exposure observations begins to undercut the ability to determine if an average C-R slope detected over the entire set of originally-observed exposure levels still remains at those lower and less frequently experienced exposure levels.

NERA applied that sliding-scale approach in the calculation of benefits that could be ascribed to the type of HDOH low-NO_x standards at issue. In doing so, by requiring more confidence in the benefit-per-truck estimates, the estimates declined somewhat, since they exclude benefits that are in areas with projected baseline $PM_{2.5}$ concentrations that are below various percentile levels of the pollutant observations in the original study (e.g., below the 25th percentile of exposures).

There is no way to select a single "best" cut-off point for limiting extrapolation uncertainties. In its last $PM_{2.5}$ NAAQS decision (i.e., the 2013 rulemaking), the EPA Administrator discussed how insufficient confidence in the continued existence of health risk associations would arise somewhere between the 10th to 25th percentiles of a study's range of observations. She chose to set the standard near the lowest of the 25th percentiles of available studies. NERA made an even more conservative choice in its analysis in this instance, and set its "best estimate" values at the 10th-percentile cut-off point of exposures from the underlying epidemiological studies.

In addition, in recognition of the significant differences in the projected $PM_{2.5}$ concentration distributions that exist between California and the rest of the country, NERA recomputed its benefits-per-truck for California (highlighted in yellow) and for the "Rest of the U.S.," separately. NERA's results, including the effects of the sliding-scale confidence-adjustments, are provided for $PM_{2.5}$ in Table 4 of NERA's Report, which is reprinted below:

Table 4: Range of $PM_{2.5}$ Benefit-Per-Truck Estimates (2019\$/truck) for California and Rest of U.S. Adjusted by Confidence Level Based on the Health Effect Estimates from the Krewski *et al*. (2009) and Di *et al*. (2017) Epidemiology Studies, Applying 3% and 7% Discount Rates

	No Adjustment	LML and Above	1st Percentile and Above	5th Percentile and Above	10th Percentile and Above	25th Percentile and Above
3% Discount Rate						
California	$9,390-$11,160	$9,050-$11,160	$8,530-$11,160	$6,300-$10,620	$3,760-$9,430	$1,600-$6,660
Rest of U.S.	$4,190-$5,080	$3,750-$5,080	$3,000-$5,080	$360-$4,180	$30-$2,620	$20-$210
National	$4,580-$5,540	$4,150-$5,540	$3,440-$5,540	$870-$4,680	$360-$3,180	$160-$780
7% Discount Rate						
California	$6,920-$8,180	$6,670-$8,180	$6,290-$8,180	$4,650-$7,780	$2,770-$6,910	$1,180-$4,880
Rest of U.S.	$3,140-$3,790	$2,810-$3,790	$2,250-$3,790	$270-$3,120	$20-$1,950	$10-$160
National	$3,430-$4,130	$3,110-$4,130	$2,570-$4,130	$650-$3,490	$270-$2,370	$120-$580

LML = Lowest Measured Level, meaning the minimum observed $PM_{2.5}$ concentration in the original epidemiological study

It should be noted that the benefits estimates NERA reports are conservative or, stated differently, weighted to the high side. That conservative approach stems from the fact that in conducting its analyses, NERA assumed, among other things, that: there is no exposure threshold to $PM_{2.5}$ or ozone below which mortality effects are no longer evident; the slope of the relative risk function for mortality is linear all the way down to zero exposure; and (as noted) it is appropriate to assess quantified benefits values at the 10th percentile of the exposure levels at issue in the underlying epidemiological studies, as opposed to utilizing a cut-point at the 25th percentile of exposures. Applying different assumptions regarding any of the foregoing points would lead to a reduction in the calculated benefits estimates. (NERA Report pp. 3-6, 9, 11, and 14-15.)

Based on NERA's confidence-adjusted analysis, and excluding only up to the 10th-percentile of the (unrepresentative) exposure data from the underlying epidemiology studies, and applying a 3% discount rate as opposed to a 7% discount rate, the per-truck benefits that could be derived from the types of HDOH low-NO_x regulations at issue range from approximately $9,400 on the high-side to $3,800 on the low-side, for an average per-truck benefit of $6,600. Comparing that average per-truck benefit against the average per-truck cost as determined by ACT Research ($54,500) yields a costs-to-benefits ratio (or a negative benefits-to-costs ratio) of approximately 8:1, which conclusively establishes that the Omnibus Regulations are cost-prohibitive and therefore invalid. There are no data in the rulemaking record sufficient to rebut that conclusion.

iii. The SRIA's cost estimates are understated and insufficient to justify adoption of the Omnibus Regulations

Turning back to the likely costs of the Omnibus Regulations, CARB's SRIA (like the ISOR) presents an incomplete and inaccurate analysis. As noted, the new lower-NO_x standards, new test cycles and new in-use requirements, coupled with the increases in FULs, warranty periods, and extended warranty and recall requirements, likely will lead OEMs to implement a series of significant cost pass-through actions to mitigate the significant regulatory-compliance obligations and risks. That is especially true given the multiple new technologies and aftertreatment control systems that must be developed to meet the near-zero NO_x levels at issue. Cost impacts for first owners, beyond the increased direct costs, also will include increases for longer warranties, extended warranty and recall protection, partial or full aftertreatment system replacement(s) during extended FULs, and additional inspection and maintenance of emission-related parts.

In the SRIA, which was prepared earlier in 2020, CARB bases its estimates of the likely engine "hardware" costs of its proposed Low-NO_x Regulations on a preliminary "literature review" that NREL conducted in February 2019. (SRIA, pp. 46-47.) That is an obviously inadequate and unreliable data source. As discussed above, NREL has conducted a far more thorough cost analysis, which was submitted to CARB in March of 2020. CARB should have used those updated (albeit still understated) NREL numbers and analyses to prepare a new and revised SRIA, but CARB has not done so, which (again) is inconsistent with CARB's administrative rulemaking obligations.

Not surprisingly, there are fundamental problems with CARB's cost analysis in the SRIA. Among them, CARB fails to account for the fuel penalties that will be associated with the proposed new low-NO_x standards in 2024, which likely will be at least 2%. Faced with those fuel penalties, manufacturers will be compelled by the current Phase 2 GHG regulations to install additional vehicle and/or engine technologies to make up that fuel-economy deficit, which will result in additional costs, complexity, weight, and potential performance impacts. Yet those costs are not considered <u>anywhere</u> in CARB's analysis.

CARB also fails (again) in its SRIA to account for the fact that truck fleet operators in California likely will engage in wide-scale "pre-buy/no-buy" strategies and will purchase out-of-state vehicles to avoid the substantial cost and product reliability impacts of the proposed regulations. Those likely alterations in vehicle-purchasing strategies will reduce significantly the already limited NO_x benefits that CARB has ascribed to the Omnibus Regulations. CARB has dismissed that possibility by assuming (wrongly) that per-vehicle costs will increase by only 2.5-6.0%, based on the NREL "literature review" (SRIA, pp. 33, 44). Specifically, in its SRIA, CARB assumes the following per-vehicle "direct cost" (engine hardware cost) increases based on the NREL literature review (<u>see</u> SRIA, p.47):

	2024 MY	**2027 MY**
HHD Vehicles	$1,625	$2,876
MHD Vehicles	$1,259	$1,625

In sharp contrast, and as previously noted, ACT Research conducted an actual comprehensive survey of all leading OEM's to assess the likely direct-cost impacts of CARB's Omnibus program, and determined that the following per-vehicle direct-cost impacts will result from CARB's proposal (as of 2027):

	2027 MY
HHD Vehicles	$7,738
MHD Vehicles	$9,056

CARB's HD vehicle direct-cost estimates in the SRIA are understated by a factor of ranging from 3 to 6.

When indirect costs are factored in, CARB's estimates in the SRIA are even more understated. That understatement results from the fact that CARB assumes (again incorrectly) that manufacturers will not fully adjust the costs of their HHD and MHD vehicles to recoup the full projected costs that will result from CARB's proposals to extend emission warranties and regulated FULs, and from the increased compliance liabilities that will stem from the amended warranty and defect reporting requirements (SRIA, pp. 36-37, 94). CARB's assumption is not reasonable. It is unreasonable to assume (as CARB also did in its ISOR) that manufacturers will choose to absorb the quantumly increased costs of the Omnibus Regulations. Based on consistent historical experience, and as a matter of sensible business practice, manufacturers will calculate and fully recoup those regulatory costs through corollary vehicle-price increases.

One specific example of the understated costs in CARB's SRIA can be found in CARB's discussion regarding the proposed extension of the FUL periods. While CARB has frequently stated that the longer FUL requirements will compel manufacturers to improve the durability of emissions-related components to meet the new requirement, CARB fails to consider any increase in cost from the design changes associated with those component enhancements. That is unreasonable. Even assuming just a 10% increase in component-part costs, when that percentage is applied to approximately $10,000 worth of existing components, the direct cost impact would be $1,000.

Not all components, however, will be capable of supporting the extended FUL requirements without a scheduled replacement within the FUL periods. That will almost certainly be true for some of the new "Stage 3" prototype components or systems deployed to comply with the dramatically lower NO_x standards and in-use requirements. CARB acknowledges as much in the proposed regulation by identifying six major emissions-related components that they intend to allow to be replaced under CARB's minimum maintenance provisions. Yet CARB does not assign any indirect cost assumptions to support any scheduled component replacements. Notwithstanding that omission, CARB's own data indicate that the replacement of just a single major emissions-related component costs on average $3,374 (see SRIA, p. 65, fn.76). Scheduled replacement of three systems within the extended FUL – not at all unlikely under a FUL requirement of 12 years and 800,000 miles – could easily amount to more than $10,000 in additional indirect costs. In that regard, a major OEM reports that the cost of parts and labor to replace the aftertreatment and NO_x sensors on today's HDOH products ranges from $14,200 to $18,100. Future aftertreatment systems developed to comply with the very stringent proposed low-NO_x standards will carry even greater costs. When considering the cost of improved designs to extend the life of many aftertreatment components, along with the replacement cost for other future aftertreatment-system components, it is clear that CARB's SRIA assumption of $309 for extended FUL costs falls well short of reality.

One additional example of the SRIA's significant understatement of costs relates to CARB's estimate of a per-vehicle R&D cost of $250. That is the scale of amortized R&D expense OEMs currently bear when developing 50-state products. When considering the high likelihood that any manufacturer choosing to develop a diesel product compliant with CARB's Omnibus Regulations would be selling that product only in California, the more accurate R&D cost estimate, amortized over California volumes, would be in the range of $23,000 to $26,000 per vehicle as of 2031, as confirmed in the ACT Research study.

CARB's overall estimate in its SRIA of the increases in per-vehicle costs that will result from its proposed Omnibus Regulations are as follows (see SRIA, pp. 47, 91):

	2025-2027	**2027-2031**	**Aggregate Total**	
HHD Vehicles				
Direct Costs	$1,625	$2,876		
Indirect Costs	$ 515	$3,352		
Total	$2,140	$6,228	$8,368	(increased to $8,478 in the ISOR)
MHD Vehicles				
Direct Costs	$1,259	$1,625		
Indirect Costs	$ 722	$5,119		
Total	$1,981	$6,744	$8,725	(decreased to $6,423 in the ISOR)

In contrast, ACT Research has calculated the following aggregate per-vehicle cost increases that will result in California from CARB's proposed Low-NO_x Regulations:

	2027	**2031**	**Aggregate Total**
HHD Vehicles			
Direct Costs	$7,738	$150	
Indirect Costs	$39,949	$10,068	
Total	$47,687	$10,218	$57,905
MHD Vehicles			
Direct Costs	$9,058	$0	
Indirect Costs	$32,416	$9,891	
Total	$41,474	$9,891	$51,365

ACT's detailed analyses demonstrate that CARB has underestimated the aggregated per-vehicle costs of its Omnibus Low-NO_x Regulations by a factor of 6 or 7 in the SRIA.

The net result is that the projected aggregate costs of CARB's Omnibus Low-NO_x Regulations will vastly exceed the reasonably projected aggregate benefits, rendering those regulations invalid under California law, and unenforceable because they will not qualify for the necessary federal preemption waiver under the CAA.[1]

[1] ACT Research has prepared a supplemental analysis of CARB's SRIA, and has confirmed that the SRIA fails to account for the full R&D, FUL, extended warranty, and pre-buy/no-buy cost impacts of the proposed Omnibus Regulations. A copy of ACT Research's supplemental analysis of CARB's SRIA is attached hereto as "Exhibit D."

iv. The proposed Omnibus Regulations will result in costs that far outstrip any monetized health benefits

Returning to the value of the benefits at issue, the first step in assessing aggregate benefits is estimating the total tons of NO_x (and secondary $PM_{2.5}$) that will be reduced due to the proposed regulations. CARB's estimates in that regard are inconsistent and incorrect. At page 34 of the SRIA, CARB states that its proposal will "reduce NO_x emissions by approximately 134,000 tons statewide between the years 2022 through 2040." The corresponding figure in the SRIA (Figure B-1) shows estimated NO_x reductions of approximately 50,000 tons between 2024 and 2040, a much lower figure. CARB also provides a third value in Table B-1 (SRIA, p.35), which indicates total NO_x reductions of 109.7 tons. Thus, it is unclear which estimate CARB thinks is correct, and even the most conservative projection in Figure B-1 (50,000 tons) is overstated as explained below.

Using one of its multiple estimates of tons-NO_x reductions, CARB calculates total monetized health-related benefits of approximately $3.15 billion as of 2032. (SRIA, p. 42.) CARB's monetized benefit calculations in its SRIA are both unclear and incorrect.

CARB's truncated health benefits analysis in its ISOR is similarly unfounded. In the ISOR, CARB states that its effort to develop quantitative estimates of potential health benefits is based ***exclusively*** on the benefits potentially attributable to the reductions in secondary $PM_{2.5}$ that could result from the implementation of the Low-NO_x Regulations. However, CARB's ISOR does not specifically quantify the expected reductions in ambient levels of $PM_{2.5}$ due to the implementation of the new low-NO_x standards. (See ISOR, Section VI.) Similarly, in Appendix "C," (at p. 86, n. 13), CARB reiterates that *all* of the monetized health benefits that it has calculated for this rulemaking are derived from its projected reductions in ambient secondary $PM_{2.5}$. But nowhere in the ISOR does CARB set forth or articulate what those year-by-year reductions in secondary $PM_{2.5}$ are expected to be starting in 2024. That critical omission, yet again, completely frustrates and undermines the notice and comment process for this rulemaking, which renders this rulemaking invalid on those grounds as well.

Notwithstanding CARB's failure to quantify the projected reductions in secondary $PM_{2.5}$ it is ascribing to the Low-NO_x Regulations, CARB posits $36.8 billion in aggregate monetized health benefits as of 2050, principally due to avoided incidences of premature mortality. (Notice, p. 22.) CARB's mortality estimates are substantially overstated (as detailed in NERA's report), especially given the reduced tons of NO_x that actually will be achieved due to the significant pre-buy/no-buy consequences at issue. In addition, CARB's utilization of 95^{th}-percentile epidemiological C-R values, its reliance on unspecified and likely ill-suited epidemiology studies, and its failure to include any uncertainty range all demonstrate that NERA's quantitative health benefit estimates are far more accurate.[2]

As detailed above, NERA has conducted a comprehensive benefits analysis of CARB's Omnibus Low-NO_x Regulations. The bottom-line results of NERA's analysis are that CARB's proposal will result in aggregate NO_x reductions in California of approximately 16,450 tons as of

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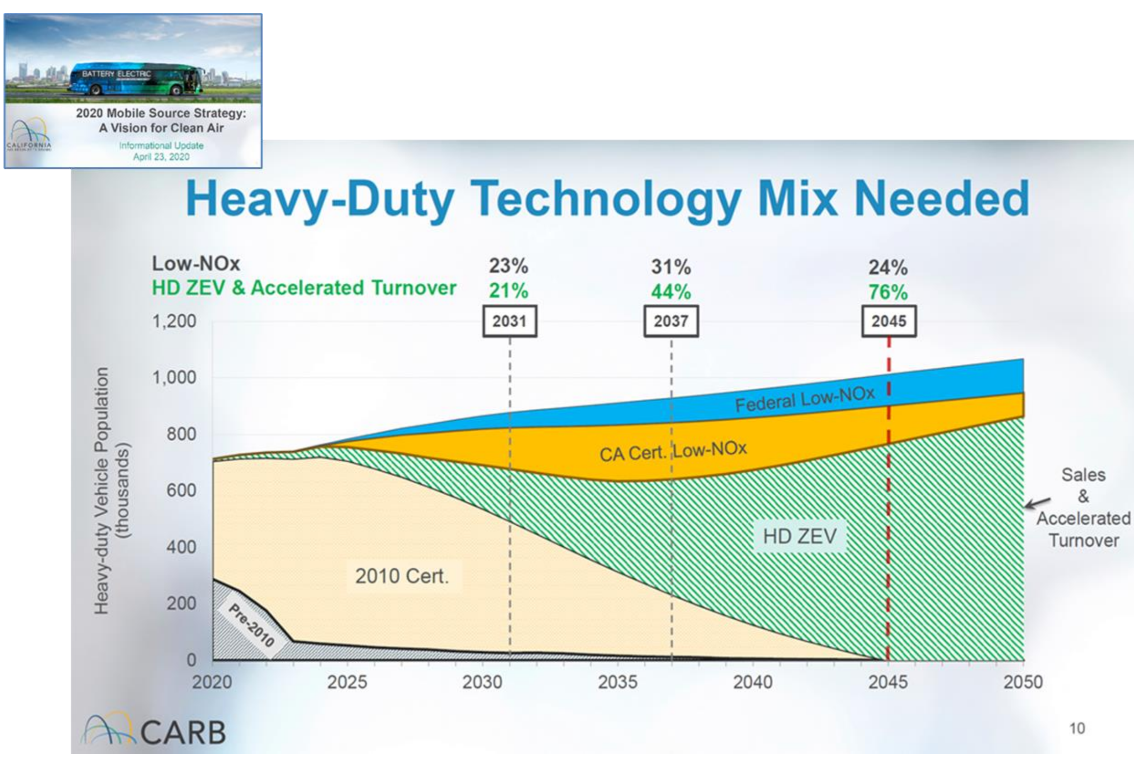
[2] CARB's health benefit calculations are internally inconsistent as well. For example, in the SRIA, CARB postulates 334 avoided incidences of premature mortality as of 2032. (SRIA, p. 41.) In the ISOR, CARB postulates 357 incidences of avoided premature mortality as of 2032. (ISOR, p. V-10.)

2032 (not 50,000 tons as CARB has projected for 2040), with a corresponding monetized health-related benefit (due to reduced secondary $PM_{2.5}$ impacts) of approximately $15,000 per ton (See NERA Report (Exhibit “B”)), “Technical Details of Analysis and Assumptions,” pp. 31, 33.) That yields an aggregate monetized health-related benefit of approximately $247 million, which is lower-than CARB’s aggregate benefits estimate as of 2032 ($3.15 billion) by a factor of more than 12.

On the other side of the benefits-to-costs ratio, ACT Research has estimated that the aggregate costs of CARB’s proposal (using per-vehicle costs and estimated new vehicle purchases in California, but without assessing any pre-buy/no-buy impacts) amount to approximately $907 million for HHD vehicles and $384 million for MHD vehicles, for a total cost of approximately $1.3 billion.

When ACT’s aggregate cost figure is compared to NERA’s aggregate per-ton benefits figure (again, without accounting for the likely pre-buy/no-buy impacts), the resultant cost-to-benefits ratio (or negative benefits-to-costs ratio) is approximately 4.5. Thus, by this per-ton metric, the likely aggregate costs of CARB’s proposal would exceed its potential aggregate benefits by at least a factor of 4.5. Using the per-vehicle metric discussed above, the more likely result is that the costs of the Omnibus Regulations will exceed their putative benefits by a factor of 8. (See pp. 9, 21, above.)

Importantly, and as discussed previously, these troubling upside-down cost-benefit results will be exacerbated by the impacts of the recently adopted ACT Rule, as evidenced by the following slide that CARB included in its April 23, 2020 presentation regarding the 2020 Mobile Source Strategy. That slide shows that CARB’s market-sales penetration forecast for HDOH diesel vehicles certified to the Omnibus Regulations is only 23% as of 2031, with much of the market displaced by the new mandated sales of ZEV trucks. Accordingly, the anticipated dynamics in the HD vehicle market in California over the next 10 years — given the expected impacts of the ACT Rule, the Truck and Bus Rule, and the significant pre-buy/no-buy response from fleets — effectively preordain that the costs of the Omnibus Regulations will far exceed any monetized benefits, as detailed above.



e. The Omnibus Regulations Will Cause Significant Pre-Buy/No-Buy Impacts

CARB's ISOR and SRIA do not account for the significant pre-buy/no-buy impacts that the Omnibus Regulations will cause. (Notice, pp. 17, 20-21; SRIA, pp. 33, 44.) That is a material omission.

The HD commercial vehicle truck market is very sensitive to the introduction of new technology-forcing emissions regulations. The most recent example of that is when EPA and CARB implemented a 90% reduction in the PM standard for 2007 MY and later heavy-duty engines, which required the introduction of diesel particulate filters into the HD marketplace. In parallel, NOx standards were reduced by 50%. HD vehicle purchasers, wary of the cost and reliability implications of the major new HDOH technology launches, significantly accelerated their vehicle-replacement purchasing cycles in 2005 and 2006 to avoid purchases of the new technology vehicles in 2007 – the classic manifestation of a pre-buy/no-buy response to new aggressive emissions regulation. More specifically, in the Class 8 market, vehicle purchases ramped up in 2005 and 2006, with the result that 40% more vehicles were sold in 2006 (284,000 units) than in 2004 (203,000). In 2007, the market dropped by a full 47%, to just 151,000 units. Among the other adverse consequence of that pre-buy/no-buy response, air quality benefits were delayed, and massive layoffs ensued at vehicle assembly plants and powertrain production sites, with similar cascading effects throughout the HDOH supply chain.

As already noted, in this case, given the significant per-vehicle cost differentials for CARB-compliant vehicles starting in the 2024 MY (see above), along with the likely negative fuel-efficiency impacts (and the associated increased costs for technologies to offset those fuel-consumption increases), and the various potential product reliability concerns that could arise among truck purchasers, fleet operators in California likely will pre-buy their new HDOH vehicles in advance of the Omnibus Regulations taking effect, and will refrain from buying new HDOH vehicles for multiple years thereafter. Indeed, CARB makes a critical admission in this regard, which largely undermines all of its cost-effectiveness assertions: "The Proposed Amendments could encourage California fleets to hold onto their existing vehicles longer, to purchase used vehicles in lieu of new vehicles in California, or to purchase more out-of-state vehicles. *Staff did not [even] attempt to quantify any such changes in fleet behavior*." (Notice, pp.17, 20.) (Emphasis added.) CARB goes on to assert that "it is not possible to quantify impacts on California's competitiveness" from the Omnibus Regulations, including "the likelihood of out-of-state and used truck purchases." (See also ISOR, pp. V-2 and 3.) (Notice, p. 21.)

Notwithstanding CARB's dismissal of this critical issue, it is possible to quantify those likely "fleet behavior" impacts, and EMA did so through its work with ACT Research. As noted above, ACT's quantification analysis shows that, at a minimum, there will be an initial pre-buy representing 39% of the market for new HHD vehicles in the two years before the 2027 MY standards take effect, followed by a secondary pre-buy representing approximately 14% of the market for new HHD vehicles in the two years before the purchase-price impacts of the extended warranty and useful life provisions take effect in the 2031 MY. (ACT Report, p. 16, Table 8.) And that is even before factoring in the additional pre-buys due to the coinciding Truck and Bus Rule vehicle-purchase deadline. CARB's failure even to attempt such a quantification establishes that its cost-effectiveness analysis, including as stated in its SRIA, is insufficient to serve as an adequate basis for this rulemaking.

The pre-buy/no-buy phenomenon in advance of the 2024 MY will be especially significant since 2023 is the deadline under the Truck and Bus Regulation for all HDOH vehicles to meet the 2010 emission standards. Thus, all California fleets will be motivated to buy trucks by 2023, leaving a dramatically reduced truck market in 2024. Consequently, the coincidence of the Truck and Bus Rule deadline and the first step of the Omnibus Regulations will have enormous impacts on the relative cost and efficacy of the proposed Regulations.

Just as significant, the anticipated pre-buys and corresponding no-buys will have correspondingly negative impacts on the already limited emission reductions that CARB is ascribing to the Omnibus Regulations (i.e., just 7.0 tpd NO_x in the South Coast as of 2031). Those negative impacts amount to an additional factor supporting the comparative cost-effectiveness of EMA's alternative proposal for a nationwide low-NO_x proposal starting in 2024, as discussed below.

4. EMA Proposed a More Cost-Effective Regulatory Alternative

EMA has proposed a more cost-effective alternative to CARB's cost-prohibitive Omnibus Regulations. Initially, in August of 2018, and in a substantially revised form in July of 2019, EMA submitted to CARB a detailed concept for an alternative nationwide low-NO_x rulemaking. While EMA's alternative concept would be less stringent than CARB's, it is inherently more effective because it would also cover the more than 60% of vehicle-miles-traveled (VMT) that are driven in California by out-of-state HHD trucks. (See ISOR, p. ES-17)

The substance of EMA's August 2018 alternative, which would have been implemented in 2024-2026 on a nationwide basis, not just in California, included a 25% lower NO_x standard, expanded in-use testing criteria also with a 25% lower in-use standard, and a commitment to work on a 2027 national lower-NO_x standard. EMA subsequently offered additional NO_x control measures in a July 11, 2019, submission to CARB, through the addition of CARB's Low Load Cycle and an even lower NO_x standard over the existing certification test cycles.

EMA's August 2018 nationwide alternative low-NO_x proposal assumed 5-plus years of leadtime to develop compliant 50-state MHD and HHD products. Since that time, with no movement from CARB toward agreement on the pull-ahead of a nationwide alternative, EMA members have lost over one-and-a-half years of development time, making the commitment to voluntary nationwide standards at that level by 2024 likely impossible at this stage, and making it questionable whether even less aggressive reductions could be implemented nationwide in advance of the 2024 MY.

While CARB claims that its proposed Low-NO_x Regulations could result in total NO_x reductions of 28,617 tons as of 2032, and that EMA's nationwide alternative would result in 21,056 tons (SRIA, p. 129) – which is a difference of 7,561 tons or 26% – that is not correct. Independent air-quality-modeling experts from Ramboll Group ("Ramboll") have compared the state-wide benefits of EMA's alternative nationwide program with the potential benefits under CARB's California-only program, and determined that EMA's alternative would yield more than 90% of the estimated NO_x reductions under CARB's proposed regulations through 2035. (A copy of Ramboll's Report is attached hereto as Exhibit "E.") Moreover, even CARB agrees that EMA's nationwide alternative would be far more cost-effective than CARB's California-only proposal.

The cost-effectiveness metric for EMA's alternative, as assessed by CARB, is $8,644 per-ton of NO_x ($182 million cost divided by 21,056 tons), while CARB's assessment (albeit understated) of the costs related to its Omnibus Regulations is $37,495 per-ton of NO_x ($1.073 billion cost divided by 28,617 tons). (SRIA, pp. 126, 129.) Thus, EMA's alternative nationwide program, even as assessed by CARB, is more than four times more cost-effective than CARB's.

In its ISOR, CARB notes that the nationwide 50-state alternative low-NO_x program that EMA proposed to CARB in 2018 and 2019 would cost "$3.59 billion less than the Proposed Amendments, about 80% less," while yielding 92.2% of the public health benefits that CARB has ascribed to the Proposed Amendments, an analysis that is in agreement with Ramboll's. (ISOR, pp. X-12 and X-14.) Thus, CARB admits that EMA's proposal would have been far more cost-effective than what CARB is now presenting for Board approval. "The total cost-effectiveness of Alternative 2 [EMA's nationwide proposal] is modeled to be $1.38 per pound of NO_x reduced, significantly less than the Proposed Amendments." (ISOR, p. X-16.) "Alternative 2 would be more cost-effective than the Proposed Amendments." (See Response to DOF, p. 17.)

Consequently, a clearly more reasonable and cost-effective regulatory alternative was available in this case, which renders the Omnibus Proposal inherently unreasonable and invalid.

5. CARB's Environmental Analysis Does Not Meet CARB's CEQA Obligations

CARB's Environmental Analysis ("EA") (ISOR, Section VII) is fundamentally deficient as well, and fails to satisfy CARB's obligations under the California Environmental Quality Act (CEQA). In submitting the proposed Omnibus Regulations for adoption, CARB is attempting to rely on the EA that was prepared several years ago in connection with CARB's 2016 State SIP Strategy document, which included a preliminary analysis of just two of CARB's proposed Omnibus Regulations. (ISOR, p. VII-1.) That is wholly inadequate in this case, and will result in an invalid rulemaking.

In support of not preparing an actual EA for this rulemaking, CARB states that, "Staff has determined that no additional environmental review is required for the current Proposed Amendments because there are no changes proposed to the originally approved project that involve significant environmental effects or a substantial increase in severity of previously identified significant effects." (Notice p. 25.) Staff's determination in that regard is plainly wrong. The full suite of proposed "Omnibus" regulations has changed and expanded significantly since 2016. The 3B-MAW proposal and multiple in-use standards is new. The phased NO_x standards are new, as is the lower PM standard. The extended warranty and FUL periods are new. The LLC standard is new. The durability requirements are new. And the EWIR changes and associated strict liability provisions are new. Moreover, CARB admits that it has done nothing to assess the significant pre-buy/no-buy ramifications that will certainly result from its "Omnibus" requirements as of the 2024 MY. And CARB has not done anything to address the increasingly relevant NO_x-disbenefit phenomenon (see infra). Nor has CARB conducted any assessment of the high likelihood that CARB's regulations will result in an absence of compliant new HDOH vehicles and engines in California starting in 2024, which will almost certainly have adverse impacts on California's air quality and economy going forward. Thus, CARB's unilateral determination that it need not prepare any updated EA for this "Omnibus" rulemaking is wrong.[3]

First, the levels of the low-NO_x standards currently at issue are different from, and are phased-in differently than, the low-NO_x standards originally assumed and assessed in the 2016 SIP Strategy. Second, the currently proposed 3B-MAW-based in-use protocols and standards are entirely different as well, since, the 3B-MAW method, with its three separately binned in-use standards, was not even contemplated let alone evaluated when the EA for the 2016 SIP strategy was prepared. Third, the prior EA did not (and could not) adequately assess the environmental impacts that will result from the significant differences, starting in the 2024 MY, between CARB's HDOH emission standards and EPA's federal HDOH emission standards. Fourth, the SIP Strategy EA failed to assess in any way the likely significant pre-buy/no-buy response from HDOH vehicle purchasers that the adoption of the Omnibus Regulations will cause. Nor did that EA consider how that pre-buy/no-buy response will be augmented due to the ACT Rule's year-by-year elimination of the diesel truck market, and due to the coincident new-vehicle purchase deadline that the Truck and Bus Regulation has set for the beginning of 2023, the year before the Omnibus Regulations will take effect, which is the same year that the anticipated pre-buy/no-buy response will reach its initial peak. And fifth, the prior EA failed to undertake any meaningful analysis of the NO_x-disbenefits that could result from the implementation of the Omnibus Low-NO_x requirements, especially in the western portions of the South Coast Air Basin (SoCAB), where the prevailing "VOC-limited" conditions mean that incremental reductions in NO_x will cause ozone levels to increase. That phenomenon is well understood, including by the leading air modelers at the South Coast Air Quality Management District (SCAQMD). (See, e.g., SCAQMD Response to Comments on 2016 SIP, pp. 383, 510.) Indeed, the recent absence of ozone reductions in the SoCAB notwithstanding the dramatic COVID-related reductions in precursor emissions is a real-world example of the disbenefit phenomenon. CARB's failure to address that NO_x-disbenefit issue in any manner in the prior EA, along with the other factors listed above, renders its use as the EA for this rulemaking wholly inadequate under CEQA.[4]

CARB's attempted reliance on the potential exemptions set forth in the CEQA Guidelines at section 15162 (see ISOR, p. VII-6) is unavailing, since, among other things: (i) there have been substantial changes in CARB's proposed Low-NO_x program; (ii) PEMS are incapable of detecting or implementing the proposed 3B-MAW-based in-use low-NO_x emission standards; (iii) OBD systems cannot measure and detect emission exceedances or emission-related component failures at the low-NO_x levels proposed under the Omnibus Regulations; (iv) the anticipated pre-buy/no-buy response (including as quantified by ACT) does raise significant new adverse environmental effects (as does the very real NO_x-disbenefit issue); (v) HDOH engine and vehicle manufacturers are likely to exit the California market in response to the Omnibus Regulations; and (vi) new information relating to the cost-prohibitive and infeasible nature of CARB's proposals has become available – information that further establishes that CARB's projected mitigation measures and

[3] CARB confirms that the Omnibus Regulations will impact small businesses (Notice, p. 24), but also fails to conduct the necessary economic analyses of those impacts. That too is a violation of CARB's rulemaking obligations.

[4] Ramboll Group has prepared a supplemental report documenting the continuing NO_x-disbenefit impacts in the Western, more heavily-populated areas of the SoCAB. A copy of Ramboll's supplemental report is attached hereto as Exhibit "F."

emissions benefits are highly unrealistic. Accordingly, CARB's claim that there are "no new environmental impacts" to consider is utterly without merit. (See ISOR, p. VII-11.)

6. Regulatory Leadtime and Stability Issues

As noted, CARB will be providing only two years of regulatory leadtime for the 2024 MY standards, including the 3B-MAW-based in-use standards. That already inadequate leadtime to comply with the 2024 MY standards is further compounded by the issues manufacturers are facing, and will continue to face, as a consequence of the COVID-19 crisis. Any manufacturer that may have had a window of opportunity to try to alter its internal quality-driven design requirements –– including those regarding concept selection, hardware verification, software planning and final calibration verification – to comply with the aggressive new 2024 standards, the new LLC, the new in-use protocol, and protracted durability demonstration requirements, has seen that window of opportunity close, all the more tightly due to the remote work and furlough constraints that the COVID-19 pandemic has imposed across most of the U.S. and Europe. With product-development efficiencies limited by those continuing constraints, and especially given manufacturers' long periods without access to engine test cells or field-test prototypes, their capability to develop high-quality, compliant products to meet the 2024 MY low-NO_x standards has disappeared, if it ever really existed at all.

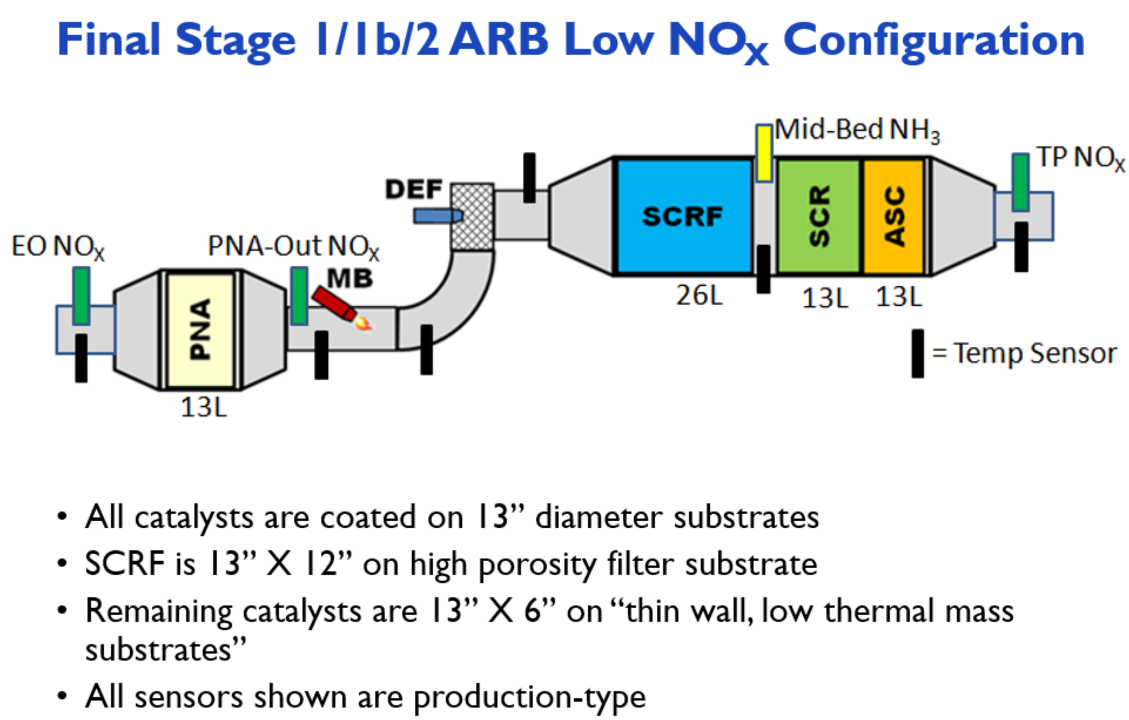
Even without consideration of the significant impacts of the COVID-19 crisis, the severely constrained leadtime at issue plainly violates the applicable provisions of the federal CAA, and will disqualify the Omnibus Regulations from obtaining a waiver of federal preemption. In order to obtain a federal preemption waiver, CARB must demonstrate that the Omnibus Regulations are consistent with Section 202(a) of the CAA. (See 42 U.S.C 7543(b)(1)(C).) Section 202(a)(3)(C) of the CAA requires at least 4 full model years of leadtime before any new HDOH standards relating to the control of emissions may take effect. Since CARB's Omnibus proposals fail to provide that requisite leadtime – and are cost-prohibitive as well - the Omnibus Regulations will not be eligible for a preemption waiver under the CAA and, as a result, will be invalid and unenforceable.

7. The Omnibus Regulations Raise a Number of Significant Technological Feasibility Issues

CARB's ISOR sets forth CARB's assessment of the technical feasibility of the proposed progressively-lower NO_x standards for HDOH engines, which standards would apply to model years 2024-2026, and to 2027 and subsequent model years. As detailed below, the low-NO_x standards that CARB staff envision under the Omnibus Regulations are infeasible without significant engine hardware changes, and so are infeasible on the timeline that CARB staff have mapped out. In addition, the proposed low-NO_x standards for the 2024 MY almost certainly would require increased fuel and DEF consumption rates, and so would have material negative impacts on fuel economy and GHG emissions as well. Thus, those standards are unworkable on top of being infeasible. Further, by the time the 2027 and 2031 requirements are in full effect, and all of the engine hardware, fuel consumption impacts, extended emissions warranty and recall impacts, and other costs associated with CARB staff's proposals are added up, it is apparent that CARB's "Step 2" proposal also would be infeasible, in addition to being cost-prohibitive, as already discussed.

a. The 2024MY Standards are Unworkable

CARB has not made any demonstration proving that engine calibration changes alone are capable of meeting the 75% NO_x-reduction standards proposed for the 2024-2026 model years. In arguing in favor of feasibility, CARB presumes that manufacturers can and will meet the proposed 2024 MY standards – which include an FTP/RMC NO_x standard of 0.050 g/bhp-hr, a PM standard of 0.005 g/bhp-hr, an idle-NO_x standard of 10 g/hr, a new low-load cycle standard of 0.20 g/bhp-hr, and a new 3-bin moving-average-window ("3B-MAW") approach for assessing in-use emissions utilizing a 1.5 compliance factor – without implementing any significant engine or aftertreatment hardware changes. Based on that assumption, CARB further presumes that there is sufficient leadtime for the 2024 MY standards (even though the low-NO_x regulations will not become "final" until sometime in 2021, and even though the leadtime at issue is in direct violation of the CAA). CARB also anticipates that manufacturers could simply utilize a "mini-burner" to help keep SCR systems at sufficiently high low-load temperatures to meet the lower 2024 MY NO_x standards. (<u>See</u> SwRI Schematic below.) CARB's presumptions are unfounded.



The 2024 requirements that CARB staff are proposing <u>*will*</u> necessitate significant engine and aftertreatment hardware changes, which are neither feasible on CARB's proposed timeline, nor cost-effective on any timeline. In that regard, CARB's proposed timeline creates fundamental difficulties for engine manufacturers, difficulties which manufacturers have explained directly and in detail to CARB staff on multiple occasions.

As CARB has heard repeatedly at this point, engine manufacturers are already well along in the process of optimizing and finalizing the calibrations of their engines and aftertreatment systems to ensure robust compliance with the Phase 2 2024 MY GHG standards, while also ensuring sustained low-NO_x emissions. To that end, manufacturers essentially have finalized the architecture, hardware, and performance specifications for their engines and aftertreatment systems to meet the 2024 MY Phase 2 GHG standards. Consequently, CARB's proposals for sweeping low-NO_x requirements in 2024 will create unworkable disruptions to manufacturers' product-development and readiness plans, and, given the inherent trade-off between lower NO_x and higher GHG emissions, unacceptable increases in fuel-consumption and GHGs, thereby threatening manufacturers' implementation of cost-effective compliance strategies for the Phase 2 GHG standards. Accordingly, not only are CARB's low-NO_x proposals infeasible on the proposed timeline, but also any low-NO_x technologies or calibrations that might be implementable could render the Phase 2 GHG standards infeasible as well, especially for the 2024 timeframe, since Phase 2's largest increase in the tractor-engine CO_2-standard stringencies occurs in the 2024 MY, relative to EPA's Phase 2 baselines. But more fundamentally, and as noted, manufacturers' design and production plans are already established for the 2024 MY, which, again, makes the type of redesign-forcing low-NO_x program that CARB has proposed inherently unworkable and infeasible. The impacts of the ongoing COVID-19 crisis only exacerbate those fundamental constraints.

With respect to specific issues of 2024 MY technological infeasibility, it is noteworthy that the schematic for CARB's Stage 1B/2 prototype system (above) includes a close-coupled passive NO_x adsorber (PNA). That device is not a simple add-on, since it has a significant impact on aftertreatment system designs, as well as a significant impact on vehicle and engine packaging to accommodate the installation of the PNA so close to the outlet of the turbocharger. Moreover, multiple PNA washcoat formulations have been developed by several major catalyst suppliers over the recent years, but none have proven sufficiently robust to pass repeated-cold-start durability tests, as conducted by a major engine manufacturer. Additionally, PNAs behave like a sponge with a fixed capacity for NO_x adsorption. Once that capacity has been reached, NOx emissions flow through the catalyst unaffected. In that regard, the capacity of a PNA is sized and designed for targeted operating cycles, such that a PNA's effectiveness in-use is highly variable. CARB has not demonstrated any in-use control capability with the prototype PNA system. In fact, CARB's analysis fails to account for any of the foregoing issues.

Packaging a PNA near the turbocharger also would require a redesign and retooling of many if not all vehicle hoods in the industry. Such a modification to hood designs will have a negative impact on aerodynamics, increasing the emission of greenhouse gases and effectively increasing the stringency of the Phase 2 GHG standards.

In addition, the experimental results produced with the above-depicted "Stage 1B/2" technical solution, as configured by SwRI, achieved a "zero-hour" emissions level on the RMC-SET test of 0.001g/bhp-hr. However, after aging the system to the theoretical equivalent of FUL, the engine and aftertreatment system RMC result was 0.038 g/bhp-hr (with an intermediate point measuring 0.042). While CARB may argue that the Stage 1B/2 system has demonstrated feasibility to a 0.050 g/bhp-hr standard, there is a serious flaw with such an assertion.

If a manufacturer were to present aging results for an engine family certification submission in line with CARB's Stage 1B/2 RMC-SET feasibility demonstration, the OEM would have to declare a multiplicative deterioration factor (DF) for NO_x of 38.0 (0.038 aged result ÷ 0.001 "zero-hour" result = 38; DFs are typically less than 2). That means that if a compliance test were conducted on a production sample, and that sample engine generated a "zero-hour" RMC test result of 0.002g/bhp-hr, just 0.001g/bhp-hr higher than the SwRI experimental article – easily within the range of measurement variability – that production engine would fail the compliance assessment against a 0.050g/bhp-hr standard by more than 50% (0.002 g/bhp-hr x 38 = 0.076 g/bhp-hr). In other words, given the rapid deterioration of the SwRI prototype, a production sample that tested at a level of just 4% of the standard, nonetheless would fail the RMC-SET standard. Consequently, CARB has not come close to making a demonstration of the technical feasibility of the 2024 MY 0.050g/bhp-hr standard at FUL, even with an elaborate and complex technical solution for which there is inadequate development time within the two-year leadtime period that would be available.

The ISOR nonetheless tries to make the case for a 0.050 g/bhp-hr FTP NO_x standard by 2024 on the basis of three considerations. First, CARB reports that the SwRI evaluation of a turbocompound engine (the "Stage 1" engine) achieved a 0.090 composite FTP result with "engine calibration only." Second, CARB asserts that about 40% of heavy-duty engine families already have certified FTP/RMC levels below 0.10 g/bhp-hr. And third, CARB cites simulation-model results from the Manufacturers of Emission Controls Association ("MECA") (the trade association that represents the manufacturers of exhaust aftertreatment systems) that projected a 0.02 g/bhp-hr result with commercially available aftertreatment systems.

It is instructive to assess CARB's first and second points together. A 0.090 g/bhp-hr FTP demonstration does little to demonstrate feasibility, when, as CARB notes, some 40% of HD engines are already certifying at deteriorated results lower than that level today – some as low as 0.050 g/bhp-hr. There are good reasons why such very low FTP-based certification test results nonetheless still need to utilize the resultant large compliance margins to ensure conformity with all of the HDOH long-term compliance requirements associated with today's 0.20g/bhp-hr NO_x standard.

For example, some manufacturers have been able to report laboratory-based FTP/RMC-SET results from in-production de-greened HDOH engine and aftertreatment systems in the range of 0.01 to 0.05 g/bhp-hr. However, no manufacturer currently certifies end-of-useful-life family emissions limits (FELs) at levels less than today's standard of 0.20 g/bhp-hr. The fact that manufacturers continue to certify FELs at 0.20 g/bhp-hr proves that they all understand the continuing need to build in a significant compliance margin for NO_x, as well as for the applicable NTE and OBD threshold requirements. CARB needs to consider all of the factors that manufacturers have taken into account, since they all have made the unanimous determination to certify end-of-useful-life FELs that are no lower than today's standard. CARB should take all of those same factors into account, and should add a similarly sufficient compliance margin to any new proposed 2024 FTP/RMC-SET low-NO_x standard. Not doing so will result in those standards being infeasible.

Regarding the third point, the fact that MECA's simulations may have generated NO_x levels as low as 0.02 g/bhp-hr over the composite FTP is not particularly relevant. Such simulation work actually amounts to the *start* of a feasibility demonstration, which should be followed by rigorous engine testing, deterioration assessment, prove-out across duty cycles and ambient conditions, and even in-vehicle testing. CARB staff would never accept manufacturers' simulation results as a full demonstration of product compliance, so they likewise should not be satisfied with MECA's simulation results as the basis for setting aggressive new emission standards.

With respect to those new certification standards, one important consideration that CARB has not addressed during the course of this rulemaking is the RMC test-point weighting factors. The SET (the precursor to the RMC) was first introduced into the regulatory certification and compliance program for HDOH engines in the early 2000's. Weighting factors were established for each of the 13 steady-state test points on the basis of typical engine duty cycles of that time. Since then, engine designs and calibrations, along with complete powertrain configurations, have led to significant engine down-speeding trends. Recognizing that trend, during the course of the GHG Phase 2 rulemakings, CARB (and EPA) used data from modern down-sped engine designs to reweight the RMC test-point weighting factors. The adjustments made were not insignificant. A full 22% of the engine's weighted emissions output was transferred from the highest speed ("C" speed) to the lowest engine speed ("A" speed). That was determined to be necessary to ensure that the resultant CO_2 emissions from the RMC test would be representative of real-world emissions. There was not adequate time available during the course of the Phase 2 GHG rulemaking, however, to assess the consequences of reweighting the RMC test points with respect to criteria emissions (e,g. NO_x and PM) certification-testing.

Currently, EPA is planning to set new HDOH criteria emissons standards through its Cleaner Trucks Initiative. As part of that rulemaking, EMA anticipates that EPA will take steps to align the criteria-emissions RMC test weighting factors with the new CO_2 RMC test weighting factors. Indeed, failing to do so would result in implementing new regulatory requirements utilizing test cycles no longer considered representative of today's lower speed engines. Not only will the reweighted RMC cycle promote the optimal technologies to achieve real-world emissions reductions, the harmonized test procedures also will provide greater efficiencies for manufacturers in their development and certification processes. EMA supports the alignment of the CO_2 and criteria-emission RMC weighting factors. CARB also should align those test cycles to achieve enhanced environmental benefits and regulatory efficiencies.

Returning to the issue of feasibility, CARB's ISOR goes on to posit an array of aftertreatment configurations that manufacturers could deploy to meet the 2024 MY standards. But conceptual drawings do not make the case for the technical feasibility of a 0.050 g/bhp-hr standard by 2024. The ISOR also states that "engine calibration strategies that may be used for rapid exhaust warm-up and reduced engine-out NO_x may include increased idle speed, intake and exhaust throttling, post injection, and increased EGR rates." Significantly, each of those potential technologies is detrimental to CO_2 control, and, without adequate demonstration as part of a complete technology package, cannot represent a showing of the feasibility of achieving the combination of the proposed 2024 MY low-NO_x standards and the rigorous 2024 GHG limits.

The ISOR further describes potential aftertreatment enhancements including "thin-walled high density catalyst substrates," and again references MECA's simulations where they describe "iron and copper zeolites in a layered structure or zone-coated with two catalyst formulations on the front and rear of a single substrate."[5] Changes to critical aftertreatment systems of that magnitude cannot be made to comply with new emissions regulations with just two-years of leadtime. That simply is not adequate time to ensure that such systems can be fully developed and verified to achieve, in a robust manner, the high NO_x-conversion efficiencies from aftertreatment

[5] *Technology Feasibility for Model Year 2024 Heavy-Duty Diesel Vehicles in Meeting Lower NO_x Standards*, Manufacturers of Emissions Controls Association, June 2019

systems that would be required. CARB should not base aggressive new NO_x standards on the modeling of unproven technologies, especially while providing inadequate leadtime.

Just as important, CARB has made no effort whatsoever to demonstrate the feasibility of any technology package to meet the new moving average windows-based in-use test procedures and standards (the so-called "3B-MAW" protocol and standards) that would come into force in the 2024 model year. Those 3B-MAW procedures, discussed at greater length later in these comments, introduce a completely new method to assess in-use emissions, over a broader range of operating and ambient conditions, and with associated standards at a fraction of where they are today, while (without justification) prohibiting the use of any PEMS measurement-accuracy adjustment factors (adjustment factors that are, by themselves, double the 2024 in-use compliance limits CARB proposes to set). There have been no test cell evaluations of the Stage 1B/2 prototype engine's ability to comply with the 3B-MAW standards, let alone any rigorous in-use in-vehicle compliance demonstration testing of the Stage 1B/2 prototype operated over the multitude of conditions encountered by heavy-duty tractors and trucks in-use. CARB's apparent effort to skip over the need to present an actual "in-use" feasibility demonstration regarding such sweeping new changes to "in-use" standards starting in 2024 amounts to another fundamental shortcoming of the pending rulemaking effort.

In essence then, CARB has not even tried to demonstrate the feasibility of the full suite of low-NO_x standards and requirements proposed for the 2024 MY. For example, while the FTP/RMC standard will be set at 0.05 g/bhp-hr, CARB concedes that the demonstration program at SwRI was only able to achieve a 0.09 g/bhp-hr composite FTP/RMC result from the 2024 MY "Stage 1" prototype engine and aftertreatment system. (ISOR, p. ES-12.) CARB also concedes that its 2024 MY requirements likely will result in fuel-penalties of at least 2-3%. Similarly, CARB has not demonstrated the feasibility of the proposed LLC standard (or the 3B-MAW standards) with the Stage 1B/2 prototype, nor has CARB quantified the additional fuel penalty that will result from the compliance requirements with the LLC and the lower idle emission standards (which will require higher idle speeds).

Just as significant, and as already explained, CARB cannot demonstrate that manufacturers will have sufficient leadtime to incorporate into their product development and manufacturing plans all of the new elements and technological advances that CARB envisions will be required to meet the 2024 MY standards, which would include: heated urea dosing, improved engine and aftertreatment system calibration, increased EGR rates and higher idle speeds, engine hardware modifications, larger SCR catalysts and improved catalyst substrates, and repackaging and reorientation of aftertreatments systems in vehicles. Indeed, by the time the Omnibus Regulations actually become final in late-2021 after OAL approval, manufacturers would have only 2 full years of leadtime to try to meet all of the 2024 MY requirements. That amount of leadtime is clearly inadequate, and, as noted, is directly contrary to the controlling provisions of the federal Clean Air Act, which would preclude a preemption waiver for the Omnibus requirements, and which would render the 2024 standards and requirements invalid and unenforceable as a matter of law.

i. The 2024 MY Standards will cause significant adverse fuel-economy impacts

With respect to specific fuel-economy concerns, the "Stage 1B/2" demonstration engine's incorporation of a mini-burner (or similar strategy) will result in at least a 2.5% fuel penalty over the FTP/RMC drive cycles, and a fuel penalty of 4.0% or more over steady-state cycles and other test-cycles (including GEM-based cycles) that model heavy-duty vehicle operation. Those fuel-penalty increases will, as stated earlier, require additional vehicle-level technologies to offset the fuel consumption impact, adding complexity and cost to the engine and vehicle. Important in that regard, CARB has not demonstrated (through its work at SwRI or otherwise) that any system that purportedly could meet the planned 2024 NO_x reductions also could comply with the Phase 2 GHG standards, or meet the same performance over the EPA/CARB engine fuel maps when assessed in GEM, without additional vehicle technologies. If more technologies or system modifications are needed, that essentially constitutes an unauthorized back-door increase in the stringency of the GHG Phase 2 engine and vehicle standards.

For example, if a given NO_x technology had a $1,000 per-engine cost and a 1.0% fuel penalty, that penalty would need to be assessed in a similar—but opposite—manner as EPA did in its HD GHG Phase 2 Rulemaking. First, CARB would need to identify new technologically-feasible fuel-saving technologies to offset the fuel-penalizing NO_x technology. Those technologies would need to be above and beyond all those that EPA (and CARB) already identified in the Phase 2 rulemaking. Even if a hypothetical new fuel-saving technology was both feasible within the limited leadtime provided, and optimistically had the same $1079 cost per-percent fuel-efficiency improvement as shown in EPA's HD GHG Phase 2 rulemaking,[6] that would nearly double the assumed $1,000 low-$NO_x$ technology cost: $2079 per engine ($1,000 + $1079). However, if CARB were not able to identify new fuel-savings technologies to offset the fuel-penalizing NO_x technologies, then the feasibility of the HD GHG Phase 2 standards would need to be reassessed.

Further, before any low-NO_x technology could be deemed not to result in a fuel penalty, that technology would need to be evaluated in a manner consistent with the Phase 2 engine FTP/RMC-SET standards, the Phase 2 GEM-based vehicle standards, and under real-world driving conditions. CARB has not conducted any aspect of that necessary full fuel-penalty assessment. Rather, SwRI has simply concluded that the CARB-sponsored Stage 1B/2 system results in a 2.5% fuel penalty on the FTP and a 1.6% fuel penalty on the RMC-SET, before even considering any GEM-based or real-world results. Thus, CARB has failed to address this core feasibility issue in any sufficient manner.

Significanthly, many of the engine technologies that might be deployed to make up for the fuel-efficiency losses at issue have the effect of reducing exhaust temperatures, which compounds the challenge of achieving additional NO_x reductions. That dynamic is illustrated in the figure below from West Virginia University, showing that exhaust temperatures are expected to be steadily reduced to meet more and more stringent CO_2 standards.[7]

[6] See EPA estimates from 2016 HD Greenhouse Gas Phase 2 regulation at 81FR73559, Table II-7, and also pp. 73620-73621, Tables III-26 and III-27

[7] 'Heavy Duty Vehicle Diesel Engine Efficiency Evaluation and Energy Audit' Final Report Oct. 2014, West Virginia University

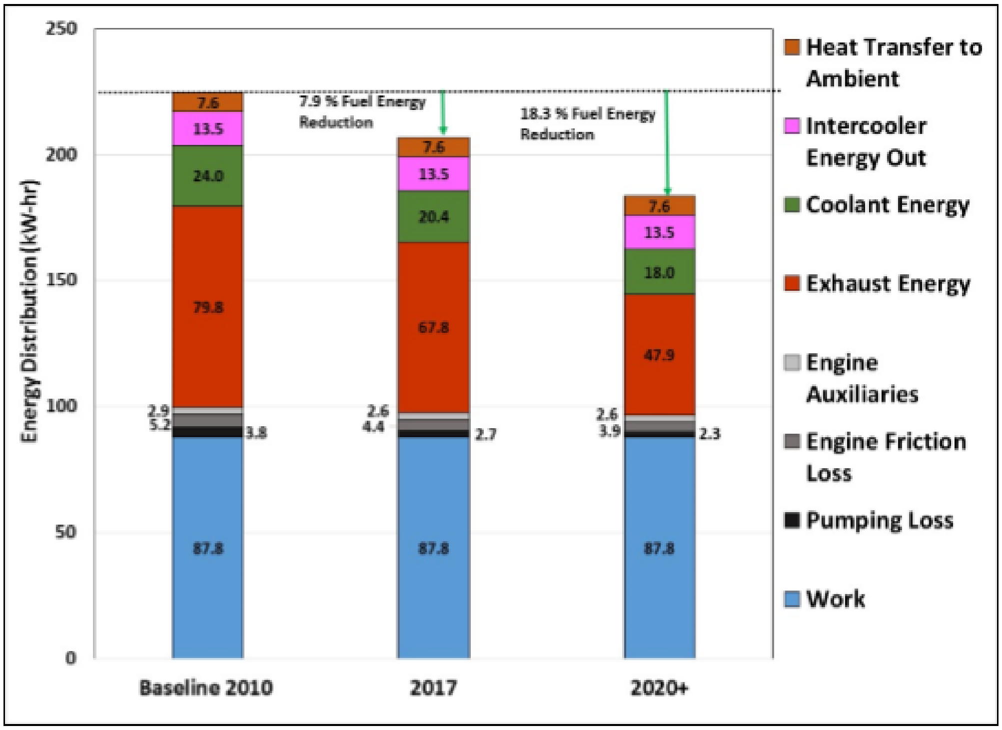


Figure 38 Heavy-duty energy loss distribution for 2010, 2017 and 2020+ engine technologies over SET

CARB clearly has not done an adequate job of quantifying the aggregate adverse fuel-economy impacts of its Omnibus proposals. Nonetheless, analyzing the RMC-SET modal data that EMA obtained from SwRI, EMA has attempted to assess corresponding GEM fuel maps. Compared to GEM results using EPA's 2024 MY stringency fuel maps, the CARB-sponsored Stage 1B/2 technology and calibrations resulted in additional significant fuel penalties for each of the Phase 2 vehicle categories.

In addition to assessing the FTP/RMC-SET fuel-penalty results, CARB also would need to evaluate any low-NO_x technology over real-world driving routes to demonstrate that the technology would not result in an additional real-world fuel penalty as well. Any real-world fuel penalty would result in significant increases in the total cost of ownership of any vehicle with such low-NO_x technology. In that regard, since EPA showed in its Phase 2 GHG analysis that an additional fuel-savings phase-in of 13-25% is worth about $90,000 per tractor within the first seven years of tractor ownership, a 1.3-2.5% fuel penalty, by the same analysis, would cost tractor-operators about $9,000 per tractor.

Because of the additional Stage 1B/2 fuel penalties and the resulting potential infeasibility of the Phase 2 GHG standards, CARB should not consider any fuel-penalizing low-NO_x technologies as support for the feasibility of the proposed 2024 MY low-NO_x standards. Examples of fuel-penalizing NO_x technologies include, but are not limited to, increasing EGR, retarding fuel injection timing, and adding post-injection fuel or mini-burners (as used for the SwRI prototype) to heat SCR as a thermal management strategy. Moreover, and as a practical matter, vehicle purchasers in California are not likely to buy HHD and MHD vehicles with those types of negative cost and complexity impacts to recover lost fuel-efficiencies (not to mention the other significant cost impacts, as detailed above).

As noted, in addition to the proposed NO_x reductions, CARB also is proposing to reduce the PM standard from 0.01 to 0.005 g/bhp-hr. The justification given is that "some engine manufacturers (are) choosing to use less efficient (more porous) DPFs to reduce engine backpressure, resulting in higher PM." (ISOR, II-10, 11.) However, the increase in backpressure that would result from manufacturers adopting "less porous" DPFs would (again) cause higher CO_2 emissions. CARB has not quantified the CO_2 penalty associated with requiring the use of DPFs with higher filtration efficiency.

Tellingly, CARB admits that manufacturers "may find it more difficult to comply with the 2024 GHG standards because of the Proposed Amendments." (ISOR, p. III-26.) CARB's answer to that is simply to state that manufacturers "may need to add additional GHG technologies to bring their engine families into compliance with the 2024 Phase 2 GHG standards," without providing any evidence of the feasibility (or associated costs) of doing so between the time that the Omnibus Regulations would be finalized and the start of the 2024 MY. Here again, CARB has utterly failed to prove the feasibility or reasonableness of its Omnibus proposals.

As a result, if CARB proceeds down its current path, HHD and MHD vehicle manufacturers likely will face the prospect of not being able to produce CARB-compliant products as of 2024, and may be forced to exit the California heavy-duty vehicle market. The net result could be that the proposed 2024 MY standards – which in effect will provide manufacturers with only two full years of leadtime – will cause the HHD and MHD engine and vehicle markets largely to dry up in California in the 2024-2026 time period, meaning that CARB will have adopted regulations to compel the production of HDOH products that few, if any, manufacturers will be able to build, and that few, if any, fleet operators will be willing or able to buy.

ii. Other elements of the Omnibus Regulations demonstrate the infeasibility of the 2024 MY Standards

CARB is proposing to add a new low-load certification test cell cycle ("LLC") to the certification requirements for HDOH engines. The new LLC that CARB staff proposes is a 92-minute test cycle that includes approximately 30 minutes of idle operation, a significant portion of high-to-low load operation with extreme air-flow-induced cooling (i.e., downhill operation), and a significant portion of low-to-high load transient operation (i.e., drayage work). The selected LLC also has an average power that is approximately 6% of maximum power, and an average vehicle speed that is approximately 10 mph. It is an extreme cycle, especially as applied to every HDOH engine, regardless of the vehicle type and application in which the engine might be installed.

EMA has repeatedly questioned the analyses that CARB, SwRI, and NREL relied on to develop the LLC. One concern relates to the portion of the LLC that has been dubbed, "v11660_5". That portion's combination of engine, transmission, 6x4 axle configuration, and 4.20 axle ratio appears to be a heavy-haul configuration, which should mean heavier parts all around. However, the mass—after SwRI's mass reduction and after EMA subtracts a hypothetical 15,000-pound empty trailer—is 11,333 pounds for a GEM-simulated tractor. That tractor weight is not at all realistic. Even a heavy-haul single unit vehicle, like a dump truck, typically is heavier than 26,333 pounds (i.e., without subtracting an empty trailer). For reference, Navistar's regional-haul day-cab with a roof deflector and a 12-liter engine is about 15,000 pounds, and a Daimler Cascadia day-cab with no roof deflector and a 13-liter engine is 16,300 lbs. Those day-cab configurations are among the lighter Class 8 vehicles, yet they are thousands of pounds heavier than the vehicle simulated to generate LLC portion "v11660_5." Thus, it would seem that CARB's LLC is not representative of the actual operation of any actual HDOH vehicle. Similarly unrepresentative is the LLC auxiliary load that CARB is applying. CARB should increase the LLC auxiliary load for HHD engines from 3.5 kW to a higher value in the range of 5.0 to 5.5 kW, so that it is more representative of real-world auxiliary loads.[8]

The SwRI testing that CARB is sponsoring has indicated that current HDOH engine baseline NO_x emissions over the proposed LLC are, on average, approximately 1.00 g/bhp-hr. An EMA survey of member companies' baseline LLC test results corroborated SwRI's 1.00 g/bhp-hr baseline conclusion for HHD engines. However, the MHD engine LLC baseline was significantly higher, on the order of 1.5-2.5 g/bhp-hr. Yet CARB is proposing an LLC NO_x standard of 0.20 g/bhp-hr in 2024 for all HD engines, which amounts to an 80% reduction from the current HHD engine baseline, and an 87-92% reduction from a likely MHD engine baseline. That is not reasonable given the available leadtime.

Although SwRI has reported 93% NO_x reductions (i.e., 0.07 g/bhp-hr) with a partially-aged aftertreatment system and no resulting net LLC test-cycle fuel penalty on its Stage 1B/2 research engine, that research engine at SwRI includes significant hardware and calibration changes that CARB concedes are not feasible to develop fully and introduce into production in the 2024 MY timeframe, including a passive NO_x adsorber and a mini-burner. Even SwRI concurs that the Stage 1B/2 (and now the "Stage 3") research engines at issue are 2027 prototypes, not 2024 demonstration engines. It is clear, then, that the proposed LLC emission standard is not feasible without significant hardware changes. Moreover, SwRI has not evaluated the prototype engine's fuel-consumption impact over any of the Phase 2 GHG engine and vehicle cycles, or over any real-world drive cycles save for CARB's "Southern Route", which was actually an assessment conducted by EMA. It is highly likely that the Stage 2/Stage 3 engine would exhibit a significant fuel penalty on those cycles.

Due to the lack of actual feasibility-demonstration data, CARB's ISOR again turns to simulation "modeling" results from MECA to make the case for the feasibility of the 0.20 g/bhp-hr LLC standard (ISOR III-14). By simulating the effects of increased PM loading on the DPF to the "high end of today's commercially available DPFs" and heated dosing (a technology not yet verified and in production in the HD marketplace), MECA claims to have modeled an LLC NO_x result of 0.18 g/bhp-hr. Critical to that modeled result was a pre-conditioning of the system using the LLC cycle to set up a 50% ammonia storage level at the start of the LLC emissions test. Pre-conditioning with the FTP (as would be required by CARB's proposed Omnibus Regulations), however, produced results in other MECA simulations of the same system that were 65% higher than with the LLC pre-conditioning. Consequently, following CARB's actual proposed procedures

[8] CARB also must detail how to include accessory loads in the power mapping procedure for both the engine-based LLC and the vehicle-based LLC.

for LLC certification testing, including pre-conditioning over the FTP, likely would generate a result as high as 0.30 g/bhp-hr, 50% higher than the proposed standard.

Compounding the infeasibility of the proposed 2024 MY standards, including the LLC standard, CARB also has proposed significant revisions to the current well-established "pre-conditioning" cycles. Pre-conditioning involves running test cycles before a certification demonstration test to "manage the representativeness of emissions and emission controls over the duty cycle and to reduce bias" (40 CFR §1065.518). CARB has proposed to reduce the number of allowed preconditioning cycles, and to mandate that any emissions occurring during preconditioning cycles must be included in the certified test results. None of those proposed changes has been assessed through any analysis linked to the already-minimal allowance for compliance testing margins.

The case of the LLC is especially problematic since CARB has proposed to require two hot-FTP emissions tests prior to the start of the LLC. There is no obvious way to include FTP-generated emissions results into an LLC test (as they are dissimilar tests), and there has been no assessment of the impacts on the LLC standard as proposed, which there almost certainly will be. For example, Infrequent Regeneration Adjustment Factors ("IRAF"s) will be impacted and increased by the new preconditioning provisions (which would require the inclusion of any emission increases that occur during all phases of an aftertreatment "regeneration" event), and will adversely impact the feasibility of the LLC. Accordingly, CARB should not proceed with the proposed changes to pre-conditioning cycles and IRAFs. Further analysis and collaboration with emissions-measurement and testing experts is needed to determine better data-driven alternatives.

Also linked to those new preconditioning requirements is a provision that "emissions performance should not deteriorate, degrade, or decrease upon successive repeats of the certification cycle." That vague requirement provides no meaningful guidance to manufacturers regarding how they should account for test-to-test variability, or small changes in calibrated settings due to changes in the initial certification-cycle test (such as stored SCR ammonia levels or SCR temperature). Moreover, such a requirement is not appropriate for inclusion in the CFR Part 1065 testing procedures; any requirement such as that should be included in the standard-setting provisions.

The new preconditioning and IRAF requirements only add to the conclusion that CARB's technology assessment to demonstrate LLC feasibility as of 2024 is built upon simulations of unproven technology, enhanced by favorable but prohibited pre-conditioning steps, that produce a result with a mere 10% compliance margin, and with clear evidence that the appropriately tested result would be at least 50% above the proposed standard. That does not amount to a sufficient showing of feasibility. In that regard, it should be noted that CARB's highlighting of "modeling" work that MECA claims to have performed to support the feasibility of CARB's proposed standards (see ISOR, p. III-14) is not equivalent to an actual demonstration of feasibility with an actual engine and aftertreatment system in an actual emissions testing facility.

CARB also is proposing to reduce the current low-NO_x engine idling standard — from 30 g/hr to 10 g/hr starting with the 2024 MY. CARB staff have presented limited data regarding the feasibility of that new low-NO_x idling standard as of the 2024 MY. The ISOR references the SwRI "Stage 2" report as justification for the reduced Clean Idle Standard. The referenced data, however, were generated with an engine equipped with an intake air throttle to achieve reduced exhaust flow, and with high EGR rates, both of which SwRI reported as key components for achieving the reported levels. However, at idle conditions in cold ambient temperatures, high EGR rates raise concerns about EGR-cooler fouling. In addition, while an intake throttle is a known technology, it is not realistic to expect that the device can be engineered onto all engines or packaged into all chassis by 2024. Furthermore, EMA has surveyed its members' optional "Clean Idle" test data submissions to CARB. Based on an aggregate analysis of those data, while it might be technologically feasible to set lower Clean Idle standards, separate stringencies would be necessary for the two different "modes" of CARB's Clean Idle test procedures.

The 2008 dynamometer-based certification test for the Clean Idle standards involves 30 minutes at low idle and 30 minutes at 1100 rpm idle after a period of engine warm-up. That certification test will present significant feasibility issues for the new Clean Idle standards, since the long periods of idle would result in SCR cooling and reduced SCR NO_x-conversion efficiencies, and a corresponding inability to sustain "certified" NO_x levels over the more-extended periods of idle. That will pose additional serious challenges relating to compliance with the new in-use idle test, described below, especially in colder ambient temperatures. Another concern is that in order to control engine-out NO_x to satisfactory levels during the low-idle mode, calibrations typically will result in elevated hydrocarbon levels. The hydrocarbons in the exhaust stream can accumulate on the surface of the SCR over periods of extended idle. When they are subsequently "burned off" as the engine resumes powered operation after extended idling, the catalyst can be damaged, resulting in reduced long-term NO_x conversion efficiencies due to the "over-temperature" conditions. SCR systems also would experience a temporary loss of conversion efficiency due to the accumulated hydrocarbons blocking catalysis sites until they are burned off. There has been inadequate demonstration during the course of this rulemaking regarding how these well-known challenges will be managed by the SwRI Stage 3 prototype.

Additionally, CARB is proposing that for 2024 and later model years, manufacturers certifying to the optional idle NO_x standard must demonstrate that there is no increase in emissions of CO, PM, or NMHC when tested over the longest idle segment of the LLC certification test.[9] That requirement will force manufacturers to use two PM measurement systems during the LLC cycle, creating unnecessary costs and test burdens. Moreover, a manufacturer using bag-sampling for gaseous emissions would have to perform continuous measurements for comparison of results to the LLC idle segment. If manufacturers are meeting the criteria emissions standards for the LLC and the idle NO_x standard, the proposed comparison should be unnecessary.

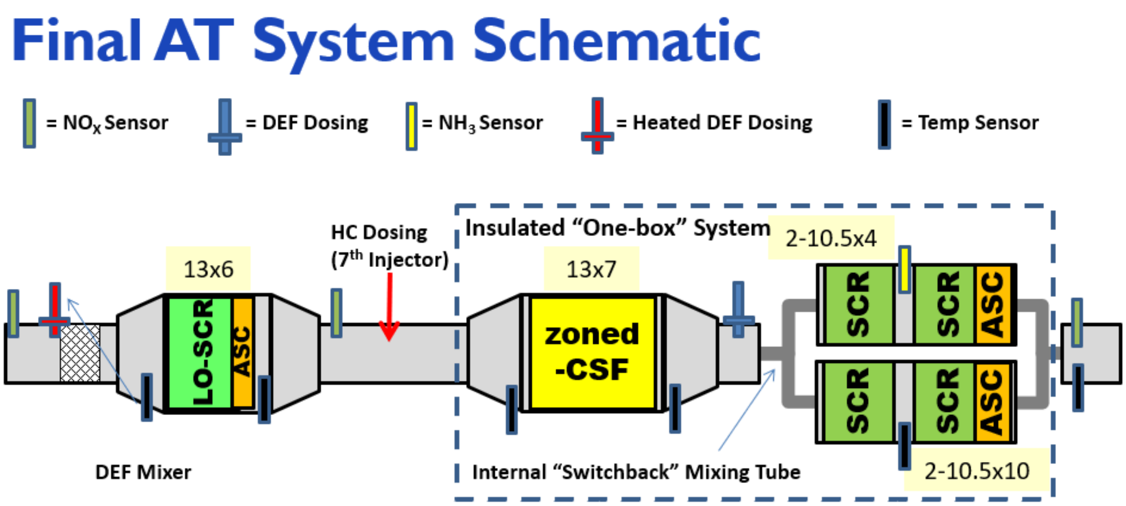
As mentioned, CARB also has introduced a new test procedure to measure "in-use" idle NO_x emissions. That in-use test, however, does not specify a minimum ambient temperature, nor any limit on the duration of the idle period. Those conditions make control of idle-NO_x emissions more challenging than under the current test procedure, used since the 2008 model year. CARB intends to certify engines according to the current idle-test procedure, and it is likely that the current procedure is the basis for any feasibility work that CARB may have done to evaluate the proposed 10 and 5 g/hr low-NO_x idle standards. Yet at the same time, CARB is proposing to add a new "in-use" idle test procedure without making any demonstration of the feasibility of compliance to the new in-use idle test. CARB should not require demonstration to the new low-

[9] The idle segment beginning at 4231 seconds and ending at 5120 seconds in the test schedule.

NO_x standards using the new more challenging "in-use" idle emissions test procedure. Otherwise, and in effect, CARB would be implementing three protocols against which idle-NO_x emissions will be evaluated: the current test-cell test; the "idle bin" protocol (which will be one of the three components of the 3B- MAW procedure); and the in-use idle test. There is no justification or need for three means (two of which are unverified) to assess the same emissions condition.

b. The 2027MY Standards are Unworkable

The proposed 2027 MY standards are similarly problematic, over and above their associated prohibitive costs. CARB envisions that manufacturers will use advanced cylinder deactivation (CDA) systems, an EGR cooler bypass, and the aftertreatment configuration depicted below to meet the proposed suite of 2027 requirements (which include a 0.02 g/bhp-hr NO_x standard, and correspondingly lower LLC, idle-NO_x, and in-use 3B-MAW standards):



LOSCR	13x6	400/4	with ASC zone coated
zonedCSF	13x7	300/7 HAC	DOC coating front CSF coating back
SCR	2-10.5x4	600	dual parallel paths
SCR-ASC	2-10.5x10	600	dual parallel paths, with ASC zone coated

As an initial matter, the complexities and costs of the envisioned 2027-compliant systems, as depicted above, will cause very significant pre-buy/no-buy responses in California, resulting in market conditions that likely will not support the manufacture and sale of CARB-compliant products. (<u>See</u> Section 3.e., above.)

On top of that, CARB again has not made a sufficient showing of feasibility. While CARB's demonstration testing at SwRI focuses on the technology set described above, CARB's ISOR goes on for four pages describing various presentations, papers, research programs and similar endeavors related to other potential low-NO_x technologies and calibration strategies (ISOR, pp. III-17 through 21.) Such passing references to academic work and aftertreatment-supplier development efforts do nothing, however, individually or in combination, to make the case for the technical feasibility of FUL-compliance with CARB's proposed 2027 MY standards, while simultaneously meeting all of the Phase 2 GHG obligations already on the books in the same timeframe.

CARB's statement that "technologies exist today that are capable of meeting the proposed 2027 NO_x standards" (ISOR p.III-16) is simply not true. Again, while it is true that CARB can list emission control strategies and components that do exist, those multiple components and strategies have never been fully deployed in a production-ready heavy-duty diesel engine, and have never been installed in any HDOH vehicle, not even in a prototype vehicle as a part of CARB's "demonstration" work at SwRI. Thus, there is no evidence whatsoever in this rulemaking record to establish that: (i) the large and complex multi-component "Stage 3" prototype aftertreatment system that CARB is relying on could be sized, configured and installed in a drivable HDOH vehicle; (ii) CDA systems can be developed in a sufficient manner to reduce the noise, vibration and harshness issues that have stymied those systems' introduction into HDOH vehicles to date; (iii) the Stage 3 prototype, if ever installed in a HDOH vehicle, could meet the proposed 3B-MAW standards; (iv) the Stage 3 engines and vehicles that CARB is envisioning could still meet the Phase 2 GHG standards in a cost-efficient manner, or in a manner that would not undermine the cost-effectiveness and feasibility premises of the Phase 2 GHG rulemaking; (v) the complex multi-component Stage 3 prototype engine and aftertreatment configuration could ever meet the durability and FUL requirements that CARB is establishing as additional elements of its Omnibus Regulations, especially since CARB concedes that the initial Stage 3 prototype was "aged" only to the current useful life period of 435,000 miles, and that the final prototype has only been aged to 290,000 miles (ISOR, p. III-27), not the 800,000 mile FUL that the Omnibus Regulations will mandate; or (vi) the complex Stage 3 prototype engine and aftertreatment system could ever be equipped with sufficient OBD sensing and diagnostic capabilities to satisfy the other myriad HD OBD regulations that CARB has imposed as preconditions to the certification of HDOH vehicles and engines.

It also is important to consider that the "Stage 3" technology set that serves as the basis for CARB's purported feasibility demonstration offers little or no improvement to NO_x emissions levels when operating over periods of sustained engine load, the types of operation that should be included in the proposed medium/high-load bin of CARB's 3B-MAW protocol (discussed, infra). For example, in the case of a line-haul vehicle pulling a load at highway speeds, a condition where SCR temperatures with current technologies would be at levels optimal for NO_x conversion, none of the proposed 2027 technologies (i.e., cylinder deactivation, EGR-cooler bypass, LO (light off)-SCR, heated dosing, zone-coated catalyzed soot filters, switchback mixing tubes) would have an impact on tailpipe emissions levels, save perhaps for some marginal effect from increasing SCR sizing. Yet, CARB proposes to set a new 90%-lower NO_x standard associated with that type of already-optimized operation. Specifically, CARB is proposing to use the RMC steady-state certification cycle and to apply a 1.5x conformity factor to the "medium/high" bin in-use limits based on a NO_x standard set at 10% of today's limits. There is no reason to expect that the level of emissions under those already-optimal conditions will be significantly improved, which again undermines the feasibility of CARB's proposal.

Another critical feasibility issue relates to whether the Low-NO_x Regulations will adversely impact compliance with the GEM-based vehicle-level Phase 2 GHG standards. Any impact to GEM outputs from the engine designs and calibrations required to meet the Low-NO_x standards could disrupt manufacturers' ability to comply with the Phase 2 vehicle standards.

In that regard, while some GEM simulations may be included in the future as part of the SwRI low-NO_x research project, it is unlikely that those limited simulations, if they are actually completed, will etablish the continued feasiblity of the Phase 2 Standards. A major OEM has performed GEM simulations using one of their 2021 MY GHG and criteria emissions-compliant engines. Those GEM simulations were coupled with EPA's "stringency setting vehicle" design configurations. EPA used those stringency-setting vehicle configurations to establish the 2021 MY CO_2 vehicle-emission standards, by powering the vehicles with an engine fuel map from a theoretical "stringency engine." Vehicle CO_2 targets were thereby established for the various HDOH vehicle categories. The OEM conducted the GEM analysis on over 100 customer vehicle configurations, modeling them with both the EPA stringency-setting engine, and the OEM's 2021 MY emissions-compliant engine. When simulated over a range of vehicle regulatory categories (vocational, line-haul, heavy-haul) and engine ratings matching those of the stringency-setting engine, the OEM's engine design generated GEM outputs anywhere from 1.6% to 13.8% worse than the stringency-setting engine, though both were compliant with the 2021 engine-based CO_2 standards. That work amounts to additional support for the conclusion that CARB's proposed low-NO_x standards are infeasible, especially when assessed in the context of the previsouly-adopted Phase 2 GHG standards.

Designing for criteria emissions and greenhouse gas (engine and vehicle) compliance is a very challenging engineering effort. The foregoing analysis performed by the OEM shows how much variation in tailpipe emissions there can be with very similar test articles (the simulated vehicles were identical, and the engines were similarly compliant). Thus, the research discussed above clearly illustrates that when setting aggressive new NO_x standards, while also adding new certification cycles and protocols, the resultant deviations in GEM performance can widen considerably. CARB has failed to address this additional, critically important issue, which is another material deficiency in this rulemaking.

In sum, nothing in this rulemaking record sustains any of the foregoing elements of technological feasibility, let alone their cost-effectiveness, an issue that is addressed in detail earlier in these comments. And, again, CARB's pointing to technology "reviews," "outlooks," and "investigations," or to MECA's "simulation modeling" exercises, is not actual evidence of the feasibility of what CARB is mandating through its proposed Omnibus Regulations. Thus, there is no adequate basis for CARB's foundational claim that meeting the proposed 2027 standards "would be feasible using the same strategies identified for 2024 through 2026 . . . along with some additional engine hardware improvements." (ISOR, p. III-16.)

i. CARB has not demonstrated the feasibility of CDA technologies for HDOH engines and vehicles

One of the key enabling technologies in the Stage 3 prototype engine's suite of engine and aftertreatment solutions is cylinder deactivation (CDA). CDA permits an engine to selectively deactivate certain cylinders from the combustion process, thereby meeting the power demand with fewer cylinders in operation. The active cylinders, doing more work than they otherwise would, generate higher exhaust temperatures, and thereby the exhaust flowing through the aftertreatment is higher in temperature than an engine producing the same amount of work when firing on all cylinders. The deactivated cylinders have no valve motion, and therefore act like air springs, but friction and heat losses are reduced, so engine efficiency improves. This combination of elevated exhaust temperatures, which improves SCR performance, and improved efficiency, which reduces brake-specific CO_2 output, is the key design feature within the Stage 3 technology set that delivers on both NO_x and CO_2 emissions reductions. Rather than having to compensate for efficiency losses typically associated with technologies that reduce NO_x at the expense of CO_2, CDA delivers reductions of both.

The real challenge, however, is to implement CDA on a heavy-duty diesel engine, and have it deliver consistent, reliable, durable performance over 800,000 miles, without creating HD vehicle and cab "noise, vibration, and harshness" ("NVH") issues or driveline torsional problems. That has never been achieved before. Nonetheless, CARB's Low-NO_x Regulations would leave engine manufacturers no alternative (save for exiting the California market) but to successfully deliver CDA on <u>all</u> HD diesel engine platforms installed in <u>all</u> HD vehicle less than 6 years after the Omnibus Regulations are finalized. That is not feasible. And no "modeling" or literature review can make it so.

It is instructive first to consider the design aspects of CDA. CDA is not a bolt-on, one-size-fits-all system that an OEM can purchase off-the-shelf from a component supplier. Each engine valvetrain design is likely to require a unique CDA design adaptation, even if the design strategy is the same. CDA likely will require higher oil flow rates, oil pressures, and distribution to the cylinder heads, and possibly significant engine block and head redesign. Additional electronic communication channels will be needed. Manufacturers also would need to undertake a very substantial amount of work related to electronic control system strategies, and for completely new mapping of gas-flow models, thermal models, and other thermodynamic functions. In addition, engines with CDA are susceptible to oil-control problems when an inactive cylinder acts against a vacuum on the intake stroke while the intake valve is closed. Oil control issues can significantly accelerate SCR degradation.

CDA also will introduce new and potentially catastrophic failure modes, for example, failures to open the exhaust valve on the exhaust stroke of a firing cylinder, and subsequent intake valve and valvetrain failures as the intake valve attempts to open under extremely high pressures. CDA designs under development for HD engine applications could experience that extreme failure mode due to a malfunction of any one of several components in the system. There are no cost-effective "maintenance" actions that can be established to overcome those concerns, and major overhaul of the CDA system before the end of FUL as a means of ensuring FUL emissions compliance clearly would be cost-prohibitive.

In addition, CDA presents complex challenges for OBD strategies and calibration. Threshold diagnostic determination becomes very difficult, since multiple valves individually or in concert may experience either partial or complete failures. In such a case, separate failure modes would require separate diagnostic validation for each failure mode permutation. The OBD challenges would not be limited to diagnostics of the CDA system itself. CDA can significantly alter the required strategies and calibrations of multiple system diagnostics. For example, CDA greatly complicates the ability to diagnose misfire, a detection issue that already is among the more challenging under the HD OBD regulations.

Another major concern associated with CDA, as mentioned above, relates to NVH. The in-line six-cylinder engine configuration that dominates the HD diesel engine market has inherent torsional balance advantages over other configurations. When individual cylinders are deactivated, that natural balance is disturbed, so the engine vibration levels are increased and torsionals in the engine and driveline systems are elevated. The result is increased noise levels and cab vibration levels that can be uncomfortable to the driver, and that can cause increased wear and stress on cranktrain and drivetrain components, and vibration levels throughout the vehicle that can cause performance and fatigue issues for on-board systems. While SwRI did model some work to assess possible deactivation combination schemes to reduce vibration as measured in the test cell, there is a vast difference between vibration characteristics "as modeled" in an emissions laboratory, and those experienced in a HD vehicle on the road. That fact was duly noted by Neely, et al., of SwRI in their related SAE article,[10] where they stated, "Acceptability standards to linear vibration (e.g. measured at the seat, steering wheel, foot pedal, frame rails, etc.) are better understood in a vehicle environment. The system driveline in a vehicle will differ from that in a dyno (test cell) as well, and it is recommended to evaluate driveline response in a typical vehicle setting."

Indeed, one OEM's experience with a prototype CDA in a Class 8 vehicle has shown that, at the lowest loads and speeds, drivers' responses to the experienced NVH issues are not favorable, especially when the minimum number of cylinders are active. Depending on the extent of CDA at a given load and speed, NVH can vary from mildly perceptible to very significant and fatiguing. The concern for manufacturers and fleet operators then becomes whether CDA would adversely impact driver attentiveness, fatigue and ultimately retention. While increasing the number of active cylinders and engine speed can result in a more positive driver response, that reduces the benefits derived from the elevated temperature of CDA. Passive or active engine mounts can help improve those negative responses, but there is insufficient data on the broad range of truck powertrain configurations to know whether those issues can be addressed in a sufficiently effective manner.

Manufacturers of Class 2b-3 vehicles (14,000 lbs and less), where gasoline engines of smaller displacements have been fitted with CDA, are very familiar with the magnitude of the engineering challenges to overcome NVH issues. Each engine installation on each unique vehicle model is its own project, requiring significant resources, multiple technical solutions, and significant verification time. The technical solution, depending on the vehicle model, can include engine-mount tuning, active noise-cancellation systems, exhaust butterfly valves and pipe geometry modifications, active-tuned dampers, and high-torque-convertor slip settings. Manufacturers do not have a sufficient body of knowledge on the broad range of heavy-duty truck powertrain configurations to know how effective those potential technical solutions might (or might not) be in larger engines and vehicles. Moreover, some of those solutions will have negative fuel efficiency impacts.

As noted, the CDA engineering challenge is multiplied by the fact that each CDA installation requires an engineering investigation and a unique combination of solutions. Given the significant differences among heavy-duty truck configurations and applications, those technical challenges could be insurmountable. When the level of customization that occurs with

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[10] *Simultaneous NO_x and CO_2 Reduction for Meeting Future California Air Resources Board Standards Using a Heavy-Duty Diesel Cylinder Deactivation-NVH Strategy*, Neely et al., Southwest Research Institute, SAE article 03-13-02-0014.

each customer's purchase in the HD vehicle market is taken into account, the level of effort, resources and time it could take to implement CDA effectively could quickly become overwhelming.

Accordingly, while SwRI used cylinder deactivation in its Stage 3 engine prototype, there are serious questions about its viability for actual heavy-duty engine and vehicle applications before the end of 2026. Engine designers strive to develop the most efficient engine assembly with the fewest moving/wearing parts to maximize reliability and reduce costs. The addition of individual electro-hydraulic valve actuators, along with all of the associated control components, represents significant diagnostic and durability challenges for successfully deploying CDA. If HDOH manufacturers are unable to address all of those significant issues and challenges with each engine adaptation of CDA, as integrated into each vehicle model and each customer specification, they will be unable to meet CARB's aggressive low-NO_x 2027 MY standards. In that regard, CARB offers no alternative solution that accomplishes the combination of CO_2 and NO_x reductions that CDA enabled SwRI to demonstrate with its Stage 3 prototype. As a result, given the inherent risks and uncertainties that pertain to the actual deployment of CDA in actual HDOH vehicles, CARB has not adequately demonstrated the feasibility of the 2027 MY low-NO_x standards.

ii. CARB has not demonstrated the feasibility of the envisioned Stage 3 aftertreatment systems

Turning to the complex multiple-SCR systems that CARB also envisions for achieving the proposed 2027 MY standards, the control of far less complex systems continues to challenge manufacturers. Manufacturers today still face significant challenges in consistently controlling stored ammonia levels over the SCR substrate under all ambient and transient operating conditions. That parameter must be modeled (it is not measurable), and fluctuations in exhaust flow and temperature can have significant impacts on ammonia levels, and can lead to NO_x "breakthroughs." The ability to accurately control DEF flow and storage with *two* SCR systems under the proposed 2027 technology scheme will be *more* than twice as challenging. The capabilities of that dual system, including the control of stored ammonia, have only been assessed to a limited degree in the CARB-funded Stage 3 prototype work at SwRI. Moreover, any new control hardware and control strategies would require compliance with CARB's extensive HD OBD regulations. CARB has not explained how manufacturers might comply with the rigorous requirements of its numerous HD OBD requirements when certifying such a highly complex system. Additionally, SwRI's adaptation of long-term "trims" in the SCR controller is not allowed under the current OBD demonstration program. The controller would still be "learning" on the cycles where detection of a failed part and MIL illumination is required. If the OBD regulations were modified to allow long-term trim functions, the considerable time it would take, perhaps 40 hours or more, to stabilize emissions through the learning process between OBD monitor demonstration tests would be prohibitive.

Among the many problems facing control and calibration engineers would be the significant challenge of dealing with the thermal inertia effects of two SCR units. While the close-coupled SCR would heat up faster than the post-DPF SCR of today's 2010-compliant systems, it also would delay the warming of the second SCR system. Without cylinder deactivation, or, without CO_2-penalizing heating strategies such as a mini-burner, there would be no more energy in the exhaust to heat those envisioned dual systems than there is today. The dual systems would share the system's heat energy, as available. While the smaller close-coupled SCR could heat up faster under load, it likewise would cool down faster at idle or light load. In real-world in-use applications (not just under highly-controlled laboratory conditions with continuous calibration adjustments), the job of converting NO_x would be shared between the two SCR units through a thermal balancing act, with little or no positively compounding thermal effect. Moreover, extended idle-NO_x would only be marginally improved by the dual-SCR concept, even where exhaust temperatures are purposely elevated to maintain SCR conversion temperatures during extended idle. Idle-NO_x emissions under those conditions are expected to be similar to those under today's systems regardless of the heating strategy that is deployed.

Additionally, the close-coupled SCR system would be exposed directly to the exhaust stream, including direct HC exposures, without the protective pre-filtering effects of the DPF in today's systems. That would necessitate increased regeneration activities ("DeSO$_x$" events) to purge the close-coupled SCR system from the accumulations of HC and sulfur contaminants and from urea (DEF) crystals, which purging can only be accomplished through sustained elevated exhaust temperatures that are not always possible to achieve on all cycles, especially idle and LLC cycles, and which would result in a significant adverse GHG impact.

Here again, CARB relies on highly complex CDA technology, of unknown feasibility, reliability and durability in the heavy-duty engine market, to provide the elevated exhaust temperatures needed for more frequent DeSO$_x$ events, including over the LLC. According to CARB, the LLC was developed to evaluate operation under lightly-loaded duty cycles, which for some vehicles is representative of practically all operation. However, CARB staff has failed to demonstrate any successful DeSO$_x$ reactions over the LLC or idle test cycle.

The importance of ensuring a complete DeSO$_x$ event over the relevant test cycles is clear from the data that SwRI acquired. SwRI recorded the following composite FTP, RMC NO_x, and LLC results at just one-third of the FUL aging of 435,000 miles that SwRI has targeted for its demonstration project. (By contrast, CARB will require an 800,000 mile FUL under the Omnibus Regulations.)

Aging	**DeSO$_x$ temp**	**Composite FTP**	**RMC**	**LLC**
Zero-hour	---	0.017	0.009	0.020
1/3 of "intermediate" FUL (145,000 miles)	500C	0.039	Not avail	0.049
1/3 of "intermediate" FUL (145,000 miles)	525C	0.025	Not avail	0.036
1/3 of "intermediate" FUL (145,000 mi)	550C	0.022	Not avail	Not avail

At just 33% aging to the current FUL period of 435,000 miles (and in reference to the 2027 MY standard of 0.020 g/bhp-hr FTP/RMC, and 0.050 g/bhp-hr LLC), the importance of achieving a complete DeSO$_x$ event is clear. Composite FTP results are approximately double the standard after a 500°C DeSO$_x$ event, and even after a 550°C DeSO$_x$ event, the emissions from the feasibility-demonstration engine exceed the proposed FTP low-NO_x standard.

The higher the required target temperature for effective $DeSO_x$, the more difficult the $DeSO_x$ event will be to achieve during normal operation of the vehicle when operating over lightly-loaded cycles, including the LLC. Temperature escalations under load will be cooled during idle periods. Consequently, it is difficult to assess the real-world feasibility implications without a vehicle test, especially in cold ambient conditions. CARB has not performed any of those necessary in-use vehicle tests. Neither SwRI nor CARB has even hypothesized how a manufacturer would achieve DeSOx with the Stage 3 system under ambient conditions as low as -7°C (the minimum for valid testing in-use). In addition, enhanced regeneration would not address the accelerated poisoning of the close-coupled SCR due to fuel impurities not pre-filtered by the DPF, since that catalyst-poisoning effect is not reversible through regeneration. That effect would be compounded by the fact that, by virtue of its close proximity to the engine and turbocharger outlet, the LO-SCR would degrade more quickly than current SCR systems.

In addition to being exposed to fuel-based contaminants, the LO-SCR, a key catalyst element in meeting the 2027 technology demonstration, will be subject to oil poisoning at a rate higher than experienced by today's SCR systems. Oil derived poisons are known to deposit heavily on the first catalyst brick encountered in the aftertreatment array. The poisons deposit on the front face of the catalyst, which acts to delay catalyst light-off under cold conditions. Oil-derived poisons are not reversible under any engine-based regeneration strategy, and they also can act to reduce the catalyst channel size. Moreover, the interaction of DEF deposits with oil deposits is unknown (particularly under cold-start, and low load operation), but may lead to a further reduction of the catalyst channel size, leading to increased backpressure and associated CO_2 penalties.[11]

Another issue not addressed by CARB is packaging the multi-component Stage 3 prototype aftertreatment system into a HDOH vehicle. One OEM that has assessed some of the relevant packaging issues has found that when parallel SCR paths are configured in a single "can," they cannot be packaged into Class 4 and 5 truck configurations. Additionally, one of the approaches to address the deterioration of catalysts (discussed above) is to increase the catalyst size, which would compound the packaging problems. Consequently, CARB needs to (but has failed to) account for the significant and costly frame redesigns that will be required to package the envisioned Stage 3 aftertreatment system, including the likely effects on payload, curb weight, and safety. CARB should update its Cost Assessment to reflect those necessary additional cost increases.[12]

The "one-box" aftertreatment system that SwRI has utilized is configured to promote heat retention of the SCR catalysts to enable more engine operation with favorable NO_x-conversion efficiencies. However, not only does that type of configuration present undue challenges to vehicle

[11] That poisoning effect would be exacerbated by the "thin wall, high-cell density" substrates proposed as a potential low-NO_x technology solution by MECA. Oil poisoning is linear with exposure. In that regard, the accelerated catalyst aging demonstration performed at SwRI exposed the catalyst to only 1/3 of the expected "intermediate UL" (435k mile) oil quantities. No consideration was given to the level of oil exposure expected under the proposed extended FUL of 800k miles (nearly double). That is an inadequate demonstration of the durability of the close-coupled SCR due to oil-derived poisoning.

[12] In the case of Class 8 chassis, the installation of the twin parallel SCR systems would be especially problematic for back-of-cab (BOC) vertical installations (for chassis where it is not possible to mount the exhaust system under or between the frame rails). The inability to configure the envisioned aftertreatment systems in BOC vehicle applications will render the Low-NO_x standards inherently infeasible for those vehicles.

packaging, it also complicates the process for designing for "replaceable" SCR cores. The proposed stringent low-NO_x standards coupled with the nearly doubling of the FUL requirements, all but guarantee that the SCR cores will require replacement at least once during the FUL of the HDOH engines and vehicles at issue. To facilitate the cost-effective replacement of catalyst cores in the field, "in-line" designs (rather than one-box) are preferred. The additional heat loss that comes with the in-line system must be made up for by adding more heat to the exhaust, which translates into higher CO_2 emissions than what CARB is currently forecasting from the results of its low-NO_x research program.

The control strategy that CARB proposes also includes an ammonia (NH_3) sensor. Today, there is only one NH_3 sensor on the market from a single supplier. That device is not adequately durable, showing significant drift after as little as 30,000 to 50,000 miles. OEMs that have used that device for emissions control systems in prior model years have found that, even when new, those sensors do not have adequate accuracy stability. The lack of accuracy, coupled with the in-use drift and sensitivity to other exhaust gases that can lead to false readings, make the current NH_3 sensors unacceptable for use in future low-NO_x emissions control systems. Other sensor suppliers are working on the development of NH_3 sensors, but they are in the early stages of development, and therefore it is highly uncertain whether they will be in a production-ready design stage when engine manufacturers would need to begin their long-term testing for 2027 MY products.

Taking a broader view of NH_3 and NO_x sensor accuracy issues, those sensors do not have the necessary long-term accuracy to provide effective tailpipe emissions control at CARB's proposed stringent low-NO_x levels. A set of NO_x sensors that is "reading low" (the system doses less DEF than nominally required) in combination with an NH_3 sensor that is "reading high" (the system thinks it is dosing too much DEF leading to NH_3 slip) will result in significant under-dosing of the system, and thereby potentially non-compliant NO_x levels. There is insufficient accuracy in the current NO_x and NH_3 sensors to deal with that issue. Additionally, adaptive control strategies, which are intended to ensure emissions compliance as components age, rely on the accuracy of those sensor-based inputs. It would be extremely challenging to design and calibrate adaptive strategies given the inherent inaccuracies of those sensors, especially since those inaccuracies only increase as the sensors age.

Taking into account all of the foregoing additional issues, it is even more apparent that CARB has not demonstrated the feasibility of the envisioned Stage 3 aftertreatment system or the 2027 MY standards.

iii. CARB has not conducted the necessary FUL durability assessment of the Stage 3 prototype system

In addition to the foregoing concerns, CARB's assessment of the durability of the proposed Stage 3 prototype is inadequate. As noted, CARB is proposing to extend the FUL requirements from today's 10 years/435,000 miles to 11 years/600,000 miles for MYs 2027 to 2030, and 12years/800,000 miles for MYs 2031 and later. Yet, when its research work is completed, SwRI will have aged the prototype Stage 3 system only to a theoretical equivalent of the current 10 years/435,000 mile requirements. CARB has made no assessment of the durability of the Stage 3 components out to the extended FULs over which the envisioned low-NO_x systems will have to remain compliant.

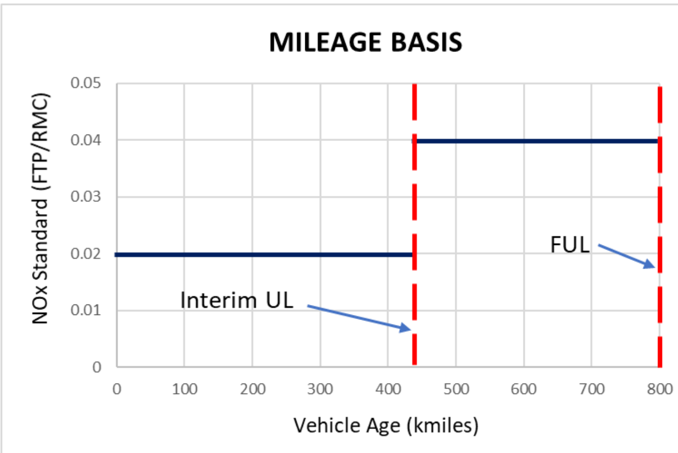
CARB has proposed adjusted, higher NO_x standards for the period after what CARB refers to as the "Intermediate Useful Life," which is today's 10 year/435,000 mile benchmark. From that "intermediate" point on, the engine would have to comply with the adjusted, slightly-higher NO_x standards until the new fully-extended FUL is reached. CARB's implicit recognition of emissions degradation, however, does not excuse CARB from having to demonstrate the technical feasibility of the 2027 and later MY standards out to the new fully-extended FUL as part of the rulemaking process. CARB's failure to make that requisite demonstration is additional proof that CARB has failed to demonstrate the feasibility of the proposed Omnibus Regulations.

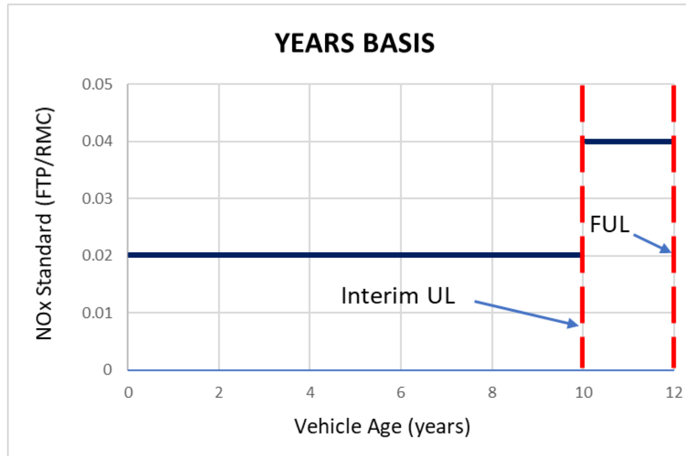
Tellingly, the limited work CARB has done to assess the feasibility of the "intermediate" FUL standards has been done with an aging protocol that CARB considers unacceptable for manufacturers to use for their own deterioration factor ("DF") testing. EMA supports the aging techniques that CARB has used for this work, which involve bench-aging of aftertreatment systems to accelerate the aging process. EMA has been pushing for that type of accelerated process for DF demonstrations for some time, as it would help manufacturers try to cope with the insufficient leadtimes CARB has proposed, including with respect to the 2024 MY standards. CARB, however, has proposed in this Omnibus rulemaking that manufacturers must age engines and aftertreatment systems out to 9,800 hours to develop deterioration factors. That is as much as three to six times longer than traditional DF demonstrations. For engines being certified for 2031 and later model years, the minimum dynamometer-based aging would be reduced to 4,900 hours, but would need to be followed by aftertreatment bench-aging equivalent to 13,100 engine hours.

CARB should not have it both ways. If CARB feels that there is not a well-enough-developed aftertreatment bench-aging protocol for manufacturers to utilize, then CARB should not utilize such an accelerated bench-aging process as a tool for setting aggressive low-NO_x standards linked to new certification cycles and in-use test protocols. Simply stated, CARB has not made a fair or robust demonstration of the long-term technical feasibility of the Omnibus Low NO_x standards. As a consequence, CARB's DF demonstration at SwRI is not sufficient to support this rulemaking.

EMA also is concerned about certain process steps related to the aging-demonstration work that CARB is sponsoring at SwRI. More specifically, the calibration of the Stage 3 test article has undergone numerous changes and adjustments over the course of the aging-demonstration process, including, to improve emissions results or to improve the effectiveness of desulfation ($DeSO_x$). As a result, SwRI has lost track of the baseline condition against which to compare final aged emission levels. A robust demonstration would have involved freezing the calibration from the low-hour test point to the final emissions test. The only way for SwRI to attempt to recover from that lack of a baseline condition is to replace the aged aftertreatment with a "degreened" aftertreatment system of the same configuration, and conduct the full suite of emissions tests with the final version of the engine and aftertreatment control calibration. That would provide an honest assessment of baseline emissions levels, deterioration impacts, and CO_2 impacts across the range of regulated and test cycles. Yet neither SwRI nor CARB has any plans to conduct any such necessary baseline testing.

The extended FUL requirements, in addition to being onerous, expensive, and undemonstrated, present especially unreasonable and unfair challenges for low-annual-mileage HD vehicles. The figures below show the proposed “Intermediate Useful Life” and “Full Useful Life” FTP and RMC NO_x standards on both a mileage-basis (for high-annual-mileage vehicles expected to reach the useful life mileage limits before they reach useful life year limits), and on a calendar years-basis (for low-annual-mileage vehicles expected to reach useful life year limits before they reach useful life mileage limits).





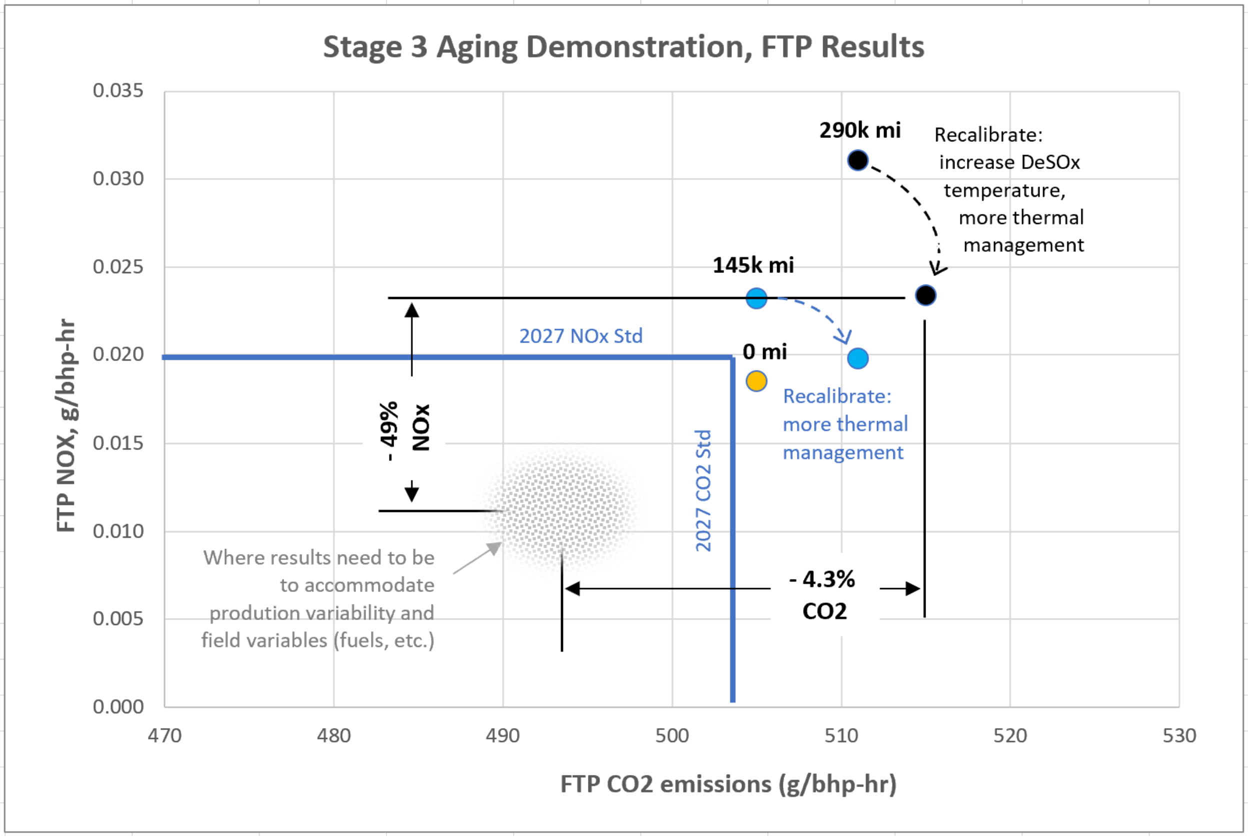
It is evident from the figures above that low-annual-mileage vehicles will have to comply with the initial extremely aggressive 0.020g/bhp-hr NO_x standard over a much greater portion of their FULs than will high-annual-mileage vehicles. Yet no feasibility demonstration has been made regarding the FUL requirements as applied to the low-annual-mileage vehicle case.

iv. CARB has failed to demonstrate the feasibility of the proposed low-NOx standards over the proposed extended FUL periods

A closer look at SwRI’s aging-demonstration test results reveals significant concerns about the inability of the Stage 3 engine and aftertreatment hardware to maintain even marginally-compliant results for just a portion of the proposed FUL requirements. In its July 2020 program update webinar,[13] SwRI presented the results of the FTP, RMC and LLC certification tests of the Stage 3 prototype at the initial “zero hour” and subsequent intermediate test points. The aftertreatment aging program (the Stage 3 engine itself was not subject to aging protocols; only the aftertreatment system was aged using accelerated aging techniques) was designed so that SwRI’s 1000 hours of accelerated aging would simulate 435,000 miles of in-use operation. That level of accelerated aging, therefore, was meant to represent the field-mileage equivalent of CARB’s 2027 through 2030 MY “Intermediate Useful Life” of 435,000 miles, rather than the proposed Full Useful Life requirements of 800,000 miles (for 2031 and later model years).

[13] Heavy-Duty On-Highway Low NO_X Update; Regulatory Status and Latest Demonstration Program Results, Chris Sharp, Southwest Research Institute, July 2, 2020, pp. 24 and 25

The figure below shows the FTP NO_x and CO_2 results when plotted against CARB's proposed 2027 NO_x and CO_2 standards.[14] The zero-hour, 145,000-mile, and 290,000-mile test points are plotted. SwRI performed a $DeSO_x$ event to purge accumulated sulfur compounds from the LO-SCR prior to testing emissions levels. The $DeSO_x$ event, however, failed to achieve the targeted 525°C exhaust temperature, leaving a level of residual sulfur that led to a 0.038g/bhp-hr composite FTP result (not shown on the figure below, which is scaled only to 0.035g/bhp-hr). SwRI then readjusted and reran the $DeSO_x$ routine to achieve the targeted 525°C $DeSO_x$ temperature, which improved the composite FTP result to the 0.023g/bhp-hr level plotted at the 145,000-mile point. (See figure's blue dots.) To make up for the non-compliant 0.023g/bhp-hr result, SwRI then made an additional engine calibration adjustment to add additional thermal management to the hot FTP, improving the composite FTP result to 0.020 g/bhp-hr. That additional recalibration increased CO_2 emissions by an additional 1%.

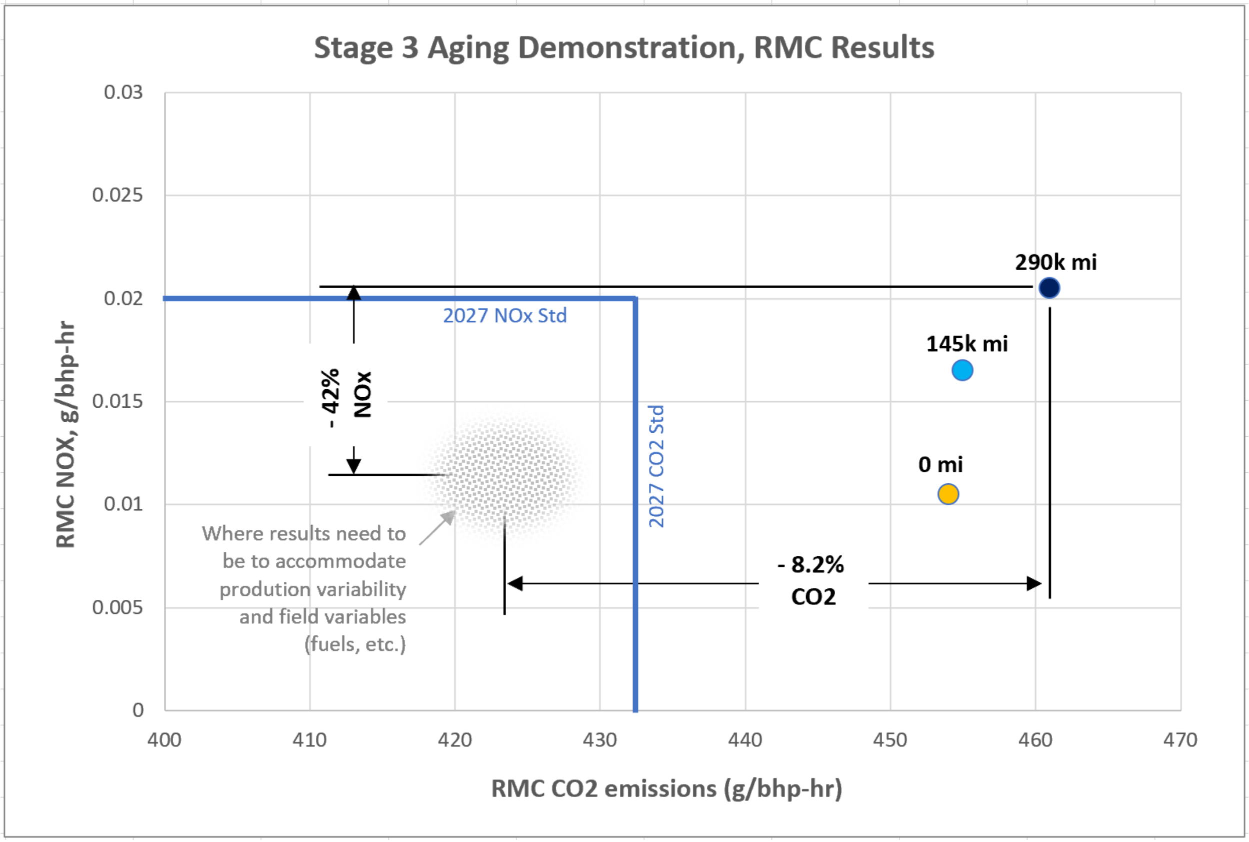


Similarly, after SwRI aged the aftertreatment system further to an equivalent of 290,000 miles, SwRI measured an FTP composite above 0.030g/bhp-hr. To improve upon that result, which was 50% higher than the proposed FTP standard, SwRI reran the $DeSO_x$ event, this time with a targeted temperature of 550°C. The increased $DeSO_x$ temperature was effective in driving off more sulfur such that, in combination with the addition of even more thermal management in the hot FTP through even more recalibrations, SwRI was able to achieve a composite FTP result of

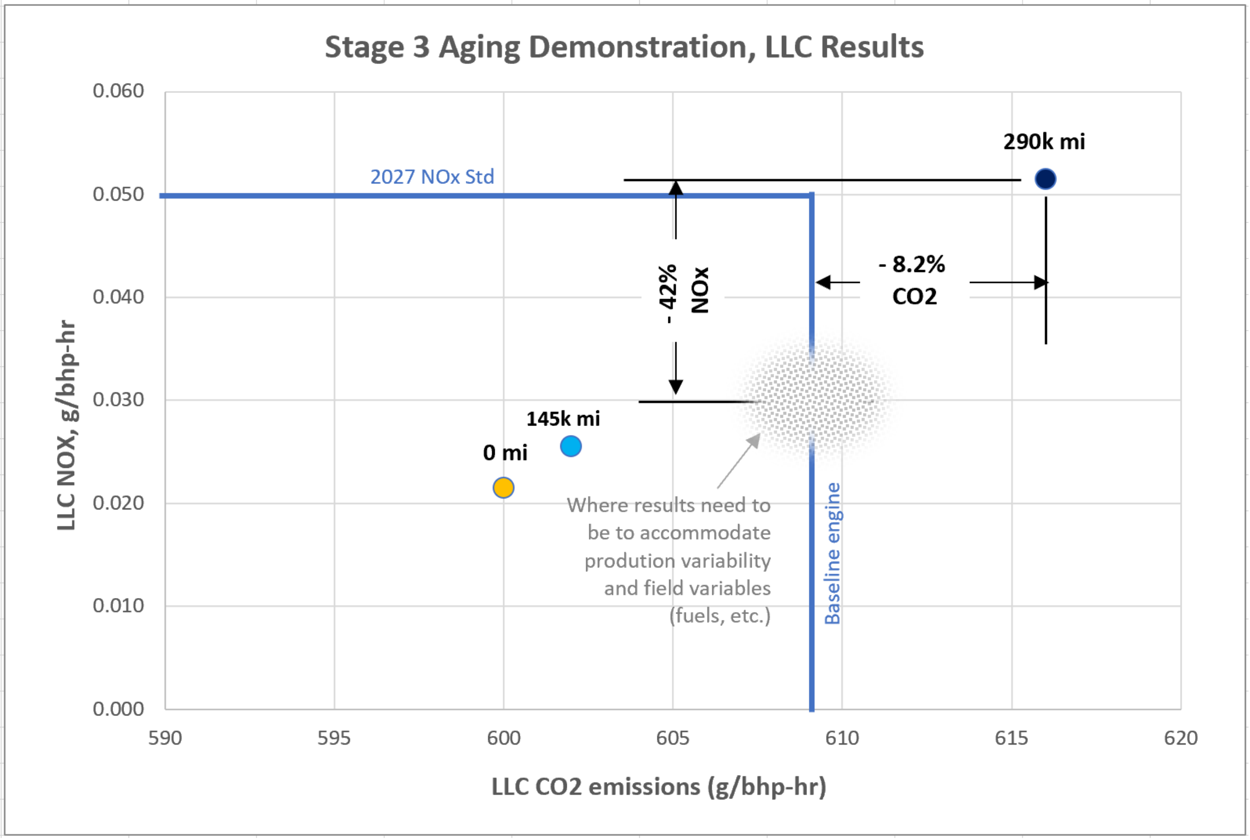
[14] The emission results reflected in the charts on pages 24 and 25 of the SwRI update do not include Infrequent Regeneration Adjustment Factors (IRAFs), as required under the applicable CARB regulations. SwRI has separately reported the IRAFs to be in the range of 0.001 to 0.002, so 0.0015g/bhp-hr (the average) was added to the reported emissions results as plotted.

0.023g/bhp-hr. (See figure's black dots.) While still non-compliant with the proposed FTP standard, that improved result came at the expense of another 1% increase in CO_2 to deliver still more thermal management during the hot FTP.

SwRI's RMC results are represented below. There is less detail in the SwRI update concerning the RMC emissions levels that SwRI achieved before the $DeSO_x$ temperature improvements described above. (Thermal management recalibrations likely did not impact the RMC results or the engine's CO_2 emissions over the RMC.) Those results, while arguably compliant with the 2027 NO_x standard (albeit without any compliance margin), are well in excess of the 2027 GHG/CO_2 standard.



Finally, the LLC results that SwRI achieved during its accelerated FUL testing are plotted below. There is no 2027 LLC CO_2 standard, so the baseline engine's CO_2 emission level over the LLC is represented instead. A fuel efficiency detriment with the 2027 product would not be accepted by a truck-purchasing customer. (Indeed, a customer purchasing a 2027 MY truck likely would expect significant fuel efficiency improvements under all operating conditions with their "GHG Phase 2 Step 3" engine purchase. Nevertheless, for this purpose, the more conservative baseline CO_2 emission level was retained.)



Several observations can be made when reviewing the foregoing data-plots. First, with testing complete only to the level of 290,000 miles, which is only 67% of CARB's proposed *Intermediate* Useful Life, and just 36% of the 2031 FUL, significant emissions deterioration occurs to the point of NOx-non-compliance on all three compliance tests. Second, the uniformly non-compliant NO_x (and CO_2) results were achieved even after SwRI made several adjustments to the $DeSO_x$ routines and thermal management strategies during the course of the test. Third, for manufacturers to include anything close to the NO_x-compliance margins that are necessary (even at the levels needed to ensure FUL compliance at today's NO_x standards, which are 10 times higher than the proposed low-NO_x standards), additional NO_x reductions on the order of 40% or more would be needed if the significant deterioration trends that SwRI observed continue out to the 435,000-mile test-point. Fourth, the additional 40% NO_x-compliance margin would help to account for the observed increases in NO_x emissions caused by sulfur compound accumulation and soot accumulation, both of which were mitigated by the high-temperature $DeSO_x$ and DPF regeneration events that SwRI performed before each emissions test. While SwRI did not report the even higher emission results immediately prior to those $DeSO_x$ and regeneration events, manufacturers' products necessarily would have to be compliant under those conditions. And fifth, while CARB did not set a goal to meet the 2027 CO_2 engine standards in this FUL demonstration, manufacturers nonetheless *must* meet those standards, and will be compelled to supplement the Stage 3 technology set with even more costly technology to reduce CO_2 by another 4 to 8% to comply with the stringent Phase 2 GHG gas standards (EPA would estimate that additional technology to cost from approximately \$4500 to \$9000). Significantly, that does not include vehicle-level CO_2 emission impacts based on GEM outputs, since SwRI did not conduct any analysis whatsoever of those issues. Thus, from all the foregoing, it is clear that SwRI and CARB have not demonstrated the feasibility of the proposed 2027 MY low-NO_x standards over the proposed extended FUL periods.

Given the fact that the prototype Stage 3 engine, even with multiple recalibrations in a well-controlled test cell environment, has not demonstrated compliance with CARB's proposed stringent low-NO_x standards, CARB, at the very least, will need to provide and implement significant "in-use" compliance margins or allowances during the first years of production of any new low-NO_x engines. There is precedent for such necessary in-use compliance allowances in the light-duty GHG regulations and in the HDOH fully-phased-in 2007 standards (effectively, the 2010 MY standards). In those cases (and as clearly pertains here) where significant compliance margins are necessary, the new significantly more stringent emission standards are applied for all certification testing, including DF and IRAF testing, but, for the first model years following the implementation of the new stringent standards and certification protocols, all selective enforcement audits and compliance tests of engines in or from the field are provided an additional compliance allowance before being declared non-compliant. CARB's Omnibus Low NO_x Regulations clearly constitute a "significant standards change and implementation of new protocols." More specifically, with 90% and 50% lower NO_x and PM standards, respectively, coupled with the introduction of the LLC certification cycle and the 3B-MAW in-use protocol, additional compliance margins are clearly warranted, for example, for the 2024 and 2025 model years, and then for the 2027 and 2028 model years. Without such necessary in-use compliance margins or allowances, the anticipated and likely absence of CARB-compliant HDOH engines and vehicles starting in advance of the 2024 MY low-NO_x standards will become inevitable.

In sum, just as CARB's 2024 MY feasibility demonstration is inadequate, so, too, is its attempted demonstration of the technical feasibility of the 2027 MY standards. The SwRI Stage 3 prototype yields emission test results that fail to meet the 0.02 g/bhp-hr NO_x standard after just two-thirds (290,000 miles) of the required first-stage "Intermediate" FUL aging. Moreover, throughout their demonstration effort, the research scientists at SwRI have been compelled to adjust their calibrations, adjust their regeneration parameters, and modify the aging protocol to improve results. All of those recalibrations and regeneration strategies have resulted in increased CO_2 emissions at levels that would not meet the Phase 2 GHG standards for the 2027 MY. And none of the SwRI "demonstration" results include any NO_x-compliance margin, let alone the 40% margins that manufacturers likely will need. In addition, and tellingly, notwithstanding the multiple recalibrations that SwRI was compelled to make to the State 3 prototype, SwRI and CARB have made no plans to rerun, or even re-baseline the Stage 3 engine and aftertreatment system to get a true view of the actual deterioration at issue. In addition, only a single "real-world" replay cycle has been used in SwRI's test cell to assess the Stage 3 systems' performance under the newly proposed in-use 3B-MAW protocol and standards. And not an ounce of fuel has been burned in any actual "in-use" vehicle test to demonstrate feasibility, nor has there been any technical evaluation of a manufacturer's ability to package the Stage 3 systems in a HDOH vehicle.

Simply stated, CARB has not made the requisite feasibility demonstration for the proposed Omnibus Regulations. As a result, the Board should not adopt those Regulations.

c. CARB Has Made No Feasibility Demonstration for Gasoline Engines

Most of EMA's comments to this point have been directed at CARB's insufficient demonstration of the feasibility of the proposed Low-NO_x Regulations as applied to HDOH diesel engines. As part of the Low NO_x Regulations, however, CARB also is proposing to set the same aggressive standards for Otto-cycle engines. Natural gas engines certified to NO_x levels as low as 0.02 g/bhp-hr have been on the market for some years. The early phases of the SwRI research program included an FTP-based demonstration of NG-fueled heavy-duty engines. There has not, however, been any demonstration of those engines' capability to conform to CARB's new B-MAW in-use protocol and standards. And, more importantly, CARB has not made any feasibility demonstration whatsoever with respect to HDOH gasoline-fueled engines. CARB should take the time to perform a proper feasibility assessment for gasoline and NG-fueled engines prior to seeking approval of any new Omnibus Regulations.

8. The Proposed "3B-MAW" Method for Assessing In-Use Emissions is Flawed

One major new element included in CARB's Omnibus Regulations is a new method for "binning" and assessing in-use NO_x emissions, using a second-by-second moving-average window approach, with 300-second windows for collecting in-use NO_x emissions, and a 3-bin approach for sorting and evaluating in-use emissions based on average normalized CO_2 rates (the "3B-MAW" protocol).

The new in-use 3B-MAW protocol and its related standards are integral components of the purported efficacy of the Omnibus Regulations. Consequently, that in-use methodology needs to be thoroughly evaluated to demonstrate its suitability and feasibility as a robust and effective in-use emissions-performance metric. That has not been done with regard to the 3B-MAW protocol that CARB has proposed. To the contrary, the 3B-MAW protocols and compliance criteria are in their early stages of development and are far from being sufficiently validated. EMA and its members see potential merit in "binned" in-use emissions concepts and in exploring new data-processing methods, and have been engaging with EPA and CARB to develop such an in-use protocol. It is clear, however, that despite the concerted efforts of EMA, EPA and CARB to find a viable and reasonable in-use NO_x-binning methodology, the proposed 3B-MAW protocol is nowhere near the level of development appropriate to be a core component of any final emissions-control regulation.

As detailed below, the principal issues demonstrating the unvalidated and, in fact, arbitrary nature of the 3B-MAW protocol are as follows: (i) CARB's NO_x-binning approach will result in individual seconds of data appearing multiple times in each of the 3 bins; (ii) CARB's methodology will result in a sorting, in effect a "smearing," of the same emission data points across all of the proposed bins; (iii) CARB's approach will disproportionally weight certain emission results over others (i.e., some data points will be included up to 300 times, while other points will not); (iv) CARB's proposed "concatenating" of data across key-off/key-on cycles will result in an unrepresentative binning of dissimilar data, which will yield wide spreads in the binned results; (v) there is no discernable correlation among the data points that end up being binned together under CARB's proposal – the data variability and spread do not yield any consistent trends or significant differences among the 3 bins of data, and so reveal no objective justification for the selected bin boundaries; (vi) CARB's proposed binning method results in randomly-binned data, and so is not suitable as a basis for separately regulating those randomly-binned data; and (vii) despite EMA's best efforts to find a workable NO_x-binning protocol, it is clear that using normalized CO_2-rate parameters alone (as CARB proposes) is not sufficient to yield a protocol for binning reasonably correlated in-use NO_x data in a manner that is suitable for applying separate regulatory in-use emission limits.

a. WVU's Expert Analysis Reveals the Flaws in the 3B-MAW Approach

The proposed 3B-MAW in-use testing method and standards do not sufficiently distinguish between modes of in-use engine operation, and so do not and cannot adequately separate in-use emissions into separate bins of idle, low-load, and medium-to-high load operations, as CARB asserts. To the contrary, and as demonstrated by the extensive analyses performed by West Virginia University ("WVU"), CARB's proposed 3B-MAW method simply spreads (or "smears") and comingles in-use emissions data across and among all of the three proposed bins. As WVU's work proves, the binned data under CARB's 3B-MAW method have no adequate correlation, trend lines, consistency, repeatability or reliability of results to support the establishment of separate regulatory standards for the three proposed bins. In fact, WVU's analyses clearly establish that CARB's proposed binning method is, in effect, arbitrary and unreasonable. Moreover, CARB's proposed NO_x-binning method is *not supported by any actual in-use testing data whatsoever*, and CARB has never even tried to assess its proposed binning method using any low-NO_x HDOH vehicle in-use. The 3B-MAW proposal is therefore unworkable, undemonstrated, and unreasonable as the basis for any sustainable regulatory program.

WVU has prepared a comprehensive report of its findings and conclusions regarding the CARB 3B-MAW in-use protocol. A copy of the WVU Report is appended hereto as "Exhibit "G." The WVU report is based on emissions data acquired from WVU's testing of 100 vehicles of multiple vocations operating primarily in the SoCAB. The chart below shows the wide range of vehicle categories that WVU tested, and the number of tested vehicles in each category.

Category	Vocation	EMFAC Class	Vehicle Count
1a	Long haul	T7 NNOOS, NOOS, CAIRP	26
1b	Short haul	T7 tractor	23
2a	Port Drayage	T7 POLA	17
3a	Tractor construction heavy	T7 single construction	5
3b	Cement mixer	T7 single construction	6
4	Tractor construction	T7 tractor construction	8
6a	Food/Beverage Distribution	T6 instate small	8
6b	Moving / Towing	T6 instate heavy	15
7a	Goods distribution	T7 Single	1
7b	Moving	T7 Single	1

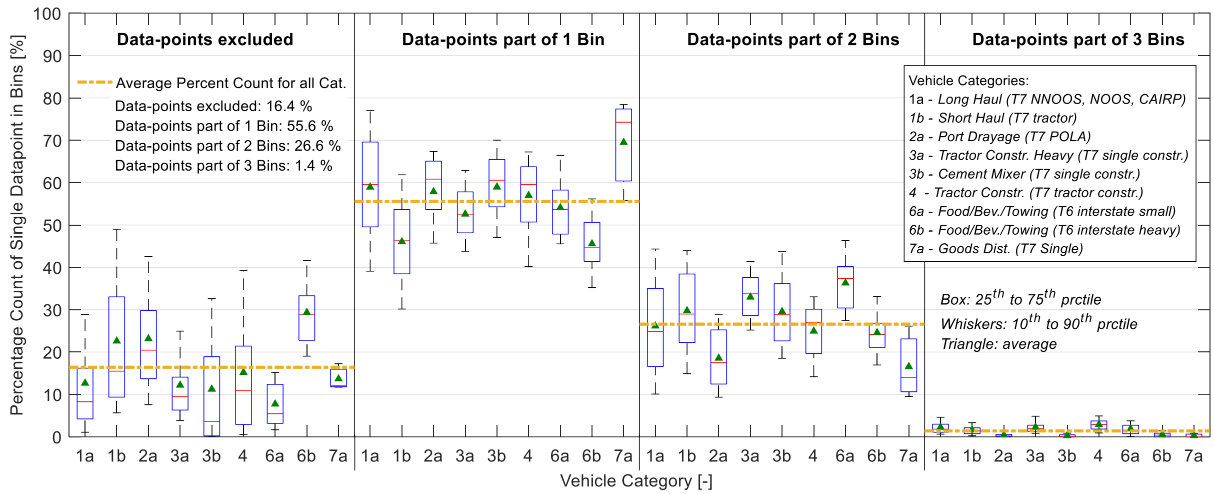
Each tested vehicle was equipped with NO_x-measurement instrumentation for a period of approximately one month. The second-by-second emissions and supporting engine and vehicle data were recorded and stored, and subsequently post-processed by the WVU Center for Advanced Fuels, Engines and Emissions ("CAFEE"). Of particular relevance, WVU has post-processed the large in-use emission data set using the proposed CARB 3B-MAW protocol, and several variations thereof. WVU's results highlight the multiple problems inherent with CARB's 3B-MAW in-use protocol.

As an initial matter, the three proposed MAW-based "bins" do not actually represent idle, low-load, and medium-to-high load operations, as CARB claims in the ISOR. (ISOR, p. ES-9.) Instead, they amount to a varying amalgam of all three bins when the binning methodology is actually applied. Moreover, in the end, the 3B-MAW protocol, with three separate in-use standards for each "separate" bin, in effect amounts to three essentially arbitrary chances to fail the 3B-MAW-based program. Such an in-use compliance-assessment protocol is inherently unreasonable.

By moving the proposed 300-second windows forward on a second-by-second basis, each measured one-second data point is included in up to 300 windows. Those windows are then sorted into one of the three bins. That means that single one-second data points end up being sorted as many as 300 times into some varying combination of the three bins. For example, when second-by-second emissions data were recorded on a vehicle tested over CARB's "Southern Route," 25% of the datapoints fell into two bins, and 7% fell into all three B-MAW bins, rendering the "data segregation" among the three bins largely meaningless. Consequently, under CARB's approach, much of the in-use data, in effect, ends up being randomly sorted and "smeared" across two or even all three of the proposed bins. One consequence of that smearing of results is that the binned data will have limited, if any, correlation to any emissions standard that might applied to the "separate" bins, which undermines the reasonableness of applying separate regulatory standards to the arbitrarily-binned emissions data.

WVU's analysis demonstrates the degree to which the 3B-MAW approach randomly assigns data to the 3 "operational" bins. In the graph below from their report, WVU shows how often single data points fall into two or even three bins over the course of a test day, as assessed for the various vehicle categories included in WVU's 100-vehicle test program.[15] The percentage ranges shown for datapoints in one or more "bins" for a given vehicle category represent the range of individual test-day outcomes for all vehicles in the category. The chart that accompanies WVU's graph shows that, in the aggregate, more than 26% of the measured datapoints end up in two bins at the end of the accumulated test-days. That level of cross-binning of data demonstrates that the 3B-MAW protocol does <u>not</u> effectively sort emissions data according to the targeted binned engine-operating characteristics.

[15] WVU's nomenclature often refers to the three bins this way: "Bin 1" is the idle bin, "Bin 2" is the low-load bin, and "Bin 3" is the medium/high-load bin.



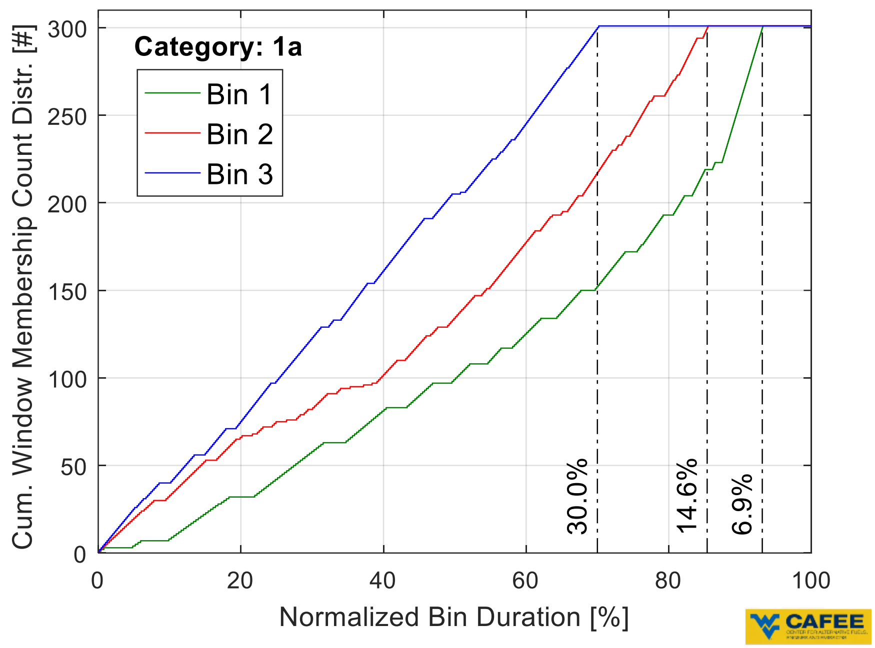
WVU 3B-MAW Report: Percentage count of single data points appearing in either none (i.e. excluded), 1, 2 or 3 bins at the same time for all vehicle categories evaluated.

WVU 3B-MAW Report: Average percentage count per vehicle category of single data points appearing in either none (i.e. excluded), 1, 2 or 3 bins at the same time; global mean represents average across all vehicle categories.

Distribution of Data-points	Mean of Distributions [%]									Global Mean
[-]	1a	1b	2a	3a	3b	4	6a	6b	7a	[%]
Excluded Data (Part of 0 Bins)	12.66	22.61	23.15	12.19	11.23	15.20	7.73	29.33	13.68	16.42
Data-points Part of 1 Bin	58.94	46.03	57.85	52.57	58.95	56.96	54.07	45.55	69.49	55.60
Data-points Part of 2 Bins	26.15	29.75	18.56	32.97	29.54	24.94	36.31	24.58	16.56	26.60
Data-points Part of 3 Bins	2.25	1.62	0.44	2.27	0.28	2.90	1.89	0.54	0.28	1.38

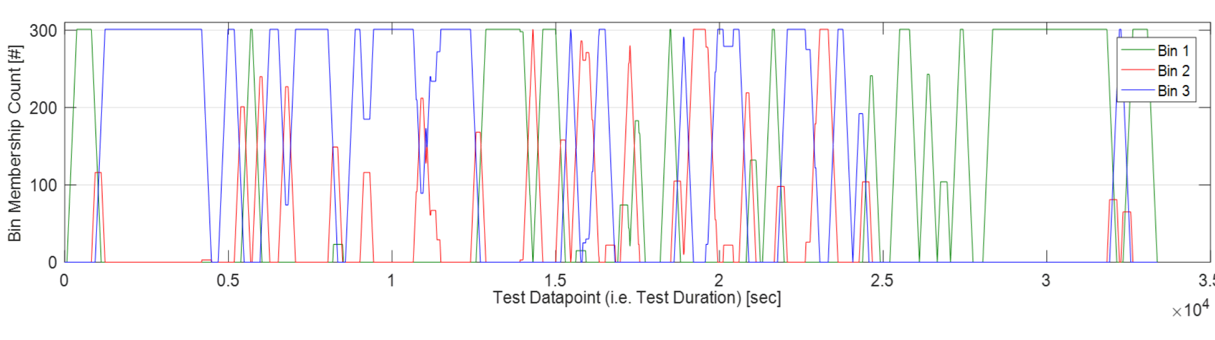
Another very important consequence of the overlapping window approach is that while some measured datapoints will be included in the data set of a particular bin up to 300 times, other points will be included only once, and other data points anywhere in between. That has the effect of variably weighting individual datapoints in the dataset as a whole, and especially within a given bin. The fact that some datapoints can have up to 300 times greater influence on the averaged bin emissions is fundamentally incongruous with a reasonable compliance assessment, especially since that varying weighting is driven solely by chance.

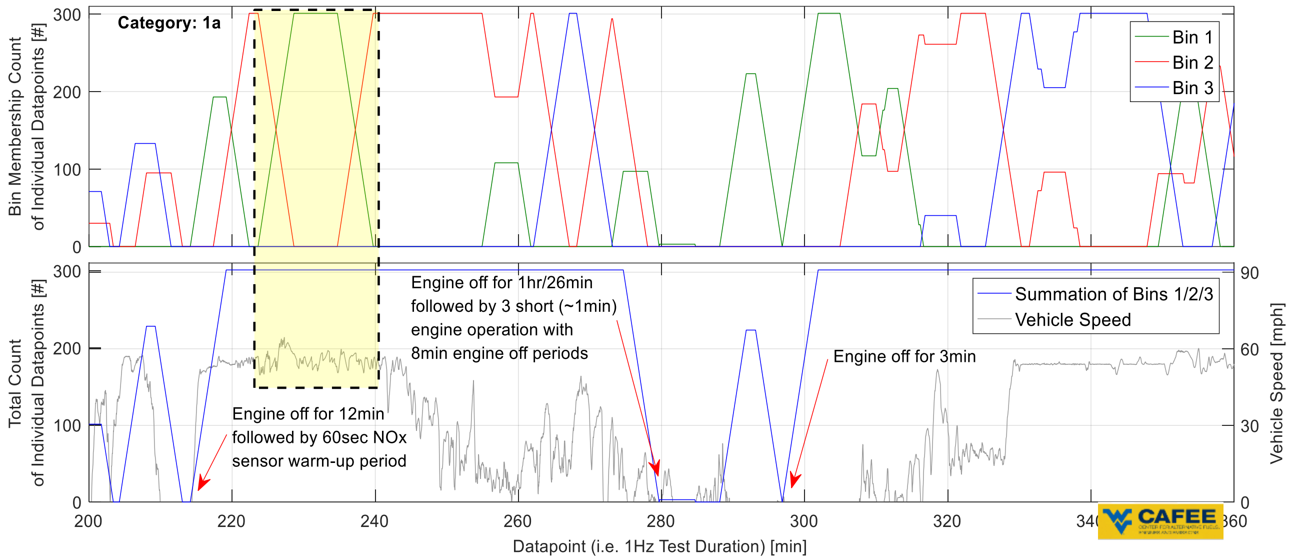
WVU depicts this variable weighting phenomena in the figure below, which indicates the number of times individual data points are used in each of the 3 bins after a shift-day of line-haul vehicle operation. (To understand how to interpret the graph, consider Bin 2: approximately 40% of the datapoints are used 100 or fewer times, 85.4% are used less than 300 times, and 14.6% are used 300 times.) Again, there is no demonstration in the rulemaking record of why this is a fair and appropriate weighting of in-use emissions data.



WVU 3B-MAW Report: Cumulative count of window membership of individual datapoints for the three different bins over the normalized shift-day route for a single vehicle of category 1a; data represents a single day of operation.

The additional graphs below from WVU's report represent 1 hour and 40 minutes of data from a line-haul truck (EMFAC category 1a). The upper graph depicts the number of times individual datapoints (any point along the X-axis) are placed into bins 1, 2, and/or 3. The middle graph uses overlapping lines to show how often the binned data appear in multiple bins. Based on those data, WVU concludes that it is "obvious from [the figures] that transitioning between different bins results in un-equal weighting of an individual datapoint in a given bin," which is a fundamentally flawed approach for regulating in-use emissions in a reasonable and representative manner.





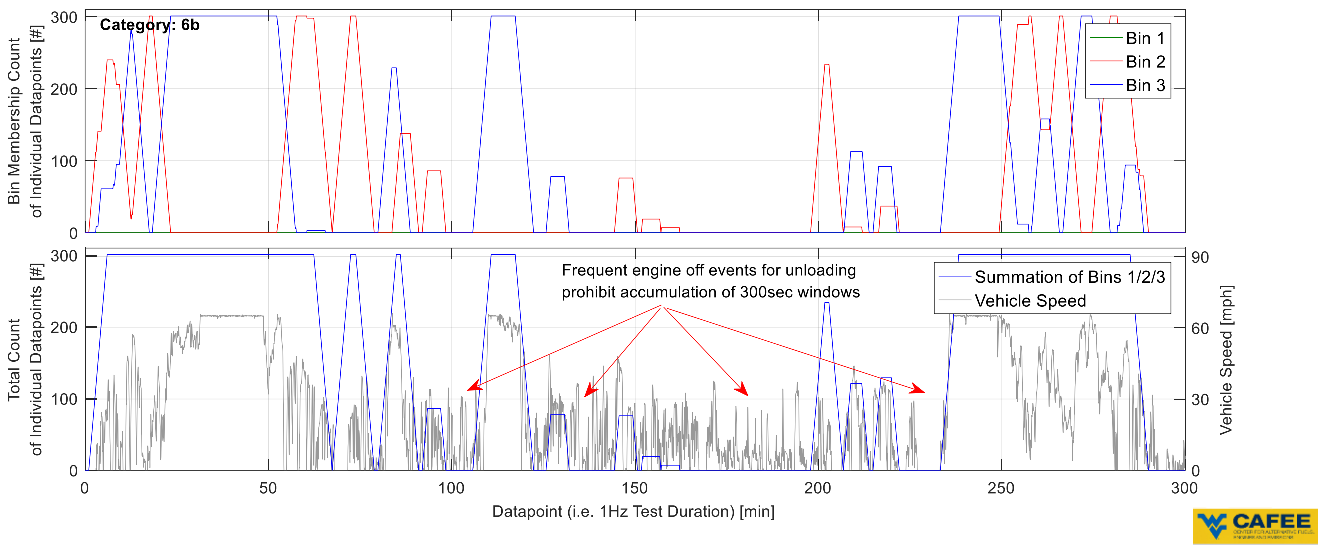
WVU 3B-MAW Report: Bin membership count of individual datapoints to either of the three bins for a category 1a vehicle (i.e. long haul); data taken from a single day of operation.

The lower portion of the preceding graphs represents *total* bin membership of individual datapoints, that is, the summation of datapoints in the 3 bins in the upper graph. This view illustrates the arbitrary weighting effects of individual datapoints within the whole of the 3B-MAW assessment, rather than from a bin-specific view. It clearly shows that individual data points are utilized to widely varying degrees under the proposed 3B-MAW protocol. In addition, key-off/key-on events have corresponding effects on the frequency of usage of individual datapoints.[16] The net result is that many valid datapoints are weighted (used) 300 times, while others are minimally included in the compliance assessment, arbitrarily skewing results through this random-weighting process.

WVU also highlights (in the yellow shaded area) a period of 55 to 60 mph sustained highway speed over a period of about 17 minutes. Illogically, the 3B-MAW protocol places the majority of this operation in the idle bin (demarked by the green line). As WVU states, "It is clearly evident from the vehicle speed trace that this type of operation is definitely not typical idle operation that should be compared to the idle emissions standard."

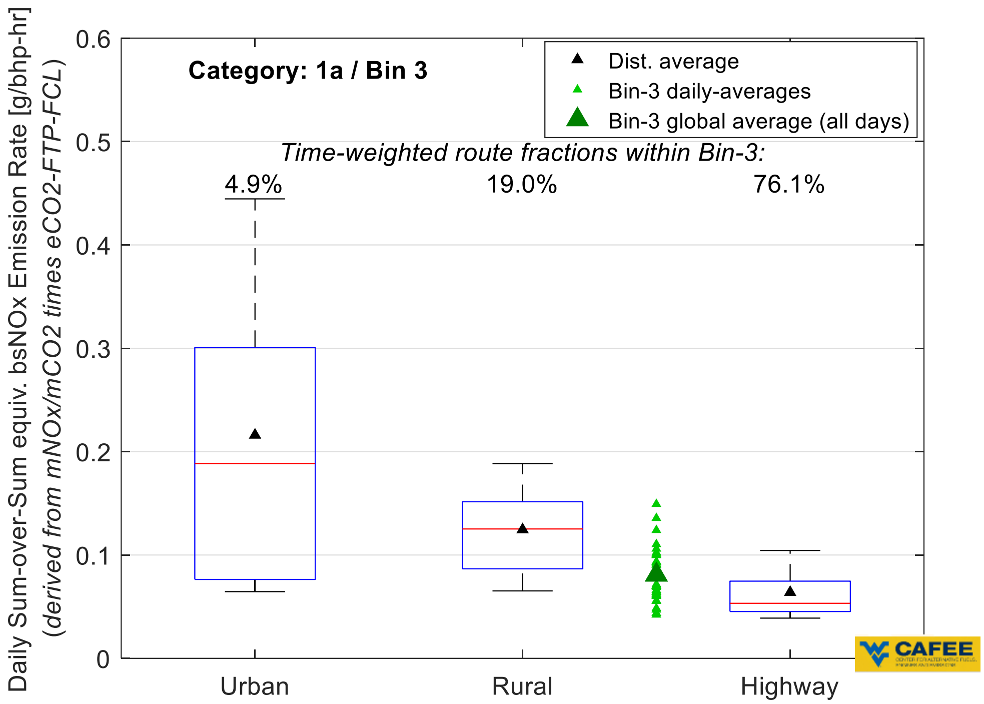
WVU also plotted a depiction of data recorded from a food/beverage delivery truck (EMFAC category 6a) in an urban setting, as set forth below. That vehicle's duty cycle is highly transient with multiple key-off/key-on events. The 300-second-window requrement results in significant data gaps in the compliance-assessment process. Almost 30% of the data is excluded from evaluation.

[16] CARB does not propose to invalidate windows including key-off/key-on events greater than 5 min as they do other data gaps, an omission in the regulation that CARB must correct if the Omnibus Regulations are finalized.



What is especially surprising from this case involving a delivery vehicle, with multiple stops and starts, is how the 3B-MAW protocol ends up sorting the emissions data into the three bins. As WVU's report notes, "the dataset comprises 18.1% idle, 46.8% urban (<31mph), 11.6% rural (>31 and <46mph), and 23.6% highway (>46mph) operation. However, despite the significant fraction of idle and urban stop/go-type driving patterns, Bin 1 does not get populated at all for this vehicle. In fact, a majority of the data, 59%, is attributed to Bin 3." It is completely unreasonable that the 3B-MAW protocol would include no assessment of idle emissions for a vehicle spending significant amounts of its operating time at idle.

In another assessment of whether the proposed 3B-MAW approach effectively segregates emissions data according to engine-operation characteristics, WVU analyzed the medium/high bin (Bin 3) windows from multiple days of testing of a single line-haul vehicle, and separated those data into three ranges of vehicle speed: urban (≤31 mph), rural (>31 and ≤ 46.6 mph), and highway (> 46.6 mph). WVU's graph below shows the variability in day-to-day emissions results from the three speed ranges within Bin 3, the supposed medium/high bin. Clearly the lower speed ranges of the urban cycle produce overall higher emissions results than the higher speed ranges, and show much greater variability from one day to the next. Importantly, the premise that there is a consistent relationship between binned emissions and vehicle operational characteristics (i.e., normalized CO_2 rates) is what CARB is relying on to justify its 3B-MAW proposal. Yet clearly, there are factors in play that have a more significant effect on the level and variability of in-use emissions than the rudimentary bin boundaries that CARB has defined.



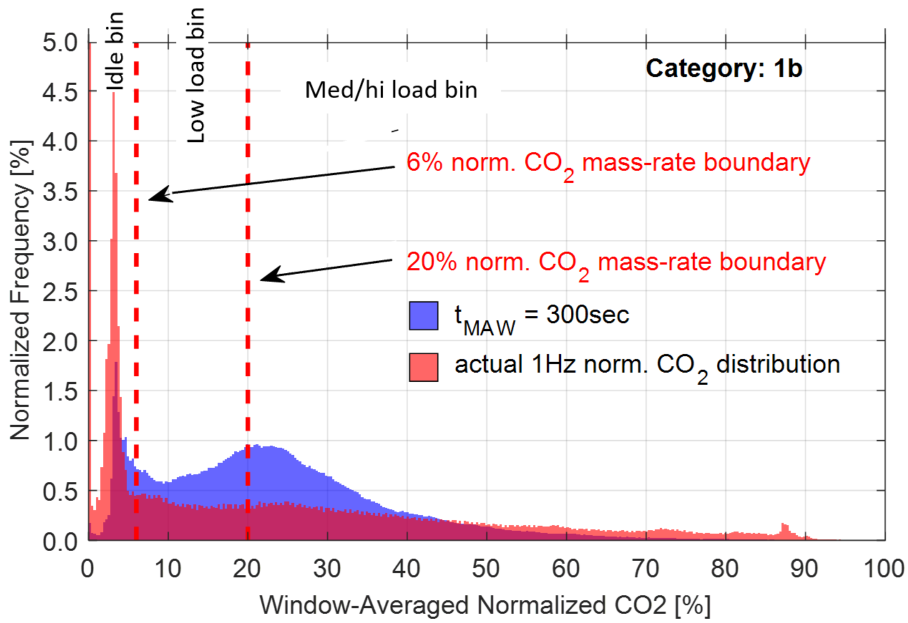
WVU 3B-MAW Summary: Bin-3 emissions for a single shift day of a category 1a vehicle divided into urban, rural, highway operation based on MAW-averaged vehicle speed; urban (≤31mph, rural >31 & ≤ 46.6mph, highway > 46.6mph).

Significantly, none of the WVU analyses concerning the inherent problems with CARB's 3B-MAW approach, with the exception of the vehicle speed breakdown in the medium/high load bin, has anything to do with specific tailpipe emissions levels or low-NO_x technologies. Those results (and problems) would bear out, and the resultant concerns hold true, regardless of whether the analyses involved assessments using today's emisions control systems, enhanced emissions control technologies, or even SwRI's "Stage 3" engine. Thus, the flaws inherent with the 3B-MAW approach will be present no matter which low-NO_x emission standards are promulgated or which low-NO_x technologies are envisioned.[17]

Perhaps the most compelling analysis in the WVU Report is a series of figures showing the real-time percentage of operation at normalized CO_2-rate data points compared to how the 3B-MAW method distributes those same data into the three bins. The figures break that information down for each of the EMFAC vehicle-types that WVU tested. As depicted below, the 3B-MAW process grossly distorts the vehicles' true operating characteristics, capturing and redistributing the data in a way that simply does not match reality. The actual real-time second-by-second operation of a category 1b short-haul vehicle, for example, exhibits predominantly idle and very light load operation with a relatively flat distribution of data at low levels of frequency across the rest of the normalized CO_2 rage. Compare that true 1Hz operation (in red), however, with the 300-second 3B-MAW windowing process distribution in blue, which shows the same vehicle as having

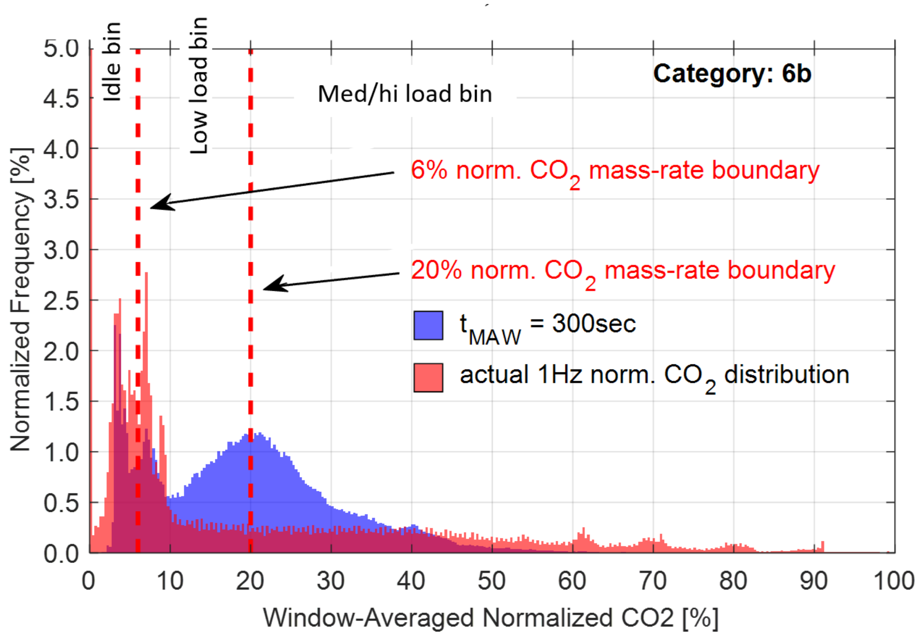
[17] WVU's Report, Exhibit "G" hereto, contains a more detailed explanation and demonstration of each of the multiple flaws inherent with CARB's unverified and untested 3B-MAW protocol.

a strong peak of operation at the boundary separating the low and medium/high load bins, an operating profile that clearly differs from reality.



WVU 3B-MAW Summary: Window-averaged (w/ t_{MAW} = *300sec*, blue dist.) vs. actual 1Hz (red dist.) normalized CO_2 mass rate distributions for category 1b (i.e. short haul) vehicles.

Similarly, comparing real-time and 3B-MAW distributions of data for a more vocational vehicle application, such as a category 6b food/beverage delivery vehicle, results in a distortion of data that is even more apparent.



WVU 3B-MAW Summary: Window-averaged (w/ t_{MAW} = *300sec*, blue dist.) vs. actual 1Hz (red dist.) normalized CO_2 mass rate distributions for category 6b (i.e. food/beverage distribution / moving/ towing, T6 interstate heavy) vehicles.

Based on the foregoing, WVU has concluded that "CARB's proposed bin boundaries are misaligned with actual in-use vehicle operations." Accordingly, much more time and effort needs to be devoted to developing an emissions-data segregation methodology that is truly representative of actual vehicle and engine operating characteristics, that can accurately reflect the real emissions contribution of an in-use vehicle, and that can give OEM's a fair opportunity to comply with the highly stringent underlying in-use standards. The proposed 3B-MAW does not meet those necessary criteria.

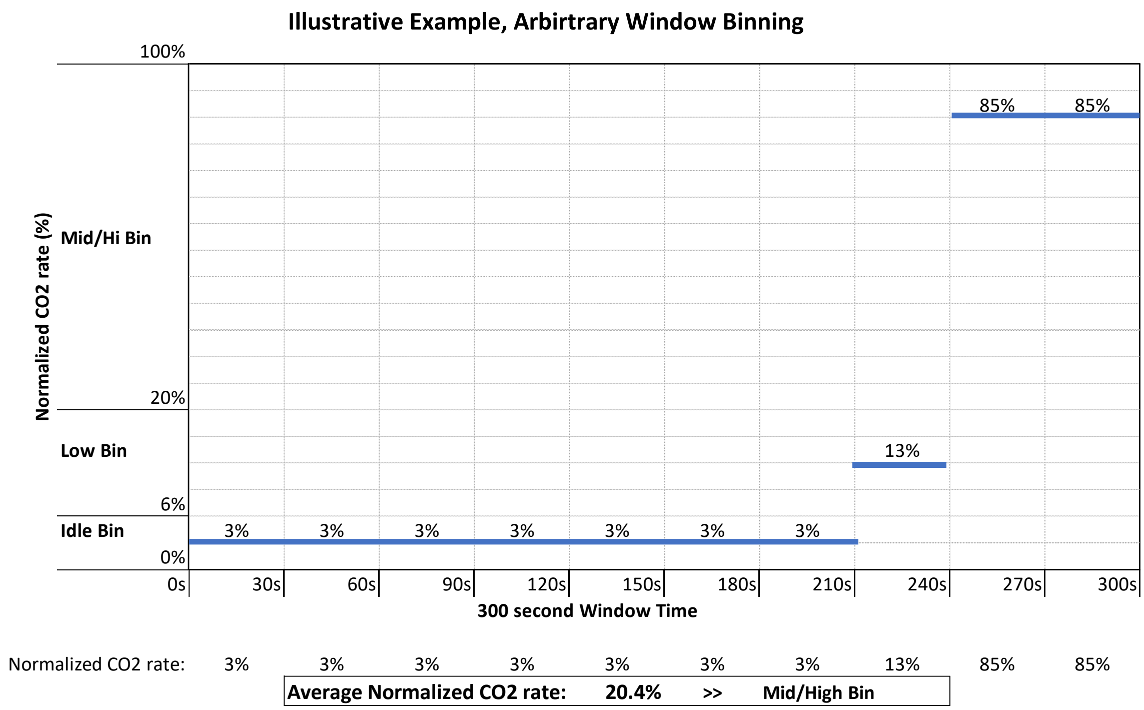
b. Other Issues Highlight the Inherent Flaws in the 3B-MAW Protocol

CARB also claims that SCR-based technology packages result in instantaneous NO_x emissions that are heavily dependent on the engine's recent operating history, which is why CARB proposes to capture "windows" of averaged data under the proposed 3B-MAW approach. While there can be short-term engine-operation effects on emissions, EMA disagrees that windows of data add value to the assessment of in-use emissions, especially when the proposed protocol makes no distinction whatsoever regarding the characteristics of engine-operating history. Two windows can have mirror-image time traces (engine speed, torque, etc.), one with rising SCR temperature, the other with falling SCR temperature, which can certainly yield very different emissions results. Yet the 3B-MAW protocol would bin those windows identically, and hold them to the same standard. Consequently, while CARB's premise is that engine operating history is important, CARB's protocol does nothing to account for the particular details of that operating history.

Instead of advancing 300-second windows on a second-by-second basis, CARB should be working with EPA and industry on the evaluation of a method that advances the in-use data sets on a window-by-window basis (i.e., a "tip-to-tail" window method), as opposed to a second-by-second basis. Additionally, evaluation of recording second-by-second data without applying averaging windows should be conducted, so that ultimately the most sensible and representative methodology could be applied. Further, some analysis has shown that the RMC/FCL CO_2 result is a more favorable normalization factor than the FTP/FCL for CO_2, as CARB has proposed. CARB should evaluate the merits of both of those options, and perhaps other possible normalization schemes, through comprehensive parametric studies. CARB also should present that comparative evaluation to industry and other stakeholders for comment and follow-up protocol- development efforts. Since CARB is proceeding without that necessary thorough parametric evaluation, it is clear that the 3B-MAW protocol is not developed or validated enough to serve as the basis for an in-use regulation.

Other defects inherent in CARB's binning proposal become evident when CARB's new Low Load Cycle (LLC) certification test is processed according to the 3B-MAW in-use protocol. A significant number of windows, especially those including long periods of idle followed by a high-load "return to service" period of operation, end-up in the medium/high-load bin. Consequently, the portions of the LLC most vulnerable to NO_x "breakthroughs" would have to comply with the in-use standard linked to the more stringent FTP/RMC standards, not the higher LLC standard. That is, those LLC windows which would fall into the medium/high load bin would have to meet a 0.030 g/bhp-hr standard, established on the basis of the conformity factor (1.5) times the FTP/RMC standard. If a vehicle is in a generally low-load application, a long idle period followed by a high-load return to power could be the only operating condition where data is placed into Bin 3, putting the in-use test at high risk for a non-compliance determination. The high (and unfair) risk of noncompliance would stem from the fact that the limited amount of Bin 3 data would be the exact type of data that most likely would not meet the Bin 3 standard.

The following figure provides another hypothetical example of the arbitrary way that the the 3B-MAW approach can assign a 300-second window of data to a particular bin. The vertical axis of the figure reflects the normalized CO_2 rate of the test vehicle's operating load/power range, which is the metric CARB proposes to use to determine the bin placement of NO_x emissions. The graph shows the engine's hypothetical normalized CO_2 rate over ten 30-second intervals as data accumulates in the 300-second window. The average CO_2 rate over the entire window determines bin placement. In this example, the engine is running at an idle condition for the first 210 seconds of the window (3½ minutes idling, as may be the case at a traffic light.). The engine then operates over the next 30 seconds at an average CO_2 rate of 13%, the midpoint of the "low-load" bin's 6% to 20% range. For the final minute of operation in the window, the CO_2 rate is in the 85% range. If one then averages the CO_2 rate over the entire window, the "bin-placement" calculation comes to 20.4%, which means the window would be placed into the meduim/high bin, even though it only operated in that bin for 20% of the time.

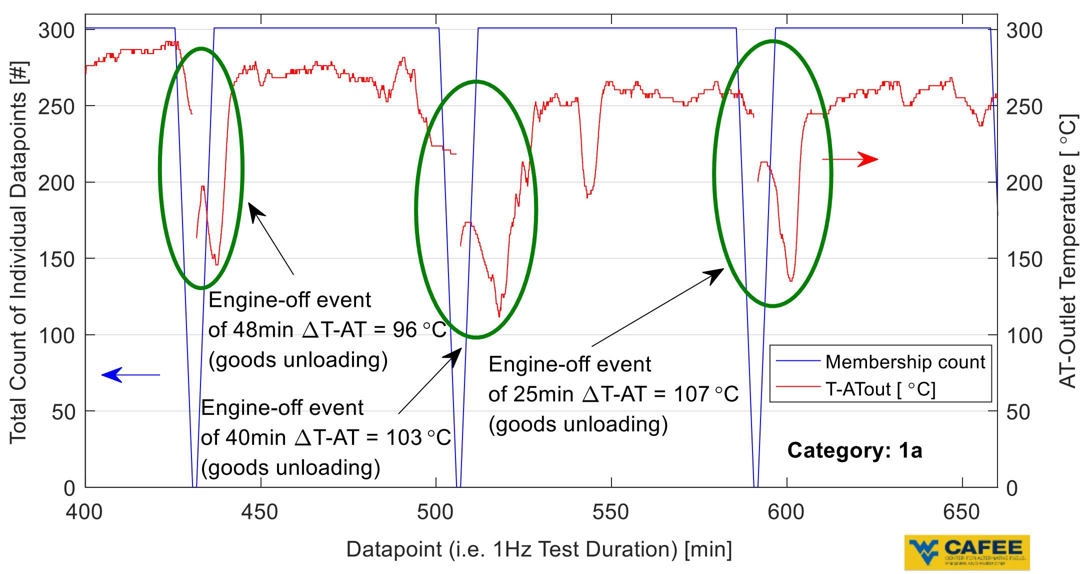


There are at least three ways that this bin placement does not make sense. First, a 300-second window that is dominated by idle operation over 70% of the window nontheless ends-up placed in the medium/high power bin. Second, in actual operation, such a window could have been preceded by SCR bed temperatures providing minimal SCR NO_x conversion temperatures, which would then be prolonged through the extended idle portion of the 300-second window. One could certainly anticipate NO_x breakthrough upon the return to power that starts at 210 seconds, yet the window's emissions would be assessed against the most stringent of the bin standards, the 0.030g/bhp-hr standard of the medium/high bin. Finally, had the window of data been represented in mirror-image, starting at 85% power then progressing to a steady idle condition starting at the 180 second point, the resulting window likely would have much improved average emissions due to the differences in short-term operating history, yet the 3B-MAW process would fail to make that distinction because of its simplistic bin-determination approach.

c. CARB's Proposal to Concatenate In-Use Emissions Data Will Yield Unrepresentative and Non-Correlated Binned Emissions Data

Another area of concern with the 3B-MAW approach, relates to CARB's proposal that data gaps stemming from, for example, key-off events, should be concatenated. That is, data points that are part of a particular operating segment of the vehicle's application, even if there is a cessation of data-generation due to a vehicle coming to a stop, should be stitched together. Stitching data gaps together, however, is directly at odds with the supposed importance of windowing.

The following additional graphic from the WVU Report shows the potential impact of concatenating data. The data set, derived from a commercial vehicle while in normal use, includes periods of "key-off" conditions where the aftertreatment temperature is reduced during the key-off event. When the engine is restarted, it starts with a core SCR temperature significantly lower than when the key-off occurred. CARB proposes to connect those two segments of data, bridging the key-off period as if it did not exist, and combining two very different emissions profiles into multiple windows (300 windows, in fact, if there are no window-excluding events).



WVU 3B-MAW Report: Exhaust after-treatment temperature (i.e. SCR catalyst outlet) as a function of B-MAW transitions during time-limited engine-off events (i.e. unloading/loading of vehicle).

CARB's approach (again) simply does not make sense. No reasoned analysis would lump those disparate emissions data together in that way. Accordingly, CARB should not deploy concatenation techniques. Alternatively, CARB should include PEMs calibration events and key-off/key-on events among the sources of invalid data for which a concatenated window greater than

600s in duration may be voided.[18] The regulation should also be clear that any events for which concatenated data would create windows greater than 600 seconds in duration would apply when such events occur *in combination.*

d. CARB's Proposed Bin Boundaries and Compliance Factor Are Arbitrary and Unreasonable

CARB also has failed to explain or justify: (i) why a bin of "idle" emissions should include tractive power emissions up to 6% of an engine's normalized CO_2 rate, while also having an extremely aggressive NO_x standard targeting low-idle conditions; (ii) why the low-power bin should have a 6% CO_2 rate as its low-range boundary, when 6% is the average power (not low-range limit) of CARB's proposed low-load cycle; and (iii) why CARB has selected a 20% CO_2 rate as the boundary marker for the medium/high-power bin, when that value seems extremely low. Just as important, CARB has provided no data demonstrating that its 3B-MAW approach is reasonable or feasible when the proposed uniform in-use emissions-compliance factor of 1.5 (1.5 times the relevant idle, LLC or FTP standard) is applied to the average emission rates in the idle, low, and medium/high bins.

The inclusion of the up-to-6% CO_2 rate in the idle bin has the net impact of increasing the stringency of the proposed "Clean Idle" standards. CARB has set the NO_x limit for what they call the "idle bin" at 1.5 times the idle standard (without any allowance for measurement accuracy). In order to meet the in-use idle bin requirement, which would include tractive-effort emissions under the 3B-MAW protocol, the actual idle emissions in that bin would need to be significantly lower than the proposed Clean Idle standard of 5 g/hr (2027 and later model years; 10 g/hr for model years 2024-2026). If an engine is operated near the 6% "idle" bin boundary, its fuel flow, and therefore its normalized CO_2 rate, will more than double compared to idle conditions, which means the NO_x rate (in g/hr) will more than double. Thus, the stringency of the required idle-NO_x emissions will be significantly and unreasonably increased. On top of that, CARB has not assessed the feasibility of complying with an appropriate in-use idle standard at ambient temperatures as low as -7C(<20°F), the threshold CARB has set for compliance. Consequently, either the stringency of the standards for the in-use idle bin need to be greatly reduced, or the binning structure needs to be rethought to eliminate these unintended consequences.

CARB envisions that the 3B-MAW protocol will assess emissions performance for all or almost all of a HD engine's operation over its entire shift-day. Indeed, that expectation is one of CARB's primary objectives in implementing a new in-use protocol, given the relatively limited coverage of in-use operations provided by the current NTE method. The NTE protocol was often problematic for manufacturers as well, because if there were only a handful of NTE events recorded over a vehicle's in-use test day, just one NO_x breakthrough event could mean failing to meet the minimum NTE-based "pass" ratio. Despite CARB's intent, a similar risk exists still with the 3B-MAW protocol. A day's testing may very well capture 99% of the vehicle's operating time,

[18] In those instances where CARB does permit concatenated windows greater than 600s to be voided, CARB would require a "detailed explanation" as to why the windows were voided in each case. That should not be necessary. The objective criteria that allow for the invalidation of a window due to excessive window length are clearly spelled out in the proposed regulatory text. Accordingly, invalidating a window would be based on a completely objective assessment; there is nothing subjective about the "decision" to invalidate windows on the basis of those spelled-out criteria. The requirement to provide a written explanation for invalidated windows should be eliminated.

yet, depending on the duty cycle, any single "bin" still may have a minimal amount of in-use emissions data stored for assessment. Consequently, EMA recommends that CARB include a minimum data requirement for each bin, expressed as a number of windows, or total operating time, or a similar metric. More analysis is needed regarding this issue, but perhaps 30 minutes of data (real data, ignoring the over-counting of individual seconds of data that results from overlapping 1Hz windows) would be a good place to start the additional necessary analysis.

CARB's proposed implementation of the 3B-MAW approach also includes the arbitrary establishment of an in-use multiplicative conformity factor of 1.5 that links each of the three bins to a unique test-cell standard. CARB has made no demonstration whatsoever that the uniform 1.5 conformity factor was derived from any analysis of the three separate bins of NO_x data, or is based on any justifiable assessment of technical feasibility. CARB similarly has made no effort to evaluate the conformity factor and resultant in-use emissions standards against the capabilities of the proposed prototype engines and aftertreatment systems. Nor has CARB evaluated whether an additive rather than a multiplicative approach would be more appropriate. In that regard, and as discussed further below, the in-use conformity factor also needs to be assessed against the limits of detection of the instruments that will be used to assess in-use compliance.

Moreover, if the in-use 3B-MAW standards are intended to be technology-forcing, CARB has made absolutely no demonstration of a proposed technology set or emissions control strategy capable of complying with each of those in-use 3B-MAW standards. On the other hand, if CARB did *not* intend for the 3B-MAW standards to be technology-forcing, such that a technology set and calibration strategy capable of complying with the underlying test-cell certification-standards also should be inherently capable of complying with the new in-use standards, CARB has not made that demonstration either. (Nevermind that the "Stage 3" prototype failed to show compliance even to the test cell standards, as discussed above.) The bottom line is that CARB still has a very significant amount of work left to do to develop and validate the 3B-MAW in-use protocol, establish technically feasible and cost-effective in-use standards, and make a compelling demonstration of that necessary work.

e. EMA Has Done Considerably More Research to Try to Develop a Workable In-Use Compliance Standard than CARB

EMA was an initial proponent of moving to a new in-use-based emissions assessment paradigm, where each vehicle would become, in effect, its own mobile emissions lab. Such a new in-use paradigm, ultimately coupled with telematics, could allow for significant regulatory streamlining and greater assurance of real-world emissions control. EMA remains highly motivated to find a new in-use emissions-assessment protocol that can provide the framework for this new in-use regulatory paradigm.

While CARB has presented little if any data in the rulemaking record to justify its 3B-MAW proposal, EMA and its members have devoted significant amounts of time and money to exploring the strengths and weaknesses of MAW-based emissions binning tools and other potential in-use protocols. To that end, as noted above, EMA contracted with WVU to equip 100 HDOH vehicles with measurement technology capable of tracking emissions in real-world heavy-duty applications over extended periods. EMA has used that vast accumulation of fleet emissions data to evaluate numerous iterations of "binning" and other in-use emissions assessment approaches.

Those iterations have included windowing techniques of various durations, exponentially-weighted moving windows, non-overlapping windows (or "tip-to-tail" windows), 1Hz-based approaches without windowed averages, and methods to better differentiate windowed emissions data on the basis of the engine's short-term operational history. EMA's research has included compliance evaluations not only on the basis of binning techniques, but also on the basis of the vehicle's shift-day "sum-over-sum" emissions. EMA's work also has included evaluation of adaptations to the Euro VI-based in-use testing protocol. Idle-bin boundaries based on vehicle and engine speed were studied, as were higher power level boundaries based on afterteatement thermal state to promote thermal management strategies, as well as brake-specific, CO_2-specific, time-specific, and distance-specific metrics. To support this tremendous effort, EMA has held no fewer than 24 all-day face-to-face meetings with manufacturers to review results, discuss conclusions, and direct the next stage of research activities. Because face-to-face meetings are no longer possible due to the COVID-19 pandemic, the EMA group has continued its research efforts through bi-weekly web meetings.

Unfortunately, notwithstanding EMA's and WVU's extensive efforts (which are detailed in WVU's Report, see Exhibit "G"), EMA has not been able to identify a suitably robust in-use emissions-data assessment protocol. EMA is continuing its investigations. And while those investigations have not yet identified a well-suited in-use testing protocol, they have made one thing abundantly clear: CARB's proposed 3B-MAW protocol is not a reasonable regulatory framework for assessing in-use emissions compliance.

f. CARB's Own Technical Consultants (SwRI) Have Confirmed that CARB's Proposed 3B-MAW Protocol is Not Developed Enough to Support a Regulatory In-Use Compliance Program

As noted previously, CARB contracted with SwRI in an effort to try to demonstrate the feasibility of the proposed low-NO_x standards. An additional component of SwRI's work included an assessment of the type of MAW-based approach on which CARB intends to build its new 3B-MAW in-use compliance program. (See Reference 191 to CARB's ISOR.) SwRI's research and findings confirm that the necessary sensors and electronically-broadcast engine parameters are not accurate or robust enough to implement CARB's MAW-based approach in any reasonable manner, and so are not capable of supporting a valid in-use regulation.

SwRI examined several of the engine sensor-based measurements that would be integral to CARB's 3B-MAW protocol, including engine torque and NO_x levels. SwRI's top-level conclusions, as depicted below in Figure 6 and 67 from SwRI's Report, are that:

> Torque measurement was problematic, especially at low loads. The NO_x sensor measurements were also problematic especially at the lower ranges typical of Low NO_x, and the measurements indicate the need for improvements on tailpipe NO_x sensor performance to support a robust in-use compliance program. At lower loads, a high bias can be seen in the torque error. (SwRI Report, ISOR Reference 191, pp. xiii, 55.)

*   *   *

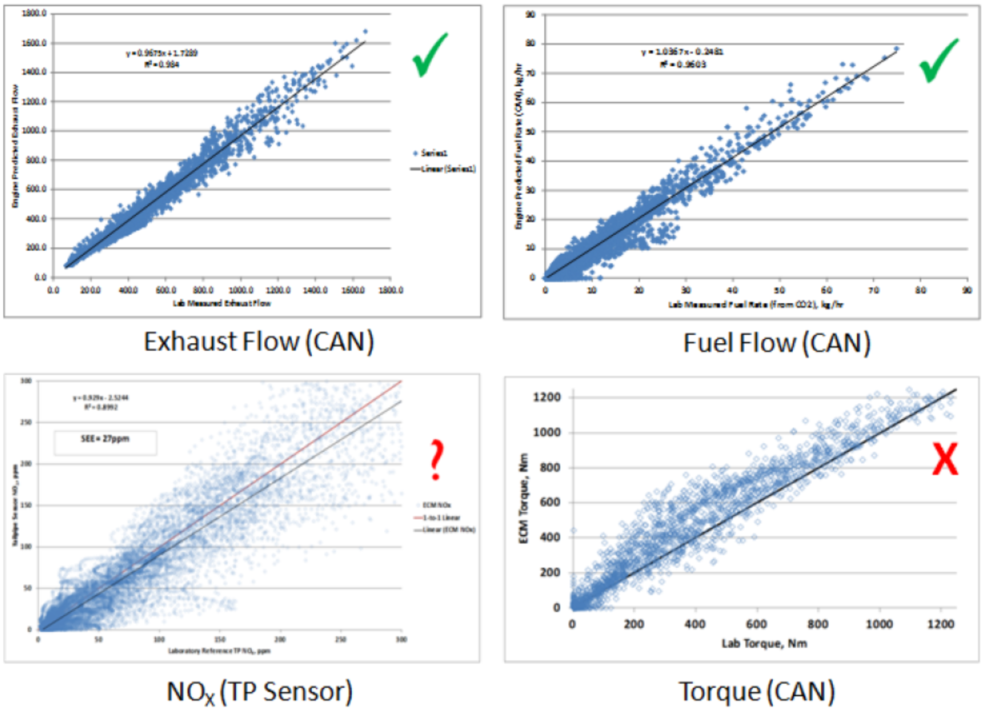


FIGURE 6. SENSOR ACCURACY ASSESSMENT – ECM SENSORS VERSUS LAB REFERENCE MEASUREMENTS

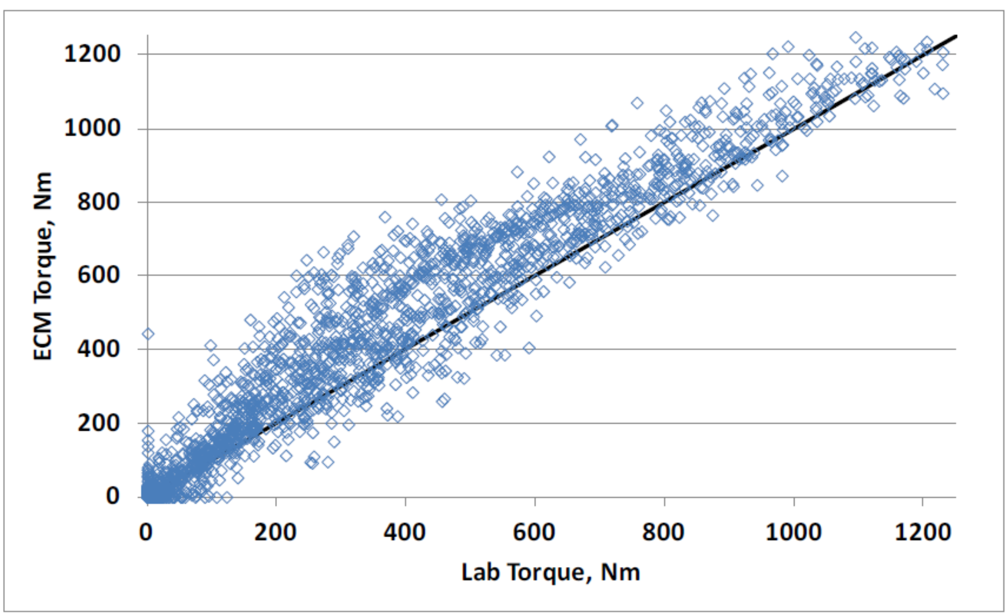


FIGURE 67. REGRESSION OF ECM TORQUE VS. LAB TORQUE – FOCUSED ON LOW-LOAD RANGE ONLY

The foregoing figures confirm that the correlation between laboratory-based measurements of torque and NO_x, and sensor-bases measurements of torque and NO_x, is not sufficiently linear or "tight" to support a regulatory-compliance program. The spread between the two types of measurement is simply too large.

SwRI examined the torque-error issue in more detail and found that "at progressively lower engine loads, larger errors are observed, and an increasing trend towards a positive bias on the ECM Torque can be seen across multiple engines." (SwRI Report, ISOR Reference 191, p.58.) As depicted below, at torque/load levels below 20% (one of CARB's proposed bin boundaries), the error ranges from -9% to +75%. Accordingly, SwRI concluded that broadcast torque (or torque-derived work) should not be used in any in-use compliance program.

TABLE 13. DISTRIBUTION OF TORQUE ERRORS AT VARIOUS LOAD RANGES

	Torque Error Distribution, % pt			
	Percent of Max Torque			
Percentile	< 10%	10-20%	20%-40%	40%+
10th	-26%	-9%	-2%	-3%
25th	8%	8%	4%	0%
50th	**31%**	**32%**	**11%**	**4%**
75th	71%	50%	22%	9%
90th	118%	75%	30%	14%

SwRI's conclusions regarding the magnitude of sensor-based torque measurement errors have clear adverse ramifications for CARB's proposed 3B-MAW approach. More specifically, SwRI made the following recommendation regarding CARB's proposed NO_x-binning concepts:

> It is understood that there is some consideration being given to a "binning" approach, wherein in-use emissions would be grouped into one or more load regimes. If this binning is based on a power metric, such as an average percent of maximum power over a measurement window, then those torque errors could result in the misclassification of measurement windows near a low-load bin. Therefore, even if torque and power are not used as a direct load metric, it is still recommended that improvements to ECM Torque accuracy would be useful under such a classification scheme. (SwRI Report, ISOR Reference 191, p. 58.)

SwRI also examined whether state-of-the-art NO_x sensors are sufficiently accurate at low-NO_x levels to support CARB's proposed in-use regulations. As depicted in Figures 72 and 73 below from the SwRI Report, SwRI found that "substantial errors can be seen on the order of 10% to 20%, which errors grow larger at low overall NO_x mass levels," and that "NO_x sensor data at present are not yet at the same level of accuracy as some of the other EMC broadcast measurements, such as exhaust flow." (SwRI Report, ISOR Reference 191, p. 63.)

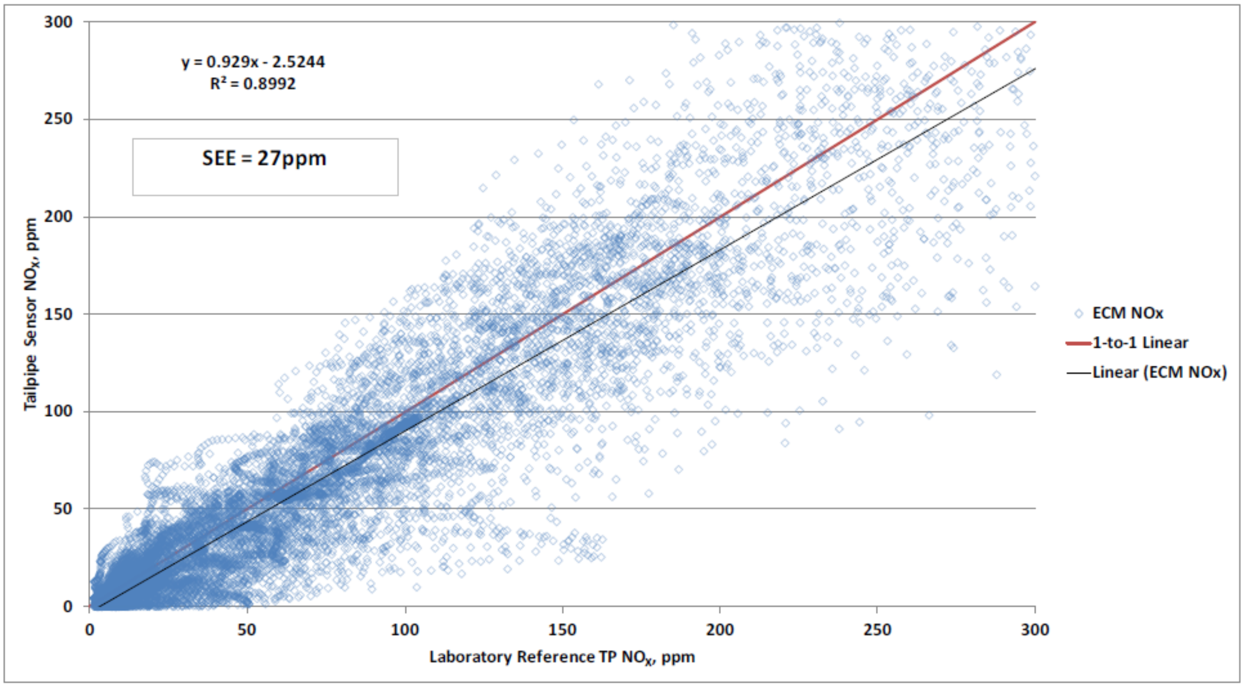


FIGURE 72. EXAMPLE TP NO_X SENSOR VS. LAB REFERENCE TP NO_X CONCENTRATION – LLC AND ARB TRANSIENT CYCLES

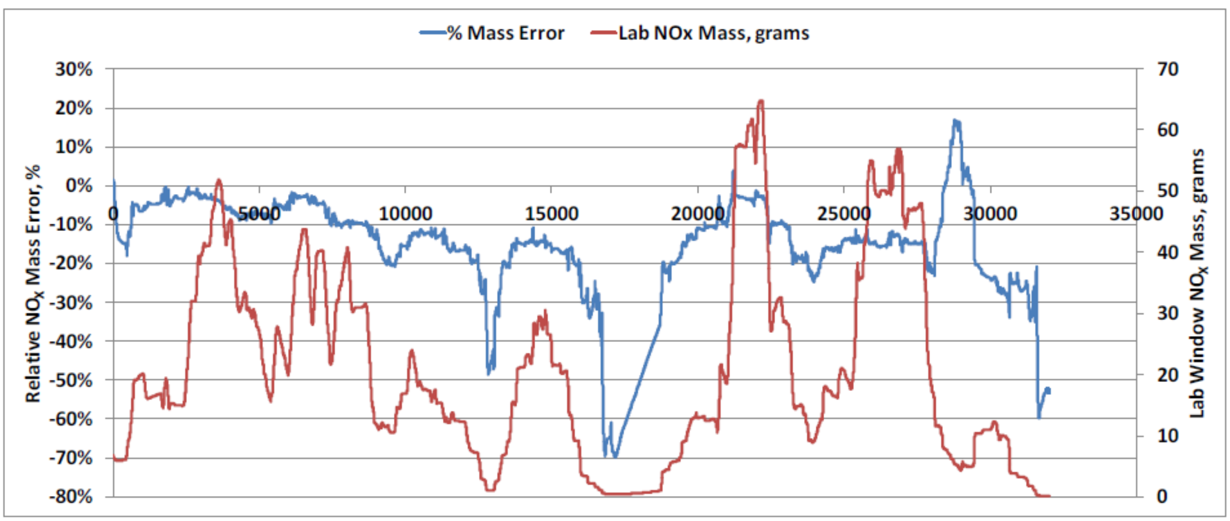


FIGURE 73. NO_X MASS RATE ERROR OVER 20-MINUTE INTEGRATION WINDOWS BASED ON SENSOR VS. LAB CONCENTRATIONS (WITH SAME EXHAUST FLOW)

Even more significant, SwRI also highlighted what WVU's analyses have confirmed: the MAW-based method does not yield any clear trends in emissions behavior, and disproportionally weights brief spikes in NO_x emissions (i.e., NO_x "breakthrough events"). SwRI's multiple observations on those fundamental flaws in CARB's approach bear repeating.

First, SwRI observed that the MAW-based approach "indicates no clear trend [in emissions] other than a high frequency of very low numbers, but the rest of the distribution is *scattered somewhat randomly* between 0.05 and 0.35 g/bhp-hr." (SwRI Report, ISOR Reference 191, p.77.) (Emphasis added.) SwRI also noted that the MAW-based approach "provides little information about where emissions are coming from in terms of engine operating modes." (SwRI Report, ISOR Reference 191, p. 79.) SwRI depicted that overall randomness in the MAW-based emissions data as follows:

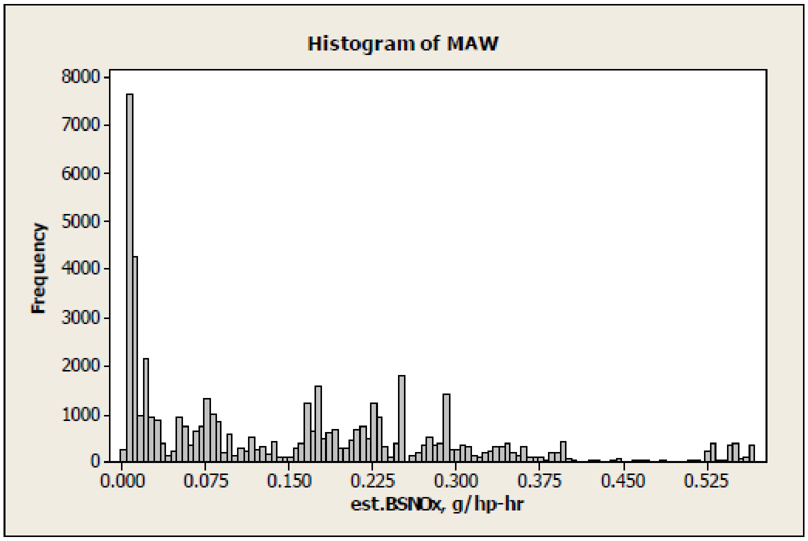


FIGURE 86. HISTOGRAMS OF EST. BSNO$_X$ FROM MAW ANALYSIS

Second, SwRI expressed its clear conclusion (again matching WVU's) that the MAW-based approach tends to overweight "return to service events after a long low-load period," and that CARB's approach "could result in an overemphasis of those relatively brief spikes in a Low NO$_x$ environment," with "a large number of windows being driven by a small number of breakthroughs." (SwRI Report, ISOR Reference 191, pp. 66, 69 and 74.)

In light of that unrepresentative aspect of the MAW-based approach, SwRI recommended an in-use program fundamentally different from what CARB has proposed. SwRI generally described its recommendation, as follows:

> It is suggested that for a more responsive in-use metric, it would likely be more appropriate to regulate in-use compliance based on a distribution [of in-use emission values] rather than a single compliance threshold (such as a conformity factor). Under that scenario, one could potentially regulate the 50th-percentile of the distribution of [MAW-based] results to a value that is near the standard, and then allow the 95th-percentile to float to a significantly higher value (i.e., 5 times higher or even more). This would ensure that the majority of data would be near the desired value, while permitting occasional excursions to higher values. (SwRI Report, ISOR Reference 191, p. 72.)

SwRI also recommended that "careful consideration be given to balance the in-use metric design with the stringency for light-load duty cycles," and that "more effort is needed to examine these and other metrics, and the implications of each approach." (SwRI Report, ISOR Reference 191, pp. xiv, 75.)

In summing up its conclusions regarding CARB's MAW-based approach, SwRI highlighted the facts that NO_x sensors will "require considerable improvement in application and accuracy to support in-use compliance measurements at Low NO_x levels," and that "further investigation of the [in-use] metrics is needed, as well as to set a proper compliance threshold for whichever new metric is chosen." (SwRI Report, ISOR Reference 191, p.88.) Accordingly, SwRI ended its report with the following admonition, agreeing with the conclusions of WVU and EMA: "More analysis needs to be performed before setting a final in-use measurement protocol, and the appropriate compliance thresholds [plural] for that protocol." (SwRI Report, ISOR Reference 191, p. 89.)

Regrettably, CARB has ignored the recommendations and admonitions of its designated technical experts, and is proceeding forward with a 3B-MAW method that is unsound, underdeveloped, untested and unreasonable. The SwRI Report (like WVU's analysis) confirms as much, and lends additional support to the clear conclusion that the proposed Omnibus Regulations are infeasible and invalid.

g. PEMS Cannot Effectively Implement or Permit the Enforcement of the Proposed 3B-MAW Standards

Perhaps even more significant than the flaws inherent in the 3B-MAW protocol, CARB has not demonstrated – and in fact cannot demonstrate — that the portable emissions measurement systems (PEMS) that CARB would rely on to implement and enforce its 3B-MAW in-use testing program are capable of measuring and "binning" NO_x emissions at the near-zero levels that CARB's Omnibus Regulations would require.

The undisputed facts are that current PEMS are not capable of measuring and sorting NO_x emissions at levels as low as 0.030 g/bhp-hr, the low-NO_x levels at which CARB proposes to set the medium/high range bin of the in-use 3B-MAW standards. To the contrary, the regulatory-capable NO_x-detection and measurement range of current PEMS is at a level (approximately 0.20 g/kWh, or 0.15 g/bhp-hr) that is roughly an order of magnitude higher than the in-use NO_x limits that CARB's regulations envision. Indeed, CARB's proposed in-use 3B-MAW low-NO_x standards are close to the measurement "drift" of PEMS' NO_x-detection instruments. CARB has no data and there are no data whatsoever in the rulemaking record that contradict the well-established facts regarding the NO_x-measurement capabilities of current or even future PEMS. In that regard, and most telling, CARB has not conducted <u>any</u> PEMS-based in-use testing of <u>any</u> HDOH vehicle to try to establish the feasibility of its 3B-MAW proposal. Consequently, it is clear that the PEMS-based 3B-MAW in-use testing protocols and standards that CARB is proposing are infeasible and unenforceable, as detailed further below.

CARB has proposed to eliminate any PEMS measurement accuracy adjustment factor for any in-use emissions-compliance testing conducted on MY 2024 and later HDOH engines under the new proposed 3B-MAW protocol. In particular, CARB proposes to eliminate the current in-use measurement allowance for NO_x, which is 0.15 g/bhp-hr. However, CARB has presented no study or evidence whatsoever demonstrating that the PEMS that will be used to conduct the 3B-

MAW-based in-use testing no longer require a measurement allowance to account for the relative accuracy and variability of emissions measurements made with PEMS, as compared with emissions-certification tests conducted in emissions testing laboratories. Similarly, CARB has not produced any data supporting its seemingly arbitrary position that the very same measurement accuracy adjustment factors that CARB's current regulations apply during an NTE-based in-use compliance test, using PEMS, are somehow no longer necessary under the new 3B-MAW-based protocols for in-use emissions-compliance assessment, using PEMS. Rather, CARB simply asserts that the corollary European Union (EU) regulations do not directly apply a measurement accuracy adjustment factor in the EU's MAW-based "In-Service Conformity" requirements. That argument is neither germane nor persuasive.

The EU in-use regulations have important measurement-variability safeguards that CARB's proposed 3B-MAW-based regulations do not. The EU In-Service Conformity regulations established in-use emission standards at a prescribed multiple of the engine dynamometer (dyno)-based certification test standard, a construct similar to what CARB has proposed to utilize to set in-use limits for its 3B-MAW approach. More specifically, the current EU in-use NO_x standard is derived by applying a multiplier of 1.5 (known as the "conformity factor") to the applicable dyno-certification standard of 0.34 g/bhp-hr (0.46g/kWh). The result from applying the conformity factor is that the actual EU in-use NO_x limit is 0.51g/bhp-hr, which is 0.17g/bhp-hr higher than the EU certification test standard. It was agreed during the development of the EU regulation that any in-use PEMS-based NO_x measurement inaccuracies would be absorbed within the 0.17 g/bhp-hr "margin" that the EU regulations provide relative to the test cell standard.

In contrast, under CARB's proposed Omnibus Regulations, where the 2027 MY NO_x limit for the medium/high-power 3B-MAW bin would be set at 0.03 g/bhp-hr, no measurement accuracy adjustment or margin would be applied at all, let alone at the EU level of 0.17 g/bhp-hr, which is roughly equivalent to the current 0.15 g/bhp-hr in-use NO_x measurement allowance provided under the relevant U.S. EPA (and current CARB) regulations. (See 40 CFR § 86.1912 (a)(5)(iii).) Accordingly, there is no technical justification (including from the EU) for CARB's unilateral removal of the current in-use accuracy margin in the absence of any new supporting PEMS-accuracy studies or data, especially considering that the current NTE-based PEMS measurement accuracy allowance for NO_x (0.15g/bhp-hr) is fully five times higher than the proposed 3B-MAW high-power bin in-use NO_x limit of 0.03 g/bhp-hr.

The current PEMS measurement-accuracy adjustment factor was determined in 2008 through an extensive series of tightly controlled laboratory and in-vehicle tests designed specifically for the assessment of PEMS measurement accuracy and variability. CARB was an active participant in the development of that testing program, which was performed at Southwest Research Institute (SwRI), the same research lab that CARB has used to conduct the technology assessment that serves as the principal technical basis for the Omnibus Regulations.[19] There have been no significant technological breakthroughs in PEMS equipment design or capabilities in the

[19] See "Determination of PEMS Measurement Allowances for Gaseous Emissions Regulated Under the Heavy-Duty Engine In-Use Testing Program." SAE Paper, 2009-01-0938/0939/0940, SAE International Journal of Fuels and Lubricants (2009); EPA Report No. EPA 420-R-08-005 (Feb. 2008); EPA, Direct Final Rule, "In-Use Testing for Heavy-Duty Diesel Engines and Vehicles; Emission Measurement Accuracy Margins for Portable Emission Measurement Systems," (73 FR 13441-52, March 13, 2008).

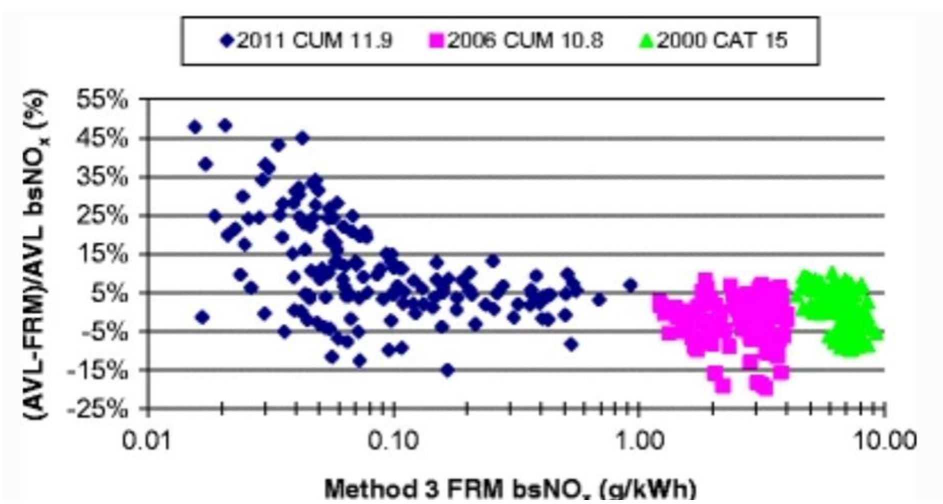
intervening years that would materially improve their emissions-measurement accuracy, including for NO_x.

To put this issue into perspective, today's NTE-based in-use NO_x standard of 0.30 g/bhp-hr (0.45 when the authorized NO_x measurement allowance of 0.15 g/bhp-hr is added on) involves measuring NO_x concentrations on the order of 45 ppm. In comparison, the proposed medium/high load 3B-MAW "Bin 3" in-use NO_x standard of 0.030 g/bhp-hr would require measuring NO_x concentrations of approximately 4 to 5 ppm, or closer to 3 ppm since manufacturers would need to design for some minimum level of compliance margin. Those single-digit ppm levels are equivalent to the "drift" of PEMS NO_x measurements over an 8-hour period, before factoring in any of the actual in-use sources of PEMS' measurement inaccuracy and variability, such as signal noise and interference from other emissions species in the exhaust stream. To be able to tolerate unavoidable NO_x breakthroughs in the medium/high bin, the overall window result would basically have to be zero. Thus, two or three ppm of drift, on its own, would lead to a non-compliant in-use NO_x result.

A 2016 UC-Riverside study of a major HDOH PEMS supplier's state-of the-art equipment is highly relevant to this issue. [20] The study report concludes that,

> "The relative [PEMS] NO_x error increases sharply below 0.1 g/kWh [0.075g/bhp-hr] from 15 % to more than 50 % at 0.02 g/kWh [0.015g/bhp-hr]. The relative error below 0.10 g/kWh is high due to the very low NOx emission rates that approach the detection limit of both the raw PEMS and dilute FRM measurement methods. For the ultra-low NO_x emission level below 0.1 g/kWh, the PEMS started to lose accuracy as the very low NO_x concentrations approach its analyzer drift. The PEMS ability to measure NO_x [at] 0.03 g/kWh (5 ppm raw and 1 ppm dilute) will be a challenge with the latest PEMS and may have uncertainties of approximately 50% at 0.03 g/kWh."

The report's conclusion — that PEMS' accuracy deteriorates significantly as NO_x levels start to fall below 0.20 g/kWh — is amply supported by the data acquired in the underlying UC-Riverside study, which compared the PEMS-reported NO_x levels to those of a Federal Reference Method, as depicted below:



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[20] *A Comprehensive Evaluation of a Gaseous Portable Emissions Measurement System with a Mobile Reference Laboratory*, Tanfeng Cao, et al., University of California Riverside, 2016.

It makes no sense, therefore, for CARB to eliminate an in-use measurement allowance for NO_x when the need for that allowance is far greater at the low-NO_x in-use levels that CARB is proposing. The available data completely refute CARB's unfounded position. Moreover, the UC-Riverside paper also points out that the measurement "drift" that is permitted under the relevant federal and CARB specifications for emissions-measurement equipment (see 40 CFR 1065.550) would equate to a 0.0008 g/bhp-hr drift limit at the low NO_x levels that CARB is targeting, a drift limit that would be difficult even for laboratory grade instruments to meet, let alone PEMS, which as noted above, have drift levels that are roughly equivalent to the proposed 0.030 g/bhp-hr in-use standard. Accordingly, on this basis as well, there is no justification (or even rationale) for CARB's proposal to eliminate the current in-use NO_x measurement allowance. PEMS simply cannot measure what CARB would require them to measure without including the necessary measurement allowance.

Recent input from a major PEMS manufacturer also is instructive on this issue. While there may be some limited avenues for marginally improving the accuracy of NO_x measurements with PEMS, they all involve drawbacks and concessions. Any small accuracy improvements would still be compromised by all of the confounding real-world practical issues associated with in-use testing, such as time alignment, fuel and exhaust-flow estimates, the influences of the high ambient temperatures and high humidity conditions common in the California climate, as well as condensation impacts and even system-freezing during the occasional tests run at ambient temperatures as low as -7°C. Moreover, the PEMS market is an extremely limited one, with PEMS manufacturers only able to make a business case for selling a single model to satisfy the emissions-measurement requirements associated with all relevant business sectors. Creating a unique PEMS model with slightly improved low-level NO_x measurement accuracy (likely at the expense of accuracy when recording NO_x emission "breakthroughs"), to be sold only to those few engine manufacturers that might elect to run the risks of staying in the California heavy-duty truck market, would not present a compelling business case to PEMS manufacturers.

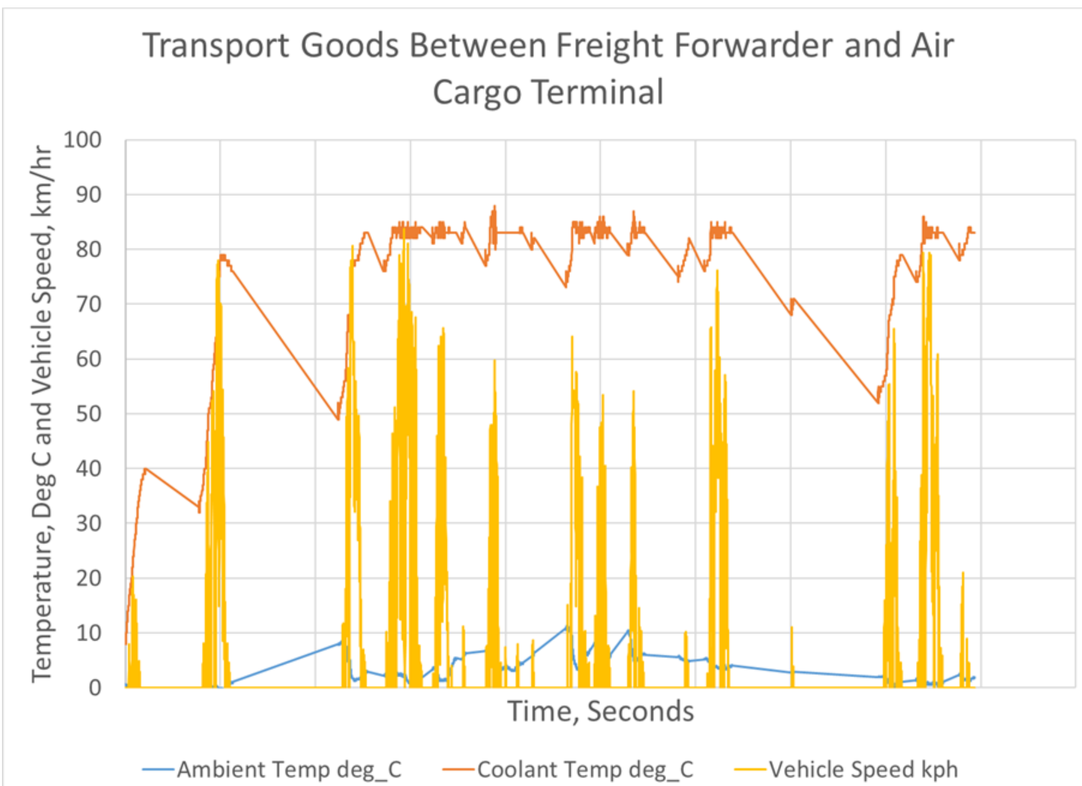
CARB's notion that the in-use measurement accuracy margin can simply be brushed aside is not based on any data or evidence, and runs counter to longstanding scientific research, understanding and practice. As a result, the current additive PEMS NO_x-measurement adjustment factor (0.15 g/bhp-hr) must be retained, as should the measurement allowances for the other emissions constituents as well. Once that necessary concession to the realities of in-use PEMS-based testing is made, it becomes clear that CARB's proposal, in effect to set the in-use NO_x standards significantly below the measurement capabilities of current PEMS (and five times lower than the current NO_x measurement allowance), is fundamentally infeasible. Promulgating emissions standards that are far below the limits of detection for state-of-the-art emissions measurement equipment is neither workable nor reasonable. CARB's 3B-MAW proposal is therefore fundamentally unsound and invalid on this basis as well.

h. Other Issues Undermine the Implementation of CARB's 3B-MAW Proposal

CARB proposes to set the "In-Use Threshold" at "the value of the [dyno test-cycle based] emission standards multiplied by a conformity factor of 1.5 for each of the respective in-use bins: idle, low load, and medium/high load." As just explained, that definition needs to be consistent with the definition applied today concerning NTE testing, which means it needs to include today's allowed measurement accuracy margin.

CARB also has proposed to set the minimum ambient temperature at which compliance with the 3B-MAW in-use standards must be met at -7°C (<20°F). That very low ambient temperature threshold is problematic on multiple levels. First, CARB has based many of its efforts in this rulemaking on the sensitivity of NO_x control technology to exhaust temperatures. Ambient temperatures on the level of -7°C will significantly reduce engine-exhaust temperatures below those under the well-controlled conditions of an emissions-testing laboratory, yet CARB has made no demonstration regarding the feasibility of compliance at such low ambient temperatures. The stability, accuracy, and function of PEMS is questionable at those very low ambient conditions as well. Moreover, ambient temperatures that low are very rare in any populated areas of California. Finally, and perhaps most importantly, no photochemical ozone-producing reactions occur at extreme ambient conditions down to -7°C, so CARB is imposing technology costs, CO_2 control limitations, and compliance risks for no environmental benefit. CARB should increase that minimum ambient temperature criterion to +7°C, which aligns with the temperature below which photochemical smog formation rapidly diminishes.

For the 2024-2026 model year engines, CARB proposes that the in-use test data would not be valid during the period of time after engine-start and before the engine coolant reaches 158°F (70°C). CARB also should consider invalid any data collected at any time when the engine coolant is less than 158°F. It is possible, on cold days and after engine-off periods, that the coolant temperature can drop below the 158F (70°C) threshold, as evidenced by the actual test plot below. The in-use testing procedures need to account for that possibility.



In addition, CARB has identified several factors that can invalidate an in-use test after it is completed. Invalidating a test puts considerable strain on the schedule for in-use testing, and can strain the OEM's relationship with a customer who has cooperated with the manufacturer in supporting the regulatory in-use testing program, if another day of testing is required or another vehicle must be identified for testing. Therefore, tests should only be invalidated where there is actual good cause to do so.

One of the conditions under which CARB has proposed to invalidate a test is if the engine coolant temperature is more than 30°C (86°F). Such a requirement is overly restrictive, especially given the warmer climate conditions typical throughout much of California. For example, consider that requirement in the context of the typical in-use test scheduling process. Should a manufacturer cancel all in-use testing for the day because the engine's coolant temperature failed to drop below 30°C during the course of the evening? What judgment is the in-use test team supposed to use to feel confident enough to start the engine and not witness the flow of warmer water resting in the engine block immediately increase to temperatures >30°C as that water flows past the temperature sensor?

Any such outcome would, by CARB's proposal, render the scheduling, time, resources and inconvenience to the customer's operations for naught if the test were to be declared invalid as CARB proposes. If CARB decides to maintain the requirement not to start the engine before commencing the start of PEMS measurement despite the concerns raised above, CARB should remove the maximum coolant temperature criteria at engine-start, or at least increase it to 50°C to reduce the chances of this kind of wasteful outcome. It adds nothing to the credibility of the test data in terms of assessing compliance to in-use standards.

While CARB also has specified several other conditions under which an in-use test should be invalidated,[21] CARB has failed to identify two key conditions for which a test should be declared invalid. The first condition is if a regeneration event occurs during some portion of the in-use test. In an April 20, 2020 Omnibus Low-NO_x work group meeting, CARB staff noted that that they were following EU regulatory practices that do not provide any special consideration for a test which happens to include a regeneration event. That is not accurate. The European In-Service Conformity regulations do, in fact, permit a manufacturer to void a test that includes a regeneration event. EU VI regulation 582/2011 (introduced in the amendment EC 2016/1718) specifies:

> *4.6.10. If the particle exhaust after-treatment system undergoes a non- continuous regeneration event during the trip or an OBD class A or B malfunction occurs during the test, the manufacturer can request the trip to be voided.…"*

Utilizing test data that includes a regeneration event to assess for compliance with the in-use standard is in direct conflict with the basic concepts of the test-cell certification procedures that involve development of infrequent regeneration adjustment factors ("IRAFs"). Those adjustment factors are used to accommodate the fact that regeneration emissions are characteristically different and generally higher than under normal operation. It is therefore inappropriate to consider an in-use test that includes regeneration as a valid test.

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[21] CARB has incorrectly required that a test be voided if it fails to meet "a minimum valid *window* requirement of 3 hours of non-idle operation." (Emphasis added.) That requirement is inconsistent with the in-use test provisions of §86.1910, which require a minimum of 3 hours of non-idle operation. CARB should amend this proposed requirement to be consistent with the already-codified and well-established in-use testing requirements.

The second condition under which an in-use test should be invalidated is when the malfunction indicator lamp is illuminated during any portion of the test. Under no circumstances, even in the event of passing test results, should CARB consider a test with any period of MIL-ON time to be a valid test.

There is another PEMS-related issue as well, in addition to the inability of currently available PEMS to accurately measure NO_x at the levels of CARB's proposed stringent in-use standards. CARB is requiring that when conducting in-use testing, if the PEMS fails to meet the allowable "range" criteria in §1065.550 for 5% or more of the test intervals, the test engine would be deemed noncompliant unless compliance is nonetheless demonstrated. The problem with such a provision is that emission levels can be high during the period following a cold-start, likely in excess of the levels to which one would otherwise set the range of the PEMS (with an appropriate concentration calibration gas.) To avoid emissions measurements that exceed the range of the calibration gas, the PEMS would have to be calibrated to a higher range than might be sensible for enhancing accuracy at the very low NO_x levels required under the Low-NO_x Regulations, especially for emissions sorted into in the medium/high normalized CO_2 bin. That is exactly the type of unresolved in-use testing issue that can only be understood and addressed through the execution of a carefully controlled PEMS measurement-accuracy program, a program that is clearly needed to assess in a reasonable manner the feasibility of CARB's very strict proposed in-use standards. Without that PEMS evaluation program, CARB simply cannot demonstrate the feasibility of its 3B-MAW proposal.

Another very significant issue (discussed further below) is that CARB has failed to explain how the comprehensive HD OBD requirements will be amended to cover the new 3B-MAW standards. For example, CARB has not demonstrated that all of the OBD-related requisite standards, sensors, software, post-processing protocols, and similar elements needed to comply with the 3B-MAW requirements will be in place by the 2024 MY.

i. CARB Has Not Shown that the 3B-MAW Proposal Constitutes a Credible Protocol for Assessing In-Use Emissions Compliance

Perhaps most disconcerting of all, over and above the numerous serious concerns discussed above, is the lack of technical rigor and scientifically-based judgment that CARB has put into the development and "validation" of the 3B-MAW protocol as a credible means for assessing in-use emissions compliance. More specifically, CARB has not made any demonstration of any kind that:

(i) The proposed emissions "bin" definitions and boundaries reasonably and consistently segregate similar emissions characteristics in a manner that appropriately reflects the varying operating conditions of the engine and vehicle;

(ii) The moving average window approach is superior to binning data without windowing, and that 300 seconds is the appropriate duration for a measurement window;

(iii) There is a direct scalable relationship (i.e., 1.5x) between the emissions sorted into each bin, and the underlying test-cell standards (as CARB has linked them), which are based on different certification cycles;

(iv) Day-to-day emissions levels from a single in-use test article tested over the same route produce reasonably repeatable 3B-MAW results;

(v) Day-to-day emissions levels from a single in-use test article run over highly variable test routes produce similar 3B-MAW results in each bin --- a minimum expectation for a tool that should be able to discern compliant vehicles from non-compliant vehicles;

(vi) SwRI's "Stage 3" prototype engine, when installed in a variety of vehicles and operated over a variety of duty cycles, is capable of meeting the proposed 3B-MAW in-use standards on a consistent basis;

(vii) The 3B-MAW protocol can reliably take into consideration the transient operating characteristics of an engine over a given route segment that lead to variations in core SCR temperature, and therefore is reflective of tailpipe emissions levels;

(viii) The proposed 3B-MAW in-use standards are achievable over the allowable range of ambient conditions for a valid in-use test using the Stage 3 technology set; and

(ix) There are alternative technology options different from the Stage 3 prototype that are capable of meeting the 3B-MAW standards, in the event that the Stage 3 technologies (including CDA) cannot withstand the rigors of heavy-duty in-use applications, or cannot be packaged for installation in heavy-duty vehicles.

In the end, CARB has based the feasibility of the entire 3B-MAW program on a single test-cell evaluation[22] of a single technology set, when tested over a single test route (the CARB "Southern Route"), and using a seemingly arbitrary "1.5 times" multiplier as a link to the test-cell certification-cycle emissions performance of that technology. CARB has not presented any data demonstrating the appropriateness of the bin definitions through any parametric study, nor through a comparison against alternative criteria and methods to set bin boundaries. Nor has CARB presented any assessment of why a 300 second window is optimum or even appropriate for the 3B-MAW approach, let alone how overlapping windows are superior as a compliance methodology to simply binning second-by-second results. And, as highlighted above, CARB has conducted no in-use testing whatsoever of its new in-use testing protocol.

Given CARB's unreasonable lack of due diligence in this regard, manufacturers would be left to face insurmountable technical challenges to achieve extremely low in-use emissions levels, over a brand new and utterly undemonstrated in-use testing protocol using technologies never

[22] Significantly, the calibration for that feasibility test was modified after the first test run to mitigate NO_x breakthroughs that were occurring, thereby improving the reported emissions results over the cycle.

before deployed in a heavy-duty vehicle. For all of the foregoing reasons, therefore, the proposed 3B-MAW protocol and standards are, in effect, arbitrary, unreasonable, and invalid.

9. CARB's Proposed Strict Liability Proposal for its Heavy-Duty In-Use Testing (HDIUT) Program is Unfair

CARB also is proposing to convert the current HDIUT regulations into a strict liability program. (See revised §86.1915 (B.1).) Under CARB's unilateral re-write of the HDIUT program, "failures" of the "Phase 1" HDIUT procedures (where 5 out of 5, 5 out of 6, or 8 out of 10 vehicles need to "pass" the NTE-based in-use compliance metrics) would be sufficient on their own to support a finding of "noncomformity" or "noncompliance," and thus sufficient for CARB to compel an HDOH engine family recall.

CARB's proposed unilateral amendment of the HDIUT program is manifestly unfair and would impose unreasonable risks of recall liability on manufacturers. The HDIUT program (codified at 40 CFR Part 86, Subpart T, §§86.1901-86.1935) is a program that resulted from a negotiated settlement of litigation that EMA filed in 2001 challenging CARB's and EPA's authority to require that manufacturers test previously-sold non-new vehicles no longer in the manufacturers' possession and control. (See 70 FR at 34597.) As a result of a duly negotiated and approved settlement agreement between CARB, EPA, EMA and manufacturers (which settlement was subject to a thorough public notice and comment process), the parties developed and specified the terms of the HDIUT program. (Id., n.2.)

As negotiated and agreed, the current NTE-based HDIUT program does not compel recall or other noncompliance liability solely on the basis of an engine family failing to meet the engine family "pass" critieria (where 5 out of 5, 5 out of 6, or 8 out of 10 in-use vehicles pass) as tested under "Phase 1" of the program. Instead, under the current negotiated regulations, the Agencies enter into further discussions with the manufacturers regarding the extent of any appropriate follow-up steps, which steps can include no further testing, additional targeted "Phase 2" testing, engineering studies, or, if deemed necessary, targeted remedial actions. The core concept is that any initial "failure" of Phase 1 testing is simply a trigger for further discussions and assessments, not a trigger for strict noncompliance liability. (See 70 FR at pp. 34595-96, 34598 and 34601.)

In light of the foregoing, CARB's unilateral move to create a strict liability HDIUT program — with automatic recall liability for any "failed" Phase 1 testing — is contrary to the foundational agreements and terms that created the HDIUT program, and will result in an unfair and unacceptable divergence between the federal HDIUT program and the revised program that CARB seeks to implement. CARB's unilateral imposition of new and unwarranted in-use compliance risks and liabilities is yet another aspect of CARB's Omnibus Regulations that likely will fracture the market for HDOH products, with several manufacturers being forced to exit California.

10. CARB's Additional Changes with Respect to In-Use Testing Practices are Unworkable

CARB staff held a Low-NO_x working group web meeting on April 20, 2020, during which they presented, among other things, additional aspects of their proposed amended 3B-MAW HDIUT program. Among those new requirements, CARB proposed new pass/fail conditions for an in-use test order. More specifically, CARB proposed that a family "failure" determination could be made on two bases. First, the family would not pass if "there are three [or more] exceedances of the same bin and same pollutant," and second, the family would not pass if "the arithmetic mean of the sum-over-sum emissions from the 10 vehicle tests is greater than the in-use thresholds for any pollutant in any bin." When a participant in the web meeting questioned the extremely restrictive nature of the latter condition for engine family failure, especially where one or two vehicles may have a compromised SCR systems (due to fuel contamination, etc.), CARB staff responded that they would anticipate that a compliant OBD system would catch such failures and would have screened-out any such vehicle from the test program.

That response from CARB is inaccurate. Consider the following hypothetical test results from ten vehicles tested to satisfy an in-use test order:

BIN 3, NO_x

FTP/RMC standard	In-use standard	OBD Threshold
0.020	**0.030**	**0.40**
Vehicle 1	0.020	
Vehicle 2	0.020	
Vehicle 3	0.080	(20% of OBD threshold)
Vehicle 4	0.020	
Vehicle 5	0.080	(20% of OBD threshold)
Vehicle 6	0.020	
Vehicle 7	0.020	
Vehicle 8	0.020	
Vehicle 9	0.020	
Vehicle 10	0.020	
Average	**0.031**	**FAIL**

In the case of this hypothetical example, 8 out of 10 vehicles comfortably pass the proposed 0.030 g/bhp-hr "Bin 3" NO_x standard. In fact, they meet the underlying FTP/RMC test cell standard without application of the 1.5x conformity factor applied for in-use. Just two vehicles (vehicles 3 and 5 in this example) exceed the in-use standard, but at levels of just 20% of the proposed OBD threshold for NO_x. Those vehicles are far from triggering an OBD MIL, so they presumably would not be excluded from an in-use test order. (This would be true even if it were technically possible to reduce the OBD NO_x thresolds to 0.10 g/bhp-hr.) This example demonstrates that CARB's restrictive pass/fail criteria are overly punitive. Consequently, the proposed secondary pass/fail criteria should be eliminated from consideration.

CARB has proposed other changes to the requirements associated with the HDIUT program as well. Those changes are not reasonable. The recruiting, planning and execution of in-use PEMS tests on customer-owned vehicles — tests conducted in the midst of customer operations — are complex and challenging tasks. CARB's proposed changes will bring additional complexity and delays to the program, with little benefit. Some of the proposed changes are simply impossible to fulfill given the normal routines of setting up and executing the requirements of the program.

As a matter of background, a recognized goal of the in-use testing program is to obtain emissions data from representative customer trucks, which should include a variety of different types of customer applications. It has been OEM policy to make sure there is little or no manufacturer influence on the final down-selection of specific routes and trucks tested according to an in-use test order once the candidate set of vehicles has been identified. A list of customer trucks is pre-evaluated for proper maintenance/repairs and mileage, and the customer then designates from that list the test vehicle to be utilized for a specific test day.

Because fleets, small trucking firms, and various vocational service companies must cater to the many different requirements of *their* customers, each trucking operation has different policies and procedures concerning their scheduling and operations. For example, some have their trailers packed and attached to the tractors during the night, waiting and ready for the driver to arrive, whereas others must load the trailer on a more irregular or "as available" basis. Many tanker-trucks rarely disconnect from the tractor. Many trucking operations are changing their routes/schedules on a minute-by-minute basis, whereas others have schedules planned weeks in advance, but always subject to disruption. So, when an OEM approaches a fleet or other operation about conducting in-use testing with their vehicles, some customers may be able to provide the detailed information requested by CARB reasonably far ahead of the testing date, but many cannot. After 15 years of in-use testing experience industrywide, OEMs have learned to expect that the truck, driver, and route are often different from what was originally planned in consultation with the fleet manager.

CARB has decided to more deeply engage in, and require CARB approvals for, numerous aspects of manufacturers' in-use test planning. To that end, CARB has introduced a long list of newly required information over and above that which is required by EPA (see e.g., CARB's proposed modifications to §86.1920(h)). Much of the information that CARB requests 30 days in advance of a fleet-test includes items that will be unknown until the manufacturer's test team arrives at the fleet-customers' location. Current practice is to select customers who have at least six units from the specified engine family, and to make the final vehicle selection based on availability once the team arrives at the customer location. PEMS testing is disruptive to a customer's operations and the test vehicles do not belong to the engine manufacturer, so the test team must be flexible in terms of vehicle selection and scheduling. Loads are being dispatched in real-time in a dynamic environment where the trucking company is trying to minimize down-time to meet its customers' needs. Most fleets will not know the availability of specific units 30 days out. In some cases fleets will not know vehicle availability as little as 24 hours out. Recruiting customers for PEMS testing is already extremely difficult. CARB's new requirements will make it nearly impossible.

More specifically, 16 of the 30 newly-designated data elements CARB would require 30 days before commencement of in-use testing are simply unknowable in that timeframe. For example, the customer vehicle-vocation is known only if working with a fleet that performs a single vocation. Similarly, specific PEMS unit details are difficult to identify because they are often deployed in rotation based on testing and calibration schedules. In fact, PEMS' certification for linear verification lasts only 35 days. That means that even if a PEMS unit were identifiable 30 days in advance of testing, the certification would very often have to be updated based on a required recalibration before testing could begin. Indeed, the only practical data elements CARB proposes to require that could reliably be provided in advance according to CARB's schedule are the engine family designation, engine displacement, and the date on which CARB selected the engine family for testing – all data elements that CARB dictated to the manufacturer when issuing the test order in the first place.

There are still other complications raised by CARB's new proposed in-use testing requirements, including the requirement to include a cold-start. Conducting PEMS tests is very different from conducting test-cell tests. In the test-cell environment, nearly all measurement equipment can be connected and verified prior to starting the test. Test cells are not reliant on signals from the engine controller, such as those required to measure exhaust flow and fuel flow. In a test cell, measurement systems can be verified independently, without interaction with the test article, before engine start. Test cell equipment and functionality also benefit from not being removed from the test cell and test article, and re-installed for every test. That is not the case with PEMS testing.

When conducting in-use testing with PEMS, each test is similar to a test-cell installation and commissioning exercise. With that tremendous complexity, plus the dependency on new controller connections for each PEMS test, it often takes a number of attempts to get all of the systems working reliably. Re-initialization of data communication is often necessary because of engine shutdowns and the reliance on engine control module data (again, not necessary in the test cell environment). Those J1939 communication initializations often cause issues during PEMS testing. What all this means is that there is a high risk, under the requirements CARB has proposed, of a test being declared invalid due to equipment malfunction during a cold-start. The consequence of that outcome is that testing would have to be rescheduled for another day, with the very real possibility that the customer would not be able to accommodate the extended request during the course of the test team's travel itinerary. That also can damage the good will that helped in recruiting the fleet customer and vehicle in the first place. CARB should eliminate the requirement that each in-use test include a cold-start in order to be counted as a valid test.

11. <u>The EWIR Amendments Will Create Unreasonable Liabilities and Costs</u>

The Omnibus Regulations also include the adoption of regulatory amendments to transform the current emissions warranty information reporting (EWIR) requirements into a strict liability program. More specifically, under the contemplated amendments, any exceedance of the "screened" 4% warranty claims-rate threshold for emissions-related components would trigger either extended warranties for the parts at issue or mandatory recalls, or both, without regard to the potential emissions impacts that might be related to the emissions-related components and warranty claims at issue. That strict liability program and the need for corrective action would "be based solely on warranty failure rates." (Appendix 2, p.4.) While CARB does not attempt to quantify the aggregate costs of moving to that type of a strict liability EWIR program, those costs could easily amount to tens of millions of dollars for individual manufacturers, and likely would prove to be cost-prohibitive, especially considering the limited (if any) corresponding monetized health benefits. (<u>See</u> Sections 2-4, above.) In cases where CARB proposes to require both a recall and extended warranties, the manufacturer would be doubly penalized. Indeed, CARB understands that its proposed changes to the EWIR program will cause substantially increased EWIR claims and corrective actions (Notice, p. 17).

The EWIR provisions as proposed can have far-reaching effects. If a part reaches the 4% failure rate, for example, and a recall is required despite the lack of an emissions increase, the remaining 96% of the vehicles equipped with that part must be removed from the road for some period of time, depriving the owner of its ability to haul goods or do work, and interfering with the operations of that trucking company and its contracts with customers. The proposed EWIR provisions also would cause a substantial increase in required parts-manufacturing, which will increase emissions, as will the transportation of the new parts to the company warehouse, and then the shipping of those parts to all dealerships. Using a “fix-when-it-fails” approach and covering the customer cost with an extended warranty is far better for the environment than recalling 96% of the vehicles that are working properly at that time.

Similarly unreasonable is the proposed requirement to report warranty claims during any extended warranty offered by an OEM, including when the tracking and reporting would have to continue out to FUL, which would cover as many as 12 to 13 years. The requirement for reporting through any OEM-extended warranty periods would penalize OEMs that offer extended warranties by increasing the reporting period and also increasing the chance that an OEM would reach the 1% level, or more importantly, the 4% field-action requirement level. Extending the period of warranty coverage for customers should not be penalized. The warranty reporting requirement should conclude at the end of CARB’s proposed longer regulatory warranty periods.

CARB also proposes to extend warranties to FUL for any parts replaced through a recall program, and to require reporting on the replaced components through FUL. When a recall is mandated under CARB regulations, the OEM in effect commits millions of dollars to fix a part on up to 96% vehicles that have no evidence of excess emissions. As mentioned, requiring both a recall and extended warranties is doubly punitive already, but also adding the reporting requirement for warranty claims on the replacement part could add one to five years of additional burden on OEMs that have already committed substantial staff and capital to resolving a 4% failure-rate issue. The additional requirement to monitor and report to that extent is extreme and unreasonable.

CARB is seeking to make other changes to the EWIR program as well. CARB is proposing to define "Emission Warranty Claim" as meaning “…an adjustment, inspection, repair or replacement of a specific emission-related component within the statutory warranty period for which the vehicle or engine manufacturer is invoiced.” That definition is overly-broad. Inspection of a component does not imply any type of failure if there is no issue found upon inspection. Moreover, the adjustment of an emissions-related component as a matter of routine maintenance, where the original setting is not found to be outside manufacturers’ allowable settings, should not constitute a failure. Finally, manufacturers will sometimes perform replacements of certain components without evidence of failure as a measure of goodwill for customers. CARB should limit the definition of Emissions Warranty Claim to remove those types of cases from the scope of the definition.

In addition, CARB’s proposed definition of “Emissions Related Component” includes not only components that (1) affect regulated emissions and (2) illuminate the MIL, but also includes any component that “is part of the configuration of a California certified heavy-duty diesel or Otto-cycle engine, or heavy-duty vehicle.” While the term “configuration” is confusing (should this be “certified configuration”?), the quoted section of the definition also appears to be redundant.

Would any component that is either affecting emissions (and on the regulated ERC list) or that illuminates a MIL not already be "part of a [certified] configuration…"? CARB should remove this unnecessary and redundant element from the definition of "Emissions Related Component" to avoid confusion.

CARB's definition of "Extended Warranty" references a time period that is "at a minimum equal to or more than the applicable certified useful life period of that vehicle or engine." CARB does not have the authority to extend warranties beyond the statutory useful life period and so should consider that useful life period as the *maximum* extended warranty period, not the minimum.

CARB's definition of "Systemic Failure" is stated as "any emissions-control component…found to have valid failures that exceed the thresholds in §2143." That definition is inaccurate, as it fails to account for the case where a component may have more than one failure mode. A single particular failure mode that exceeds the §2143 thresholds should be what constitutes a "systemic failure." The same argument applies to the corollary provisions of §§ 2167 and 2168, which should be revised accordingly.[23]

Additionally, warranty claims and component failures can be the result of upstream failures or system performance issues. In those cases where a root-cause investigation determines that the failure is actually caused by an upstream issue, CARB should not compel corrective action for the downstream component.

Another unreasonable element of the proposed requirements is that CARB would require a corrective action plan within 90 days of exceeding the corrective action threshold, including root-cause analysis (§ 2169).[24] Inasmuch as the threshold could be reached with as few as 25 failed components, 90 days is inadequate time to determine a root cause, define a solution, verify its effectiveness, plan the tooling changes needed, verify and release the software changes, and plan the procurement of a sufficient stock of parts to allow the recall to proceed. The required timing is therefore wholly unreasonable and unworkable. Having an initial discussion with CARB within 90 days of reaching the corrective action threshold for a potential recall may be appropriate. Having all of the data required in § 2169, however, is absolutely not reasonable within the proposed 90-day period.

Another burdensome and unreasonable EWIR-related proposal specifies that if a manufacturer amends a Field Information Report ("FIR") by adjusting the number or percentage of failures, it must be done on the basis of an analysis of a new set of components. Often, the reason for amending the FIR is because the *population* of engines with that component has changed, typically due to new information or additional vehicles being sold into or out of California. There is no basis in such a case for an examination of "new parts." The exercise would be wasteful. CARB should remove that provision from any Final Rule.

[23] The definition of "on-board computer" is overly broad, and should be limited to a "computer" that monitors and/or controls five or more sensors, systems or actuators.

[24] The regulation should clarify that thresholds are not "met" if the determination is the result of a rounding-up of the calculated failure rate.

CARB also is seeking to impose requirements on OEMs to retain failed components for a minimum of two years following the submittal of an FIR. That proposed requirement is problematic for several reasons. The 2022MY to 2026MY emissions warranty requirement is 5 years/350,000 miles. CARB has estimated that this could require the retention of 70 component parts for each FIR. FIRs are filed by part number, by engine family, and by model year. If we make the simple assumption that the average FIR is filed 3 years after the build model year, and that the accumulation of parts (by part number, by family and by model year) is 15 parts in Year 1, 40 parts in Y2, and 70 parts in Y3 through Y5, the average number of parts in storage per part number, per family, per model year, is 53 parts ((15 + 40 + 70 + 70 + 70)/5). Among the 30 or so emissions related component (ERC) part numbers on an engine, not all ERC part numbers will have FIRs filed. If we assume FIRs are filed for 10 out of 30 ERC part numbers, and another 10 ERCs only reach one-half the FIR threshold, and that the final 10 of 30 total ERCs have no failure issues, then the average accumulation of parts in storage for all ERCs per family, per model year, is 800 parts (10x53 + 10x0.5*53 + 10*0). If we further assume that an OEM has 3 engine families, and multiply that number by the 5 overlapping model years of "average" storage requirements, the typical required quantity of parts in storage at any point in time could be 12,000 parts (800 x 3 x 5). Some OEMs have estimated parts-storage requirements much higher than this estimation based on their reading of the proposed regulations. In any case, these estimated numbers are expected to grow substantially as additional ERCs are added to the engine-systems to comply with the proposed Low-NO_x standards.

Current warranty processes do not lend themselves to retaining failed parts in this unreasonable way. Very often, failed components are sent to suppliers for analysis. Sometimes the fault investigation involves destroying the failed component. Even where parts are retained, there would be no benefit to holding 50 components with exactly the same failed condition.

There also are a number of concerns regarding CARB's interest in having the parts sent to CARB facilities. These are components that the OEM has openly declared through the FIR process to be failed parts. If CARB anticipates performing additional inspection or analysis of failed components, are manufacturers to expect that CARB will be second-guessing the conclusions drawn by OEMs' technical experts and suppliers? Does CARB expect to be able to draw better, more accurate conclusions from its own component analysis, especially without the benefit of drawings, specifications, test rigs, supplier interaction, and the extensive history that OEM specialists have gained with those components during the course of their development? Moreover, parts that may have sat on a shelf for up to seven years may have undergone degradation, including due to the effects of corrosion, that could lead to incorrect conclusions by CARB personnel in any follow-on inspections.

In sum, the requirement to catalog failed parts, and to store them for extended periods in large quantities, would be extremely costly and time consuming, with little or no corresponding value. Those provisions related to the storage of failed components should be eliminated from the Omnibus Regulations, and a much more practical solution should be identified.

12. **CARB's Push for Strict Liability and Unilateral Enforcement Authority is Not Justified**

Included in the proposed Omnibus Regulations are a number of new provisions that would expand CARB's authority to impose an increasing array of penalties and recall-related liabilities on engine and vehicle manufacturers. CARB's unfair and unjustified proposed conversion of the HDIUT program into a strict liability recall-oriented program is discussed above.

CARB also is proposing to convert its EWIR program into a strict liability program. Currently, if the warranty claims rate for an emissions-related part exceeds the EWIR threshold for "screened" (i.e., presumably valid) warranty claims, manufacturers can avoid any vehicle/engine recall or extended warranty liability by demonstrating that vehicles and engines operating with the potentially defective component part still achieve emission levels that, on average, comply with the applicable emission standards. CARB is proposing to do away with that affirmative defense, and instead seeks to mandate that "corrective action is required based solely on whether the failure rates of emissions-related components meet or exceed the EWIR corrective action thresholds," without regard to the potential emissions impact of the potentially defective component part.[25]

CARB's "justification" for its unilateral imposition of EWIR-related strict liability boils down to its assertion that having to demonstrate a meaningful emissions impact from a potentially defective component part "has required CARB to expend excessive resources and unduly limited both the scope and timing or recalls." (ISOR, p. III-66.) Based solely on that "justification," CARB proposes to shift all of the attendant costs of substantially expanded recall liability onto manufacturers, even in cases where no material adverse emissions consequences could result. The magnitude of the resultant costs to manufacturers – which will be passed along to vehicle owners through proportionally increased purchase prices – makes this element of the Omnibus Regulations, among others, cost-prohibitive and unreasonable.

The EWIR change at issue, the proposed conversion to a strict liability program, is being done solely as a matter of convenience for CARB, without any real regard to the cost impacts on manufacturers and vehicle purchasers. Rather, CARB simply claims that "currently, identifying potentially defective emission control components by warranty reporting requirements, and the process of negotiating corrective action with manufacturers and determining the emissions impact of a component failure is lengthy," and that CARB does not want to deal anymore with having to assess whether any emissions impacts are at issue. "Hence, amendments to the current EWIR requirements are needed to make it easier for CARB to force recalls." (ISOR, p. ES-7; II-19; II-20.)

Objecting to having to discern whether emissions-related components with higher warranty claims rates could actually impact emissions performance "because it can be a lengthy process" is an insufficient justification for imposing tens of millions of dollars of costs per manufacturer. Indeed, making recalls "easier" for CARB is not, by itself, a sufficient basis for a rulemaking, nor

[25] Despite CARB's proposal to ignore emissions impact when a failure exceeds the recall or extended warranty threshold, CARB would still require manufacturers to provide estimations and available data regarding potential emissions impacts (§2146 (c)(7)). If CARB finalizes the Omnibus Regulations to allow for corrective actions without regard to emissions impact, the requirement for manufacturers to provide estimations of those impacts should be eliminated as well.

is it equivalent to making the requisite showing of cost-effectiveness. The cost-prohibitive nature of the Omnibus Regulations is detailed elsewhere in these comments and in the independent expert report that ACT Research and NREL have submitted.

The concerns relating to CARB's strict-liability approach to EWIR-related issues are heightened due to CARB's push to extend the emissions warranty provisions in 2027 and again in 2031 (up to 10 years and 600,000 miles) to "California-certified 2027 and subsequent model heavy-duty vehicles, *regardless of whether they are registered in California*." (Emphasis added). CARB does not have the authority to burden interstate commerce to such an extent, especially for vehicles registered and operated outside the borders of the State of California.

Another example of CARB's push toward unilaterally expanded enforcement authority is found in CARB's proposal to reject manufacturers' "good engineering judgement" whenever CARB staff determines, presumably based on their own subjective assessments, "that a different decision would reflect a better exercise of good engineering judgement." (See Proposed Regulation §2141(f)(4)(D)(2).) The potential ramifications of that new, largely unfettered authority are both sweeping and fundamentally disruptive of the regulatory paradigm that has existed on a nationwide basis for decades, where manufacturers' good engineering judgement is an accepted criterion for multiple testing and certification-related requirements. For CARB to seek to claim unto itself the sole authority to determine in all cases what might be "a better judgement" could completely undermine the orderly implementation of critical well-established certification protocols and practices. CARB should abandon that additional effort toward unilateral and largely unbridled enforcement authority.

To the extent that CARB remains set on questioning what constitutes a manufacturer's good engineering judgement, CARB should clarify that: (i) the proposed provisions in §2141(f) apply only to the implementation of the EWIR regulations, and not generally across all of CARB's HDOH regulations; (ii) any decision to reject a manufacturer's good engineering judgement will be applied on a prospective basis only, and not retroactively to assess liability after the fact; and (iii) in those instances where CARB determines to reject a decision that a manufacturer has made using good engineering judgement, the manufacturer will have the right to challenge CARB's determination in proceedings held before an administrative law judge. Finally, if CARB elects to proceed with this regulatory shift away from the established principles of good engineering judgment, CARB should provide clear examples of the types of cases where, in CARB's view, it would be appropriate for CARB staff to substitute their good engineering judgement for the manufacturer's. Without those clear examples, this regulatory revision could quickly become arbitrary and unreasonable.

Similarly unreasonable and unjustified is CARB's proposal to eliminate subsection (e) of 40 CFR 1068.5. That regulatory provision expressly allows manufacturers to request an administrative hearing if the manufacturer disagrees with the agency's determination to reject a manufacturer's application of good engineering judgement. CARB proposes to strip away that basic element of due process and to create a new power for itself to act as the sole arbiter of what is and what is not good engineering judgement. CARB should refrain from assuming that role as it would be manifestly unfair, violative of basic due process rights, and fundamentally inconsistent with the manner in which EPA administers the parallel provisions of the corresponding federal regulations. There is no justification for CARB's proposal to eliminate administrative due process.

As noted, it remains to be seen whether manufacturers will continue to be able to remain in the California HDOH market under CARB's enhanced strict-liability orientation, especially when coupled with the extreme costs and product-development burdens associated with the other multiple elements of the Omnibus Regulations.

13. <u>The Durability Demonstration Program is Overly Onerous and Time Consuming</u>

CARB's Omnibus Regulations would significantly change and lengthen the aging protocol for the determination of deterioration factors ("DFs") by requiring FUL aging. That is a dramatic change from the current practice of aging between 35% and 50% of FUL. The proposed Regulations would allow a somewhat shorter alternative demonstration period, but a manufacturer would have to submit sensor-based emissions field data to utilize the shorter DF testing routines.

For the 2024-2026 MYs, if a HHD engine manufacturer uses the longer "Option 1" approach of 9,800 hours of dyno-aging, including periodic emissions tests, such a DF demonstration test would require approximately two calendar years to complete. If a HHD engine manufacturer uses the alternative "Option 2" approach of 4,900 hours dyno-aging, followed by accelerated aftertreatment bench-aging (for example, 500 hours), including periodic emissions tests, such a test would require more than one calendar year for the DF demonstration. For the 2031 MY and later, a full dyno-based aging process would last 18,000 hours, or four years.

CARB's proposed lengthening of the DF process will compel manufacturers to map-out their product-development projects and timelines in a very different way. DF tests are typically conducted using components from production-like tooling, and generally having design and materials characteristics consistent with manufacturers' final production intent. The calendar time that would be consumed by the greatly expanded DF testing would force manufacturers to freeze designs much earlier in the development cycle (for example, DF testing would need to start at least one year earlier), limiting manufacturers' ability to get the best possible technical solutions in place, and further exacerbating the leadtime concerns EMA has already highlighted. CARB should permit manufacturers to use accelerated aging cycles to reduce the total calendar time that otherwise will be consumed by DF testing.

The aging cycles that CARB is proposing involve intervals of testing over the FTP, RMC, LLC, CARB Transient Cycle, 55-mph cycle and 65-mph cycle (the last three cycles are elements of EPA's and CARB's heavy-duty greenhouse gas regulations). Some of those cycles include periods where the engine is operated in the "motoring" condition, which means that the dynamometer would provide power to turn the engine at various speeds under "zero" fueling conditions. Running those cycles would require a dynamometer with motoring capability. Most manufacturers use their engine-durability test cells to accumulate engine hours during the aging phases of DF testing, and those test cells are generally not equipped with the more costly dynamometers capable of operating with the engine in a motoring condition. Upgrading test cells to have that motoring capability would cost manufacturers approximately $1.5 million per test cell, and perhaps double that figure if electrical supply upgrades would be needed as well. That is an unreasonable expense that would be added to the already unreasonably expensive DF test requirements. In addition, aging cycles are used for more than just DF testing. There are aging cycles used to fulfill OBD requirements as well, which could lead to upgrade requirements for more than just a single test cell at an OEM's lab. CARB should permit manufacturers to conduct the required aging cycles without any motoring requirements.

CARB's proposed optional accelerated aftertreatment aging process is largely undefined in the draft Omnibus Regulations, other than through the statement that good engineering judgment must be used to determine thermal and chemical degradation, and that the aging process must equal 50% of FUL (for the 2024-2026 model years, with a greater percentage for future model years) for the aftertreatment system, using the same aging cycles used in the test cell with the DF engine. EMA has been working with CARB and EPA on the development of a more cost-effective accelerated aftertreatment-aging protocol, including verification testing to demonstrate the validity of the rapid-aging procedure. EPA and CARB should work expeditiously to get that much-needed development work underway, so that it can be utilized for MY 2024 and later engine families. Importantly, to the extent that CARB continues forward unilaterally to develop unique and far more onerous California-only DF demonstration requirements, out of sync with EPA's requirements, that significant misalignment will stand as another compelling reason for HD truck manufacturers to exit the California market.

CARB's DF demonstration proposal also does not address the question of whether multiplicative or additive deterioration factors should be applied. The draft regulation text only directs that additive or multiplicative factors must be calculated for 2024 and subsequent model years, but gives no indication of the criteria that CARB would use in deciding whether to accept a manufacturer's proposal to apply one or the other. When considering the extremely low-NO_x levels that CARB is proposing, multiplicative DFs would pose serious challenges for manufacturers given the degree of measurement variability that will occur during emissions testing. There is a material risk of falsely projecting unduly high and inaccurate DF factors for NO_x if multiplicative DFs are applied. Multiplicative DFs of 2.0 or greater may be common, especially for NO_x pollutants. CARB should eliminate that undue and unfair risk by working collaboratively with EMA on a guidance document to address the appropriate application of additive deterioration factors to assess compliance with the low-NO_x standards.

CARB also is defining two different FUL NO_x stringencies, one for 2027 to 2030 model years, and one for 2031 and later model years. For a HHD engine, one standard will apply up to 435,000 miles of useful life, and a second, higher standard for the remainder of the fully-extended FUL. The Omnibus Regulations do not address this "two-stage" aspect of the NO_x standards as it relates to the determination of the DFs for NO_x. CARB should include clear DF testing and application requirements as applied to the proposed two-stage FUL standards.

There also appears to be no provision in the Omnibus Regulations for carry-across DFs. The draft Omnibus Regulations state that aging under "Cycle 1" or "Cycle 2" must be assessed for each engine family, and that the one with the highest load factor must be used. It is unclear from the draft language if that truly requires a DF for each engine family, or if this is just the assessment methodology that must be used for the selected DF engine for a DF group. CARB should clearly indicate that carry-across DFs are permitted. That long-standing, practical and cost-effective provision is an important aspect of controlling the already significant costs — which would more than triple under CARB's proposal — of the DF demonstration testing requirements. Further, carry-across DFs should not be limited on the basis of one engine family having generated, for example, a "Cycle 1" aging cycle result, while the candidate carry-across family may have generated a "Cycle 2" cycle result.

It is important to recall that DF testing needs to be done before emissions certification-testing starts so the final calibration for compliant emissions testing can be determined. Emissions testing typically begins 11-12 months before the applicable certification date (which would be prior to January 1, 2024, under the Omnibus Regulations). For Option 1, as noted, that would put the start of DF testing at January 2021 (5 months from the scheduled date of the Board hearing to adopt the Omnibus Regulations). For Option 2, the start of DF testing would be January 2022. Yet, the Omnibus Regulations are not expected to be fully finalized until mid-2021. That provides significantly negative leadtime for Option 1, and just five-months leadtime for Option 2, which is clearly insufficient and violative of multiple statutory and administrative rulemaking requirements.

Other aspects of CARB's proposed DF testing requirements are unreasonable as well. In amended §86.004-26.B.1.2.1 of the Omnibus Regulations, CARB proposes to increase the default break-in period for MY 2024 and later engines with SCR systems from 125 hours to 300 hours to ensure that stabilized emissions are achieved on emissions-data and durability-data engines. Alternatively under that provision, a manufacturer may run a minimum of three emissions tests at 60-hour intervals until emissions are sufficiently stabilized. However, the proposed regulations do not provide criteria for what constitutes "stabilized emissions." CARB should provide specific criteria for stabilized emissions, and should provide flexibility to utilize intervals different than 60-hour test intervals.

CARB's proposed amendments also fail to indicate which duty cycles manufacturers should use for the accumulation of hours during the break-in period. If CARB intends for manufacturers to use the same cycles as are proposed for durability aging (i.e., certification cycles or GEM cycles), EMA has concerns that the load factor of those cycles is not sufficient to demonstrate stabilized emissions. More specifically, CARB's proposed sequence of low load-factor operations is insufficient to produce the stabilized DF anchor point required for establishing a true (accurate) baseline value prior to conducting the DF testing exercise.

The process of degreening an engine and aftertreatment assembly relies on the use of a prescribed speed-load profile for a period of time that replicates either a normal or accelerated break-in period. In the absence of an extended break-in period, manufacturers rely on accelerated stabilization to reduce the time needed for both sliding assemblies (pistons/rings) and rotating assemblies (plain, roller or ball bearings) to establish a stabilized sealing surface and wear pattern. Low cylinder pressures resulting from sustained low-load operation are insufficient to provide the necessary combustion pressure, exhaust flow and temperature needed to produce consistent power cylinder sealing, and initial bearing surface mating.

In addition, lower temperature and exhaust flow rates do not adequately provide a stabilized ash layer to the DPF substrate. The initial frequency of regeneration events can help to stabilize catalysts by creating initial hydrothermal and sulfur exposures. Nonetheless, the aftertreatment stabilization process is defined by the exhaust flow, temperature, fuel consumption, chemical composition of the exhaust, and total run time. Manufacturers use different accelerated aging procedures, balancing those factors to achieve an equivalent stabilized point prior to a baseline DF measurement.

Low load-factor operation during the degreening process would unnecessarily extend the duration of the stabilization process due to the lower cylinder pressures and exhaust temperatures, and would lead to greater variability at the initial and subsequent test points. CARB should clarify that manufacturers can determine their own accelerated cycles for break-in, including for the 300-hour break-in, and should also, as noted, clarify the criteria for determination of "stabilized emissions."

Additionally, under CARB's DF proposal, in the case where manufacturers select the reduced dynamometer-aging option (Option 2), they would have to provide CARB with annual reports of in-use emissions and other data from vehicles originally sold in California. The emissions data would be derived from CARB's OBD NO_x-binning requirements (the "REAL" requirements). That is the same data CARB is proposing to require annually from every vehicle that operates in the state of California as part of CARB's upcoming HD Inspection and Maintenance ("I/M") Regulation. OEMs would be required to provide the annual report for each engine family certified by CARB, and would need to include data from at least 20% of the California-sold vehicles annually. It is unclear if the provisions would require reporting from the covered vehicles throughout their FUL, but, if so, that would lead to required annual reporting for 10 years for all diesel families from three consecutive model years.[26] For 2027 and later model years, the data-submission requirements would more than double, increasing to 50% of engines within all engine families for an even longer FUL from as many as 5 consecutive model years. That is an extreme and unduly burdensome requirement, especially for manufacturers that have not implemented telematic systems to facilitate the acquisition of those data. To alleviate this unreasonable and redundant data-submission requirement, CARB should limit the period of reporting to 3 years, or perhaps 5 years for 2031 and later model year families. Because owners often allow telematics contracts with OEMs to expire after as little as two years following purchase, CARB should have a declining percentage reporting requirement over the later reporting years of a model year. Additionally, many of the parameters to be required have nothing to do with emissions deterioration, and should be eliminated from the reporting requirement.

If CARB feels that having access to on-board-derived emissions data provides additional and sufficient assurances such that accelerated durability demonstrations can be allowed, and if CARB is already going to be receiving those data through the HD I/M program, CARB should simply allow for the accelerated durability demonstrations without imposing the reporting burden on manufacturers.[27] To that end, CARB should include the elimination of the REAL emissions-reporting obligations from the Omnibus Regulations as a provision of the soon-to-be adopted HD I/M Regulations to ensure that this excessive and duplicative requirement is removed from the Omnibus Regulations as expeditiously as possible.

As a final point on this topic, CARB should include in the regulatory provisions relating to durability demonstration testing a statement that manufacturers are permitted to design their DF programs with controls in place to establish "like" starting conditions for the emissions test points to ensure that the DF demonstration is assessing *deterioration*, and is not impacted by other conditions that can influence DF results. An example would be ensuring that emissons are

[26] The consequences of an OEM's failure to provide this REAL data after utilizing Option 2 are unclear.

[27] As discussed above, joint work is underway by CARB, EPA and EMA to validate a more cost-effective accelerated aftertreatment aging protocol, which will make any additional reporting associated with Deterioration Factor testing unnecessary.

measured consistently either before a regeneration event, or consistently after a regneration event. That would create the "all other things being equal" condition to assure that the measured (and calculated) emissions effect is faithfully representing the deterioration of the emissions control system and not something else.

14. CARB Has Not Fully Considered All of the OBD Requirements and Capabilities that Could Frustrate the Implementation of the Omnibus Regulations

Just as current PEMS NO_x-measurement capabilities render CARB's proposed in-use low-NO_x standards unenforceable and invalid, so too do the current emission-assessment capabilities of OBD systems and sensors. As CARB notes in its ISOR (at pp. I-10, II-10, III-10), OBD systems "assist CARB in verifying compliance with emission requirements." At the same time, CARB concedes, as it must, that current OBD systems and sensors are not capable of detecting and flagging emission exceedances at the proposed low-NO_x levels. CARB's concession on that point is as follows:

> "While OBD detection of faults at these proportionally lower [NO_x] levels will likely be required in the future, as it will be necessary to ensure that the maximum benefits of the proposed standards are maintained in-use, engine manufacturers have expressed concern about not knowing with certainty what impact the lower standards will have on their OBD monitoring capability. As such, engine manufacturers have requested interim relief until they have more certainty on what emission thresholds are achievable, and CARB concurs that the requested relief is reasonable and needed." CARB further concedes that "these higher OBD thresholds could allow emissions to exceed existing malfunction thresholds before detecting a fault, which could reduce the benefits of the proposed emission standards by allowing affected engines to operate without an indication of the need for repair." (ISOR, pp. II-10, III-10.)

Through its necessary acknowledgements of the detection limits of current OBD systems and sensors, CARB admits that current in-use enforcement systems and compliance protocols are incapable of assessing emissions at the low-NO_x levels that CARB is proposing, and that, as a result, the proposed emission standards are, again, as in the case with PEMS, inherently unenforceable as a practical matter, which renders them inherently unreasonable and invalid.

Beyond the foregoing critical issue going to the fundamental disconnect between the proposed NO_x standards and their in-use detection and enforceability, additional OBD issues also arise under the proposed Omnibus Regulations. CARB has rightly acknowledged that the multiple HD OBD requirements amount to real constraints on lowering emission standards, and that retaining the current HD OBD requirements and monitoring thresholds as they would scale-down to the proposed low-NO_x standards (i.e., at 2.0 x 0.020 g/bhp-hr) would hinder the implementation of the technologies necessary to implement the low-NO_x targets that CARB seeks to mandate. To mitigate that effect, CARB has agreed to leave the OBD thresholds where they are.

However, it still is not clear at what level of emissions impact a component will need to be measured by OBD systems to determine whether or not it has a meaningful impact on emissions. While that criteria is currently not measured directly against the emissions standard (e.g., a % of the NO_x standard) it is often used as an informal metric of emissions impacts from a component failure. With the new proposed NO_x standards set to 0.020g/bhp-hr, a component failure that could have been considered to have no significant impact on emissions might now be considered significant if it approaches the level of the NO_x standard. That could cause the new regulations to have a significant impact on OBD development costs and feasibility, even though CARB intends to keep the HD OBD standards as they are.

Accordingly, it is very important for CARB to consider fully all of the impacts that the Omnibus Regulations will have on the myriad HD OBD requirements, and all of the necessary OBD revisions that should be included in the relevant OBD Regulations (see e.g., §§ 1971.1, 1971.5). That necessary effort will help to promote the implementation of revised HD OBD regulations in the future that do not frustrate the implementation of the Omnibus Regulations.

To that end, and as noted, the latest proposals from CARB rightfully acknowledge that effective and accurate OBD functionality at the low-NO_x emissions levels of the proposed standards is infeasible. CARB has tried to account for that reality by maintaining the OBD thresholds at their current levels — e.g. 2x the existing NO_x standard, and an additive 0.020 g/bhp-hr to their existing PM standard, for final OBD thresholds of 0.40 g/bhp-hr for $NO_{x;}$ and 0.030 g/bhp-hr for PM. However, the current in-use emissions standards also are tied to the certification-cycle emissions standards — e.g. 1.5 x the FTP NO_x-threshold is the current NTE/In-Use emissions testing threshold. Today, that approach for correlating test-cell standards to in-use testing standards leads to an in-use NTE standard of 0.30 g/bhp-hr NTE NO_x, with a 0.15g/bhp-hr additive measurement allowance, for an aggregate in-use NO_x limit of 0.45 g/bhp-hr. The corresponding result, with respect to today's standards, is an effective OBD NO_x threshold of 0.40 g/bhp-hr, at which failed components must be detected and diagnosed. That currently leaves a small gap (0.05 g/bhp-hr) between the two emission values, where a component is required to be diagnosed, before a vehicle equipped with such a component could fail the PEMS-assessed in-use NTE standards.

Under the proposed new low-NO_x standards, the in-use NO_x standard would be lowered substantially, to 1.5x a standard of 0.050 g/bhp-hr (2024-2026MY) or 0.020 g/bhp-hr (2027 and later MYs), with a corresponding OBD NO_x threshold (if not adjusted) of 0.040 to 0.10 g/bhp/hr. The in-use PM standards would be similarly reduced. CARB acknowledges that it is impossible to diagnose emission thresholds at those values, and therefore would not require it for OBD at this juncture, but nonetheless is leaving the issue open for a potential tightening of the OBD thresholds through a follow-on OBD rulemaking. It is unrealistic to expect that OBD systems, strategies and calibration schemes will advance to the extent that CARB seemingly envisions. If a manufacturer cannot diagnose a system at such a low NO_x level, then guaranteeing emissions performance at such levels is inherently infeasible. CARB must take this into account fully before finalizing any new in-use emission standards. In that regard, CARB also should respect its own longstanding position that manufacturers should not be required to implement technologies that they cannot diagnose.

In the process of determining what emissions thresholds are achievable given the proposed substantial reductions in PM and NO_x, CARB and industry may determine that the state of the art for monitoring key components such as catalytic converters, particulate filters, aftertreatment system sensors, or EGR components will require intrusive monitors. Intrusive monitors temporarily increase tailpipe criteria and GHG emissions when executed. CARB requires the average emissions impact of an intrusive monitor to be included in a manufacturer's certification results (much like IRAFs). As such, the applicable standards either will need to be adjusted to reflect those necessary temporary increases, or the OBD thresholds will have to be maintained at levels high enough to not require intrusive monitoring. Without such measures, the current OBD provisions would make the proposed standards even more stringent and infeasible.

There is a similar concern regarding the diagnosis of multi-bed catalyst systems. SwRI has suggested that partial-volume OBD monitoring strategies might be deployed for configurations similar to the Stage 3 system. However, CARB OBD-certification staff have refused to approve partial monitoring strategies when proposed previously by some OEMs. CARB will need to clarify whether there has been a change of policy to account for the advent of systems such as those used on the Stage 3 engine.

Even if the NO_x and PM thresholds are maintained at today's absolute levels, manufacturers will be left with too little time to develop monitoring strategies for the host of the new envisioned low-NO_x emissions control devices and sensors, or to modify existing strategies to cope with the new technologies and control strategies. Working backwards from the 2024 MY start-of-production date, OBD certification approvals take several months to complete, and the OBD certification testing process takes longer than 4 months. The necessary component aging in advance of certification testing can take as long as 6 to 9 months on a HD engine, where software and calibrations are expected by CARB staff to be mature. To account for those certification timelines, final engine calibrations must be in place approximately 1.5 years before the start of engine production. Therefore, to release a certified product for the 2024 MY, a manufacturer must develop new systems, new sensors, new controls and actuators, *and* develop robust and complete OBD diagnostics, all by mid-2022, just one year following OAL approval of the Omnibus Regulations. As already noted, that is a wholly inadequate leadtime period. Consequently, and for the myriad other reasons discussed above, CARB should abandon the 2024MY requirements, and should focus instead on working with EPA to develop more carefully considered 2027 MY standards, which would allow for the necessary and legally mandated lead-time for the development and implementation of the complex low-NO_x emission technologies at issue.

CARB's OBD-related proposals also could create disincentives for manufacturers seeking to certify their engines to lower Family Emission Limits (FELs) between now and 2024. In that regard, the alternate NO_x and PM OBD thresholds proposed in 13 CCR §§ 1968.2(e) and 1971.1(f) are only available to 2024 MY and subsequent engines (and 2023 MY engines for manufacturers choosing to certify all of their engines to the full HD Low-NO_x program a year early). That would require a manufacturer attempting to certify an engine to a lower NO_x FEL prior to the 2024 MY to meet a more stringent OBD threshold than would be required in the 2024 MY. For example, if a manufacturer attempted to certify an engine to a 0.10 g/hp-hr NO_x FEL in MY 2022, that manufacturer would have to meet several OBD thresholds twice as stringent as would apply to the same engine certified to the same standard/FEL in the 2024 MY. EMA recommends that the alternate NO_x and PM thresholds be applied to all engines certified to FELs lower than the current NO_x and PM standards starting with MY 2022.

Similarly, the alternate NO_x OBD thresholds proposed in 13 CCR §§ 1968.2(e) and 1971.1(f) are only available to engines certified to NO_x standards/FELs of 0.10 g/bhp-hr or lower. That would result in manufacturers certifying engines to NO_x FELs above 0.10 g/bhp-hr, but lower than the current 0.20 g/bhp-hr standard, being subject to more stringent NO_x OBD thresholds than manufacturers certifying to 0.10 g/bhp-hr or lower. For example, a manufacturer certifying to a NO_x FEL of 0.12 g/bhp-hr would be subject to an OBD threshold of 0.32 g/bhp-hr for a NO_x catalyst monitor. Another manufacturer certifying an engine to a NO_x FEL of 0.10 g/bhp-hr would be subject to an OBD threshold of 0.35 g/bhp-hr for the same monitor. Requiring more stringent OBD thresholds for engines certified to less stringent emission standards is not logical. EMA recommends that the same alternate NO_x OBD thresholds be applied for all engines certified below the current 0.20 g/bhp-hr NO_x standard.

15. <u>CARB Has Not Considered the Impacts of Fuel Quality and Lubricants Issues</u>

Today, many engine manufacturers limit the fuel types that are permissible for use in their engines, and clearly detail those specifications in their owner's manuals. Limiting permissible fuels is essential to protect engine and aftertreatment components from the degradation that can occur due to aftertreatment-poisoning elements in the fuel, or due to the wear-aggravating or other characteristics of certain fuels.

CARB has proposed as part of the Omnibus Regulation that manufacturers' "maintenance instructions may not prohibit the use of commercially available diesel and biofuel blends that meet California's fuel specifications in title 4, CCR, § 4148." Critical fuel properties for engine hardware may vary depending on the specific hardware components used. The fuel properties required under ASTM D6751 (B100), ASTM D7467 9B6-B20), and ASTM D975 (B0-B5), as well as properties not specified in those standards, can have a broad range of impacts on fuel-injection system durability and performance. For example, acids and metals will react to form injector deposits, but total acid number is not controlled in ASTM D975, and the limits outlined in ASTM D7467 and ASTM D6751 are higher than some comparable global standards and OEM recommendations. Malfunction of critical engine hardware due to fuel quality also may lead to significant emissions-performance issues. As hardware challenges are unique to the OEM, the OEM should have the sole discretion to designate compatible fuels with their products. CARB should eliminate this provision prohibiting an OEM's specification of allowable fuels.

There also is clear evidence that engine emissions control can be directly impacted by fuel properties, not simply from the longer-term poisoning effects of fuel contaminants. Several SAE papers have explored those shifts in engine-out emissions. Testing on a MY 2017 engine showed an engine-out NO_x increase of 7% in the composite FTP with a B20 fuel, compared to operation with ULSDF due to the higher oxygen content inherent in biodiesel. On the other hand, the same engine tested with NESTE R100 saw a 9% reduction in engine out NO_x compared to the ULSDF baseline. The fuel requirements CARB has proposed leave very little "margin for error" from those types of fluctuations in emissions.

To account for the fact that modern engines and aftertreatment systems cannot tolerate certain fuel types, and because fuel constraints are required to assure long-term emissions compliance, current regulations allow a vehicle that has been mis-fueled in contravention of manufacturer instructions to be rejected from consideration for in-use testing.

CARB's Omnibus Regulations would no longer permit candidate vehicle rejection on that basis. A manufacturer would have to allow a candidate vehicle to be tested, provided it had been operated on any "commercially available" fuel in California. That would include all commercially available diesel and biofuel blends that meet California's fuel specifications detailed in Title 4, CCR, § 4148 (which could include fuels up to B100). That is inconsistent with EPA's guidance regarding vehicle-rejection criteria for in-use testing, and is unacceptable for engine manufacturers seeking to ensure long-term emissions control by limiting the allowable fuel types that may be used. It also is inconsistent with the goals of the Omnibus Regulations, where CARB is implementing 90% reductions in the applicable NO_x standards, with significantly extended FUL periods. Those major challenges for engine and vehicle manufacturers would be undermined by CARB's lax control of fuel characteristics in the marketplace, which allows (and even promotes) fuels that can result in damaging effects to emissions control systems. CARB's proposed elimination of manufacturers' ability to specify the allowable fuels for their products to ensure long-term emissions control is not only a significant impediment to manufacturers' ability to comply with the proposed stringent low-NO_x requirements, it is environmentally detrimental. CARB should retain the current criteria for rejecting candidate vehicles from HDIUT as they pertain to fuels.

Similarly, CARB needs to consider the relationship between engines and engine oils. Engine oils, or engine lubricants, perform critical functions such as: reducing engine wear, enhancing fuel efficiency, and helping to provide protection for both the engine and the emissions system. CARB's Omnibus Regulations are infeasible and unworkable not only because they provide insufficient lead time for engine technology development, but also because they fail to consider related effects – such as those pertaining to engine lubricants. The proposed low-NO_x standards and extended useful life and warranty periods could have significant impacts necessitating new engine oil formulations. CARB's proposed implementation schedule, however, does not allow for such necessary considerations.

16. EMA Recommends Changes to CARB's Proposed AB&T NOx-Credit Trading Provisions

CARB has introduced major changes to the federal AB&T emissions credit program as part of the Omnibus Rulemaking. CARB proposes to appropriate a portion of a manufacturer's credits earned through the federal program, and segregate them to create a dedicated California-only credit bank. EMA has several concerns about that proposal.

First, CARB should not assign an expiration date to credits generated under the federal AB&T program. It is manifestly unfair to retroactively assign a shelf-life to credits that were generated under the provisions of a regulation where no expiration or sunset dates were defined, and therefore not anticipated. Second, establishing a sunset date for newly-generated credits under this program disincentivizes manufacturers that might otherwise seek to take advantage of the AB&T provisions to launch earlier introductions of lower-emissions engines. The 5-year sunset provision thereby limits the environmental gains CARB consistently attributes to AB&T programs. CARB should follow the practice established under EPA's and CARB' MY 2007/2010 regulation, which does not assign an expiration date to credits generated under the program.

Another limiting provision proposed within the AB&T program relates to the assigned NO_x FEL caps. CARB has proposed to apply a NO_x FEL cap of 0.10g/bhp-hr to model year 2024 through 2026 engines, and 0.05g/bhp-hr to MY 2027 and later engines. Those unnecessarily low FEL caps again will disincentivize manufacturers from participating in the AB&T program. EMA recommends that CARB follow the historical practice of setting the FEL cap at the level of the prior emissions standard; that is, to 0.20g/bhp-hr for NO_x.

CARB has proposed to define "California Sales Volume" as "the number of new California-certified engines or new vehicles sold in a given model year within the State of California." CARB's definition requires additional detail. CARB staff need to address key considerations, such as how to account for certified engines in the production pipeline, and for complete vehicles that remain unsold on dealer lots after the beginning of a subsequent model year. Further, the draft regulation should clarify the treatment of engines or vehicles "first introduced for sale" in California, to avoid OEMs having to track vehicles that are traded among dealers across state lines.

CARB proposes to allow manufacturers 90 days to submit year-end AB&T reports, and an additional 180 days to send corrections to the AB&T report. Corrections could be submitted after the 180-day deadline, but only if the corrections are not favorable to the manufacturer's credit position. The 90-day period is insufficient to prepare and submit an accurate AB&T report, given the intricacies of tracking production, distribution, and sales of engines and vehicles in the heavy-duty market. EMA recommends, consistent with our comments on EPA's 2020 Technical Amendments NPRM,[28] that the AB&T report be submitted within 180 days of the year-end. Manufacturers should then have an additional 90 days (in effect, 270 days from the end of the model year) to submit any corrections, if necessary. EMA also recommends that errors in AB&T reports should be corrected regardless of impact to a manufacturer's credit position, or should not be accepted at all after the 270-day correction deadline. Finally, manufacturers should be allowed to request a reporting extension as circumstances may warrant.

CARB has proposed that NO_x emissions credits generated from the sale of HD zero-emissions vehicles (ZEVs) may be applied to any other heavy-duty AB&T credit averaging sets where a deficit may exist. EMA supports the use of zero-emissions credits for achieving compliance with the proposed NO_x standards, while maintaining the prohibitions that currently exist for NO_x and GHG AB&T where credit transfers are not allowed across averaging sets. EMA objects to the use of any ZEV credits in an averaging set other than the averaging set from which they were earned. More fundamentally, CARB should make it clear that any HD ZEV NO_x credits that are earned are the property of the certifying powertrain manufacturer, not the vehicle manufacturer as implied in the proposed amendments to the California provisions of § 86.007-15.

[28] Improvements for Heavy-Duty Engine and Vehicle Test Procedures, and Other Technical Amendments; Proposed Rule, EPA-HQ-OAR-2019-0307, 85FR28140-28361, May 12, 2020.

17. <u>The Restrictions Imposed upon the Carry-Over/Carry-Across Process Are Unclear and Unreasonable</u>

CARB proposes that if a 2024 or subsequent model year engine family or test group does not comply with the HDIUT requirements in title 13, CCR, §§ 2111-2140, and Part II, Subpart T, or with the EWIR requirements in title 13, CCR, §§ 2141-2149, or if engines in a family are equipped with an emission control component that exceeds the thresholds specified in title 13, CCR, § 2143 (and the component was not improved for the model year for which certification is requested), a manufacturer cannot request a carryover or carry-across certification application based on data from that engine family or test group.

This new California provision focusing on failures of in-use testing of prior model years, or emission control component failures as low as 4% or 25 units where an improved component is not ready for production, would dramatically impact an OEM's ability to complete CARB's certification process in a timely or cost-efficient manner in advance of the start of new model years. Moreover, CARB is not clear regarding the potential consequences of this proposed change. It is unclear whether the restriction is that an OEM cannot check the "Carryover certification" box on the application, or that CARB will not certify the family without a complete set of new data, or, even worse, that the engine family cannot be certified at all. CARB should clarify the limitations imposed under its proposal. In that regard, CARB should recognize the potentially severe timing consequences at issue. For example, if a manufacturer's currently-certified family should reach one of the carry-over-disqualifying conditions late in the year during which the carryover or carry-across application is pending, what will the consequences be? Any potential consequences clearly should not lead to production shut-downs or other overly-disruptive outcomes.

18. <u>The Coronavirus Pandemic Has Fundamentally Changed the Regulatory Landscape</u>

Beyond the many significant cost-effectiveness and feasibility concerns expressed in these comments, including those regarding leadtime, EMA and its member companies, along with all individuals and organizations in the U.S. and around the world, continue to face the unprecedented disruptions and uncertainties caused by the ongoing COVID-19 pandemic. In response to the resulting public health crisis, many manufacturers have instituted work-from-home policies, and have instituted reduced operations to ensure the health and safety of their employees.

The global COVID-19 pandemic also is forcing EMA-member companies to consider the practical realities of how the pandemic will impact their operations and business outlook over the next several years. While it is difficult to predict the scale and duration of the impacts on member-company operations and finances (including access to capital) — let alone the scope and duration of the likely damage to the U.S. and global economies — we all must acknowledge the gravity of the situation, and consider and plan for practical measures to deal with the crisis, and ultimately its aftermath. To that end, EMA strongly urges CARB to reconsider the program elements and effective dates of the Omnibus Regulations, which, even before the current crisis upended the world, would present unworkable and cost-prohibitive challenges to manufacturers, and may, as we have noted, preclude future HDOH product availability in California.

19. Conclusions

CARB's proposed Low-NO_x Regulations are cost-prohibitive, infeasible, unenforceable and illegal. The cost implications, and the related pre-buy/no-buy response to the proposed requirements, will be highly disruptive to the California trucking industry, and potentially the economy as a whole, with marginal air quality benefits, especially as those benefits might relate to ozone attainment in the South Coast. The contemporaneous ACT Rule will further strain and dilute manufacturers' research and product-development resources, and thereby OEMs' ability to comply with those overlapping and overly burdensome provisions. The net result could be an absence of CARB-compliant HDOH products in California starting in 2024. Consequently, CARB should pause and fundamentally rethink the proposed Omnibus Regulations.

Respectfully Submitted,

TRUCK AND ENGINE
MANUFACTURERS ASSOCIATION

Exhibit A

FINAL DRAFT

COST STUDY:

PROPOSED HEAVY-DUTY ENGINE AND VEHICLE EMISSIONS REGULATIONS

PREPARED FOR:

TRUCK & ENGINE MANUFACTURERS ASSOCIATION

333 WEST WACKER DRIVE

CHICAGO, ILLINOIS ▪ 60606

March 19, 2020

Contents

Exhibit A

ACT Research Cost Study of the Proposed Omnibus Low-NO_x Rulemaking

Executive Summary

Based on a survey of the commercial vehicle and engine manufacturing industry completed in Q1, 2020, this study presents ACT Research's best estimates of the sum of the direct and indirect costs of meeting the goals of the California Air Resources Board (CARB) Omnibus Low-NO_x Rulemaking (Omnibus Regulations), as also referenced in the ANPRM for EPA's Cleaner Trucks Initiative (CTI). We present estimates for costs of both a nationwide and a California-only program.

This study's focus is on the costs (including per-vehicle costs) that the truck and engine manufacturing industry likely will incur to comply with the proposed Omnibus Regulations. The study's primary conclusion is that full compliance with the proposed low-NO_x emission standards and other requirements, assuming they track the proposed Omnibus Regulations, will cost the truck and engine manufacturing sector a Net Present Value (NPV) of **$9.1 – $13.0 billion**.

Assuming the proposed Omnibus Regulations are implemented, manufacturers ultimately will recoup most of those costs through higher vehicle prices. It is the trucking industry that will bear most of the increased costs going forward. Longer-term, the trucking industry eventually will be able to pass the higher costs of compliance on to the shipping community, which in turn will pass them on to consumers. However, given the highly competitive nature of the trucking industry, we also detail the costs of the very likely scenario of a substantive equipment "pre-buy/no-buy" to avoid, at least initially, the higher truck and engine costs associated with the proposed Omnibus Regulations. In ACT's modeling, the resulting overcapacitization in the freight hauling industry (due to pre-buys of vehicles) likely will yield aggregate pre-buy impacts between **$6.5 - $8.6 billion** in 2019 dollars, solely as a result of lower freight rates due to overcapacity, and there will be little opportunity to recoup the lost shipping revenues during the periods of overcapacity.

The combined regulatory impact on the manufacturing sector and trucking companies falls between NPVs of $15.6 and $21.6 billion.

Our estimates do not model the increased costs out into perpetuity. Rather, our cost estimates are focused on the two key years when costs are likely to rise significantly: 2027 and 2031. In our analysis, fixed costs were allocated over multi-year product programs. In addition, we have not tried (yet) to estimate the long-run costs to the trucking industry from deploying higher-cost equipment. The costs studied here are solely for the truck and engine manufacturing sector, and just include the pre-buy related effects on trucking. In our judgement, adding the long-run costs on trucking, while likely worth a more thorough analysis, would effectively be double-counting the costs we have estimated for the manufacturers. We include an analysis of the costs for the trucking industry in the Pre-buy/No-buy section, but only to inform our modeling regarding the degree of excess capacity. It should be noted that the increased taxes, insurance costs, financing costs, and emissions fluid costs that trucking companies will face are not included in this aggregate cost estimate of $15.6 to $21.6 billion.

Summary Tables. Tables 1-3 summarize the results of our cost study. Our findings related to the costs associated with the **MY2027** step of the proposed Omnibus Regulations are itemized in *Table 1: Cost Estimates to Meet Proposed MY2027 Vehicle Standards*. In MY2027 at the national level, and using the 3% and 7% discount rates to bracket the ranges, we estimate the proposed emissions requirements would cost the industry $1.8 – $2.4 billion for medium-heavy duty vehicles and engines, and $4.5 – $6.1 billion for heavy-heavy duty vehicles and engines, which **sums to $6.3 billion at a 7% discount rate, and $8.5 billion at a 3% rate**. **On a per-unit basis, the cost of compliance ranges from $17,610 to $23,886 for heavy-heavy-duty (HHD) diesel vehicles, and $11,752 to $15,940 for medium-heavy-duty (MHD) diesel vehicles.** The total cost figures are smaller for a California-only program, but per-unit costs rise sharply because of the relatively small number of units sold in California.

Table 1: Cost Estimates to Meet Proposed MY2027 Vehicle Standards

	National MY2027 from MY2018 base				California MY2027 from MY2018 base			
Discount Rate	7%	3%	7%	3%	7%	3%	7%	3%
	MDD	MDD	HDD	HDD	MDD	MDD	HDD	HDD
per unit								
Total Direct Costs	$3,688	$5,002	$5,376	$7,292	$9,058	$12,286	$7,738	$10,495
Total Indirect Costs	$8,064	$10,938	$12,234	$16,594	$32,416	$43,968	$39,949	$54,184
Cost Increase per Unit ($)	**$11,752**	**$15,940**	**$17,610**	**$23,886**	**$41,474**	**$56,254**	**$47,686**	**$64,679**
$ in millions								
Total Direct Costs	$562	$762	$1,380	$1,872	$72	$98	$122	$166
Total Indirect Costs	$1,228	$1,666	$3,141	$4,260	$258	$349	$631	$856
Total Cost Increase ($M)	**$1,790**	**$2,428**	**$4,521**	**$6,132**	**$329**	**$447**	**$753**	**$1,021**

Source: ACT Research Co., LLC: Copyright 2020

The cost estimates itemized in *Table 2* summarize the results of our cost study for **MY2031** compliance. Those costs are primarily related to meeting the extended useful life and emission warranty provisions of the proposed Omnibus Regulations. The cost figures amount to additions to the baseline MY2027 costs (in Table 1), and show the incremental cost estimates for MY2031. **For HDD vehicles, our survey indicated an additional $8,352 – $13,194 in costs per truck, depending on the discount rate utilized. For MHD vehicles, the additional costs would range from $3,689 – $5,827 per truck.** Combining the HHD and the MHD diesel model outputs, we estimate a discounted cost that ranges between **$2.7 – $4.4 billion for the MY2031 proposals on a nationwide basis.**

Table 2: Additional Cost Estimates to Meet Proposed MY2031 Vehicle Standards

	National				California			
	MY2031 from MY2027 base				MY2031 from MY2027 base			
Discount Rate	7% MDD	3% MDD	7% HDD	3% HDD	7% MDD	3% MDD	7% HDD	3% HDD
per unit								
Total Direct Costs	$0	$0	$157	$248	$0	$0	$150	$238
Total Indirect Costs	$3,689	$5,827	$8,196	$12,946	$9,891	$15,624	$10,068	$15,904
Cost Increase per Unit ($)	**$3,689**	**$5,827**	**$8,352**	**$13,194**	**$9,891**	**$15,624**	**$10,219**	**$16,142**
$ in millions								
Total Direct Costs	$0	$0	$42	$66	$0	$0	$2	$4
Total Indirect Costs	$585	$924	$2,189	$3,458	$55	$86	$152	$240
Total Cost Increase ($M)	**$585**	**$924**	**$2,231**	**$3,525**	**$55**	**$86**	**$154**	**$244**

Source: ACT Research Co., LLC: Copyright 2020

Table 3 aggregates the cost estimates for the **MY2027 and MY2031** cost models, reflecting our estimates of the combined costs of the proposed Omnibus Regulations. On a nationwide basis, the total combined cost of the Omnibus Regulations for both MHD and HHD vehicles is **$9.1 billion to $13.0 billion**, depending on whether a 7% or 3% discount rate is utilized. **On a per-unit basis, the nationwide cost for HHD vehicles ranges from $25,963 at a 7% discount rate, to $37,079 at the 3% rate. For MHD vehicles, the per-unit costs range from $15,441 to $22,767, respectively.** On a California-only basis, the aggregate total costs range from $1.3 – $1.8 billion, which are much smaller than the nationwide costs, but some expense line-items like R&D were relatively fixed. Therefore, on a per-unit basis, the per-unit cost increases range from $57,905 to $80,821 per HHD vehicle, and from $51,365 to $71,878, per MHD vehicle.

Table 3: Cost Estimates to Meet Proposed Combined MY 2027 and MY2031 Vehicle Standards

	National				California			
	MY2027 + MY2031 from MY2018 base				MY2027 + MY2031 from MY2018 base			
Discount Rate	7% MDD	3% MDD	7% HDD	3% HDD	7% MDD	3% MDD	7% HDD	3% HDD
per unit								
Total Direct Costs	$3,688	$5,002	$5,533	$7,540	$9,058	$12,286	$7,888	$10,732
Total Indirect Costs	$11,753	$16,765	$20,430	$29,540	$42,307	$59,591	$50,017	$70,089
Cost Increase per Unit ($)	**$15,441**	**$21,767**	**$25,963**	**$37,079**	**$51,365**	**$71,878**	**$57,905**	**$80,821**
$ in millions								
Total Direct Costs	$562	$762	$1,422	$1,938	$72	$98	$124	$169
Total Indirect Costs	$1,813	$2,590	$5,330	$7,718	$312	$435	$783	$1,096
Total Cost Increase ($M)	**$2,375**	**$3,352**	**$6,752**	**$9,656**	**$384**	**$533**	**$907**	**$1,265**

Source: ACT Research Co., LLC: Copyright 2020

Methodology

This cost study was performed using federal guidelines that correspond to EPA's Guidelines for Economic Analysis and OMB Circular A-4. The baseline assumptions for our analysis are that:

1) Heavy-duty truck manufacturers would continue to work toward meeting the established GHG-2,
2) but would otherwise not explicitly target
    a. incremental NO_x emissions reductions,
    b. improved low-load SCR performance, or
    c. longer useful lives for aftertreatment systems.

In light of the pending GHG-2 regulations, we used professional judgement to discount some of the cost inputs that we received from manufacturers, if those inputs did not take into account the improved fuel economy and reductions in fuel consumption, which will help to meet the proposed Omnibus Regulations.

We followed the methods specified by the Environmental Protection Agency (EPA) and the Office of Management and Budget (OMB) to conform to the government's Social Cost definition, though we have noted where we otherwise would differ with those methods (i.e., inflation and discount rates). We have also presented below an additional set of values that discount the future costs at the private weighted average cost of capital, which for this industry is quite high. Our "Private Cost" estimates below are only alternative results, not EPA/OMB recommended results, and so are not included in the summary tables above.

ACT Research's cost estimates are based upon industry inputs consisting mainly of confidential business information (CBI), and as a result, specific technology solutions will not be discussed here except to note that those anticipated solutions were not uniform. As explained below, we used conservative analytical judgements where possible. For example, the current regulatory baseline for warranty coverage is 100,000 miles (five years, 3,000 hours). However, our research confirmed that the industry standard for new heavy-duty trucks is a 2-year/250,000-mile warranty that is built into the price. As a result, our study uses 250,000 miles as the baseline, resulting in lower incremental costs than otherwise would have been the case had we used the more common government research practice regarding the existing regulatory baseline.

Discount Rates, Social and Private. Consistent with EPA and OMB guidelines to discount future costs back to their present value at 3% and 7% discount rates in order to determine NPV, we have presented our results discounted at both of those rates. However, considering the significant uncertainty involved in estimating the future costs at issue, we also present the results of our cost estimates discounted using an alternative private cost methodology. The private cost methodology provides for the use of the Weighted Average Cost of Capital (WACC) for the truck and engine manufacturing industry as our discount rate. In calculating the 10% WACC, we used current equity values, as of January 2020, and debt and interest rates from the manufacturers' most recent annual reports.

Accordingly, in addition to utilizing the 3% and 7% social cost discount rates, we also present an alternative cost estimate (in Table 4) using our more conservative 10% WACC discount rate. While this is more conservative than the social cost methodology, we believe it accounts for some of the uncertainty inherent in this study, including: significant uncertainty about the future state of emissions-control technology, and regarding the most likely compliance pathways that manufacturers may follow. For example, we are estimating that manufacturers will need to budget for two replacements to aftertreatment systems in the life of their trucks in order to comply with the extended useful life and warranty provisions of the Omnibus Regulations. However, between now and MY2027, it is possible that durability could be improved to remove some of those costs. It also is possible that replacement aftertreatment systems will not last as long on older engines, which also is reflected in this cost study.

In light of these and other uncertainties, the alternative 10% WACC-based discount rate could be a reasonable way to estimate more conservatively the unknown variables pertaining to the various potential cost inputs and impacts. The larger alternative discounting mechanism that we have used, in essence, could serve fairly well in lieu of a more formal sensitivity analysis at a point in time when specific technology paths are not yet known.

Inflation methodology. We used inputs in 2019 dollars as it was the year our cost survey was initiated, adjusting for the OEMs who responded in 2018 dollars using the BEA's GDP Price Deflator. We thought it would be fair to use a lower inflation rate or perhaps even deflationary figure given the historical experience in this industry, but EPA (through EMA) indicated that the GDP Deflator is the standard. Adhering to EPA's recommended use of the GDP Deflator may inflate the estimated cost of the Omnibus Regulations, leaving room for further study.

Heavy-Heavy Duty Market Sizing. We used 2018 vehicle manufacturer (OEM) market shares as our baseline and assumed those shares as a constant into the future. However, instead of using the 2018 market size and simply rolling it forward, we took into account the fact that 2018 was the fifth-largest year ever for U.S. Class 8 truck production. As it happens, two of the higher production years were 2005 and 2006, with 2006 being the biggest U.S. Class 8 production year ever. Not coincidentally, those two "top-five" years occurred immediately ahead of the expensive EPA07 emissions standards for heavy-duty trucks and engines. We will discuss this "pre-buying" issue later in this report.

To provide a representative baseline, we used a five-year trailing average of U.S. Class 8 truck production (HHD diesel), or 239,000 units, and scaled it up at 1% per-year to account for economic growth, and adjusted for freight productivity. While freight demand grows over time as the population grows, shippers also find ways to improve design and packaging in ways that require fewer truckloads for a given set of goods. As a result, our analysis uses a MY2027 U.S. Class 8 nationwide market size estimate of 257,000 units.

For the California market, based on industry inputs, we used a baseline of just under 7% of nationwide industry sales, and scaled that starting point down by 7.5% in MY2027 to reflect assumed progress toward CARB's target of 15% zero-emission heavy duty tractors by 2030. We therefore estimate that California will represent just over 6% of nationwide HHD sales in MY2027.

For MY2031, we continued to scale nationwide HHD sales up by a 1% cumulative annual growth rate, bringing the nationwide HHD market to 267,000 units. We also continued with the assumption that California would achieve its 2030 target of 15% zero emissions heavy-duty vehicles, taking California down under 6% of nationwide HHD duty diesel truck sales.

Medium-Heavy Duty Market Sizing. For the MHD market, we used a trailing five-year average of U.S. sales of 142,000 units per-year, scaled up at 1% per-year to account for economic growth and adjusted freight productivity, in line with the above discussion regarding the HHD market. That resulted in a nationwide MHD market size of 152,000 units.

For the California market, we used a baseline of just under 7% of nationwide industry sales, also based on industry inputs, and scaled that down by 20% in MY2027 to reflect progress toward CARB's target of 50% zero-emission MHD vehicles by 2030. We estimate that California will represent just over 5% of nationwide MHD sales in MY2027.

For MY2031, we continued to scale nationwide MHD sales up at a 1% cumulative annual growth rate, and we made the assumption that California would achieve its target of 50% zero-emission vehicles, taking California down to 3.5% of nationwide MHD diesel truck sales.

State versus Federal Considerations. Based on this cost study, we conclude that the local benefits of California-only regulations do not justify the very significant costs that would impact trucking-related business on a nationwide basis. Due to the relatively small number of trucks sold in California, the research and development costs of advanced aftertreatment on a per-unit basis could be unacceptably high. Our survey of OEMs showed that only about 7% of heavy-duty trucks are sold in California, significantly less than the State's share of GDP.

Our cost survey also shows that the industry would spend $715 million on research and development for the proposed standards nationally, and $603 million on a California-only standard. The difference between the two totals reflects that fewer models would be offered under a California-only scheme. However, on a per-unit basis, using the market size detailed previously and amortizing the costs over an industry-standard three-year product platform cycle, those R&D costs amount to about $2,800 per-unit at a national level and $38,200 per-unit if the regulations applied only to California.

MY2024 Infeasibility. We are not providing separate estimates for the MY2024-26 elements of the proposed Omnibus Regulations because we did not receive indications that manufacturers can, or will, develop and introduce the technologies that could be used to meet those proposed standards by the 2024MY at reliable product-quality levels. The industry respondents to our survey cited numerous feasibility problems with the MY2024 time horizon. We believe that for some key vehicle categories, the standards proposed under the Omnibus Regulations are technically infeasible within the lead time allowed. Accordingly, we have not fully estimated the costs for the initial phase of the Omnibus Regulations for tractors and vocational vehicles. The lack of sufficient lead times for the development of the required additional technologies would result in significant risks of quality issues later in vehicle life. Simply stated, we could not develop any realistic cost estimates for a near-term regulatory program that manufacturers indicated is essentially unworkable. We believe that the MY2024 proposals would result in a decrease in the in-use reliability and durability of new heavy-duty vehicles, and we cannot accurately quantify the costs that would be associated with such problems. Instead, we merely note that unit costs would likely be greater than the costs we have estimated in this study for a nationwide MY2027 and MY2031 standard.

Heavy-Heavy Duty MY2027 Costs. We estimate in Table 4 that the low-NO_x standards proposed for MY2027, including a carry-forward of the MY2024 proposals, would cost HHD truck manufacturers $6.6 billion on a nationwide level, or $25,825 per-unit, in 2019 dollars. For California, our cost estimate of $1.1 billion for the HHD vehicle sector equates to $69,930 per-unit. That level of price increase would in all likelihood significantly reduce the choices of vehicles available in the California market, and could force some smaller volume manufacturers out of the California market. **On an inflation-adjusted and discounted basis, using the 3% and 7% discount rates recommended in the EPA and OMB guidelines, the net present value of the HHD costs associated with the Omnibus Regulations on a nationwide basis is $17,600 – $23,900 per HHD vehicle, and $4.5 – $6.1 billion for the HHD industry. For California-only, the net present value ranges from $47,700 – $64,700 per HHD vehicle, and $750 million to $1.02 billion for the HHD industry.** Note that in the far-right column of Table 4, we present the cost figures discounted at the 10% WACC, and those costs are considerably lower and could be a better way to account for the uncertainties relating to the possible incorporation of unforeseen technology improvements in the coming years.

Direct Costs. The direct costs included in the foregoing estimates incorporate specific changes to engines, aftertreatment systems and on-board diagnostics. Those costs do not represent any specific technology path, but rather a weighted average of the various manufacturers' inputs.

Those inputs add up to $7,900 per-unit for HHD diesel vehicles nationally, and $11,350 per-unit in California in 2019 dollars. The net present value of those figures is $5,375 – $7,290 nationally, and $7,740 – $10,500 in California, using the 3 and 7% discount rates to bracket the ranges. (See Table 4.)

Indirect Costs. The industry estimated $603 million in R&D costs to meet the MY2027 requirements (including the MY2024 elements) of the Omnibus Regulations in California, and $715 million for a nationwide program. Using inputs from the manufacturers, we amortized the R&D costs over the typical program life in the industry of three to four years.

The other indirect costs were primarily associated with the proposed extended warranty and useful life periods, as well as the related compliance-enforcement programs. The warranty and useful life costs are largely variable, but the compliance programs and R&D requirements are largely fixed. Some manufacturers may plan to find savings by offering fewer vehicle options, but applying those fixed costs to California's 15,800-unit HHD market still results in major per-unit cost increases relative to the 257,000-unit nationwide market.

Table 4: Cost Estimates to Meet Proposed Combined MY2027 Standards for HHD Vehicles

Heavy-heavy Duty Diesel										
Social Cost Methodology				MY2027 - from MY2018 baseline					*Private Cost (not Social)*	
Costs to Develop & Build Ultra-Low-NOx products	2019 dollars		Inflation-adjusted at:		Discounted at:		Discounted at:		Discounted at WACC	
			2%		3%		7%		10%	
	National	California	National	California	National	California	National	California	National	California
Industry Units	256,712	15,789	256,712	15,789	256,712	15,789	256,712	15,789	256,712	15,789
Per unit costs ($)										
Direct manufacturing costs										
Engine	$3,157	$3,811	$3,699	$4,465	$2,920	$3,525	$2,153	$2,599	$1,675	$2,022
Aftertreatment	$4,589	$6,171	$5,376	$7,230	$4,244	$5,708	$3,129	$4,208	$2,434	$3,274
Vehicle + On-Board Diagnostics	$139	$1,365	$162	$1,599	$128	$1,263	$95	$931	$74	$724
Total Direct Costs	**$7,884**	**$11,347**	**$9,237**	**$13,294**	**$7,292**	**$10,495**	**$5,376**	**$7,738**	**$4,183**	**$6,020**
Indirect Costs to Manufacturers										
Research and development costs	$2,786	$38,171	$3,265	$44,723	$2,577	$35,305	$1,900	$26,029	$1,478	$20,251
Warranty on new technology	$2,208	$2,511	$2,587	$2,943	$2,042	$2,323	$1,506	$1,713	$1,171	$1,332
Warranty Step 2	$3,311	$3,757	$3,880	$4,401	$3,063	$3,475	$2,258	$2,562	$1,757	$1,993
Useful Life extension	$9,451	$11,178	$11,074	$13,097	$8,742	$10,339	$6,445	$7,622	$5,014	$5,930
Compliance program costs	$184	$2,966	$215	$3,475	$170	$2,744	$125	$2,023	$97	$1,574
Total Indirect Costs	**$17,940**	**$58,583**	**$21,020**	**$68,639**	**$16,594**	**$54,184**	**$12,234**	**$39,949**	**$9,518**	**$31,081**
Cost Increase per Unit ($)	**$25,825**	**$69,930**	**$30,258**	**$81,934**	**$23,886**	**$64,679**	**$17,610**	**$47,686**	**$13,701**	**$37,101**
EOEM Costs ($M)										
Direct manufacturing costs										
Engine	$810	$60	$949	$70	$750	$56	$553	$41	$430	$32
Aftertreatment	$1,178	$97	$1,380	$114	$1,090	$90	$803	$66	$625	$52
Vehicle + On-Board Diagnostics	$36	$22	$42	$25	$33	$20	$24	$15	$19	$11
Total Direct Costs	**$2,024**	**$179**	**$2,371**	**$210**	**$1,872**	**$166**	**$1,380**	**$122**	**$1,074**	**$95**
Indirect Costs										
Research and development costs	$715	$603	$838	$706	$662	$557	$488	$411	$379	$320
Warranty on new technology	$567	$40	$664	$46	$524	$37	$387	$27	$301	$21
Warranty Step 2	$850	$59	$996	$69	$786	$55	$580	$40	$451	$31
Useful Life extension	$2,426	$176	$2,843	$207	$2,244	$163	$1,654	$120	$1,287	$94
Compliance program costs	$47	$47	$55	$55	$44	$43	$32	$32	$25	$25
Total Indirect Costs	**$4,606**	**$925**	**$5,396**	**$1,084**	**$4,260**	**$856**	**$3,141**	**$631**	**$2,443**	**$491**
Total Cost Increase ($M)	**$6,629**	**$1,104**	**$7,767**	**$1,294**	**$6,132**	**$1,021**	**$4,521**	**$753**	**$3,517**	**$586**

Source: ACT Research Co., LLC: Copyright 2020

Medium-Heavy Duty MY2027. We estimate (in Table 5) that the low-NO_x standards contemplated for MY2027, including the MY2024 proposals, would cost $2.6 billion on a nationwide basis, or $17,230 per-unit. On a California-only basis, the program would cost $500 million, which equates to $60,820 per-unit. That level of price increase would in all likelihood significantly reduce the choices available in the California truck market, thereby decreasing competition by forcing some low-volume manufacturers out of the market. **The net present value of those figures is $1.8 – $2.4 billion for the MHD industry on a nationwide basis, or $11,750 – $15,940 per-vehicle, using the 3% and 7% discount rates. For California-only, the net present value ranges from $330 – $450 million at the discounted cost rates, which boost the per-unit costs to $41,500 – $56,250.** Those MHD costs are largely similar to the cost estimates for HHD diesel vehicles. While smaller in absolute terms, they represent similar proportional price increases relative to new vehicle prices.

Table 5: Cost Estimates to Meet Proposed Combined MY2027 Standards for MHD Vehicles

Medium-heavy Duty Diesel										
Social Cost Methodology	**MY2027 - from MY2018 baseline**								*Private Cost (not Social)*	
Costs to Develop & Build Ultra-Low-NOx products	2019 dollars		Inflation-adjusted at:		Discounted at:		Discounted at:		Discounted at WACC	
Phase 1, part 1			2%		3%		7%		10%	
	National	California	National	California	National	California	National	California	National	California
Units	152,340	7,944	152,340	7,944	152,340	7,944	152,340	7,944	152,340	7,944
Per unit costs ($)										
Direct manufacturing costs										
Engine	$1,894	$4,882	$2,220	$5,720	$1,752	$4,516	$1,292	$3,329	$1,005	$2,590
Aftertreatment	$3,186	$7,762	$3,733	$9,094	$2,947	$7,179	$2,173	$5,293	$1,690	$4,118
Vehicle + On-Board Diagnostics	$328	$640	$384	$749	$303	$592	$224	$436	$174	$339
Total Direct Costs	**$5,408**	**$13,283**	**$6,337**	**$15,564**	**$5,002**	**$12,286**	**$3,688**	**$9,058**	**$2,869**	**$7,047**
Indirect Costs										
Research and development costs	$1,575	$30,198	$1,845	$35,382	$1,456	$27,931	$1,074	$20,593	$835	$16,022
Step 2 warranty	$5,588	$8,873	$6,547	$10,396	$5,168	$8,207	$3,810	$6,051	$2,965	$4,707
Useful Life extension	$4,543	$6,157	$5,323	$7,214	$4,202	$5,695	$3,098	$4,199	$2,410	$3,267
Compliance program costs	$120	$2,309	$141	$2,705	$111	$2,135	$82	$1,574	$64	$1,225
Total Indirect Costs	**$11,826**	**$47,537**	**$13,856**	**$55,697**	**$10,938**	**$43,968**	**$8,064**	**$32,416**	**$6,274**	**$25,221**
Total Cost Increase per Unit	**$17,234**	**$60,820**	**$20,192**	**$71,261**	**$15,940**	**$56,254**	**$11,752**	**$41,474**	**$9,143**	**$32,268**
EOEM Costs ($M)										
Direct manufacturing costs										
Engine	$289	$39	$338	$45	$267	$36	$197	$26	$153	$21
Aftertreatment	$485	$62	$569	$72	$449	$57	$331	$42	$258	$33
Vehicle + On-Board Diagnostics	$50	$5	$59	$6	$46	$5	$34	$3	$27	$3
Total Direct Costs	**$824**	**$106**	**$965**	**$124**	**$762**	**$98**	**$562**	**$72**	**$437**	**$56**
Indirect Costs										
Research and development costs	$240	$240	$281	$281	$222	$222	$164	$164	$127	$127
Step 2 warranty	$851	$70	$997	$83	$787	$65	$580	$48	$452	$37
Useful Life warranty	$692	$49	$811	$57	$640	$45	$472	$33	$367	$26
Compliance program costs	$18	$18	$21	$21	$17	$17	$13	$13	$10	$10
Total Indirect Costs	**$1,802**	**$378**	**$2,111**	**$442**	**$1,666**	**$349**	**$1,228**	**$258**	**$956**	**$200**
Total Cost Increase ($M)	**$2,625**	**$483**	**$3,076**	**$566**	**$2,428**	**$447**	**$1,790**	**$329**	**$1,393**	**$256**

Source: ACT Research Co., LLC: Copyright 2020

Heavy-Heavy Duty MY2031. We also estimate (in Table 6) that the additional low-NO_x requirements for MY2031, using the MY2027 proposals as a baseline, would cost HHD truck manufacturers an additional $4.0 billion on a national level, or $14,830 per-unit, in 2019 dollars. For California, our estimate of $275 million in costs equates to $18,150 per-unit. While there may be modest aftertreatment changes associated with the MY2031 step, there are no additional engine or on-board diagnostics requirements. The costs at issue are almost exclusively related to further extensions to the emissions warranty and useful life periods. On an inflation-adjusted and discounted basis, using the 3% and 7% discount rates recommended by EPA and OMB, **the net present value cost ranges from $8,350 – $13,200 per HHD vehicle, for a total of $2.2 – $3.5 billion for the HHD industry at the national level. For California, we estimate the MY2031 proposed requirements would increase the cost of a HHD truck by $10,220 – $16,140**. Note again that in the far-right column, we present the cost figures discounted at the 10% WACC. These costs are considerably lower and, again, could better reflect the uncertainties relating to the possible incorporation of unforeseen technology improvements in the coming years.

Table 6: Cost Estimates to Meet Proposed Combined MY2031 Standards for HHD Vehicles

Heavy-heavy Duty Diesel										
Social Cost Methodology	MY2031 - from MY2027 baseline								*Private Cost (not Social)*	
Costs to Develop & Build Ultra-Low-NOx products	2019 dollars		Inflation-adjusted at:		Discounted at:		Discounted at:		Discounted at WACC	
			2%		3%		7%		10%	
	National	California	National	California	National	California	National	California	National	California
Industry Units	267,135	15,098	267,135	15,098	267,135	15,098	267,135	15,098	267,135	15,098
Per unit costs ($)										
Direct manufacturing costs										
Engine	$0	$0	$0	$0	$0	$0	$0	$0	$0	$0
Aftertreatment	$278	$267	$353	$339	$248	$238	$157	$150	$108	$103
Vehicle + On-Board Diagnostics	$0	$0	$0	$0	$0	$0	$0	$0	$0	$0
Total Direct Costs	**$278**	**$267**	**$353**	**$339**	**$248**	**$238**	**$157**	**$150**	**$108**	**$103**
Indirect Costs to Manufacturers										
Research and development costs	$16	$301	$20	$382	$14	$268	$9	$169	$6	$116
Warranty on new technology	$0	$0	$0	$0	$0	$0	$0	$0	$0	$0
Warranty Step 2	$4,729	$5,243	$5,997	$6,649	$4,206	$4,663	$2,663	$2,952	$1,827	$2,026
Useful Life extension	$9,810	$12,336	$12,441	$15,645	$8,726	$10,973	$5,524	$6,947	$3,791	$4,767
Compliance program costs	$0	$0	$0	$0	$0	$0	$0	$0	$0	$0
Total Indirect Costs	**$14,554**	**$17,880**	**$18,458**	**$22,676**	**$12,946**	**$15,904**	**$8,196**	**$10,068**	**$5,624**	**$6,909**
Cost Increase per Unit ($)	**$14,833**	**$18,147**	**$18,811**	**$23,014**	**$13,194**	**$16,142**	**$8,352**	**$10,219**	**$5,732**	**$7,013**
EOEM Costs ($M)										
Direct manufacturing costs										
Engine	$0	$0	$0	$0	$0	$0	$0	$0	$0	$0
Aftertreatment	$74	$4	$94	$5	$66	$4	$42	$2	$29	$2
Vehicle + On-Board Diagnostics	$0	$0	$0	$0	$0	$0	$0	$0	$0	$0
Total Direct Costs	**$74**	**$4**	**$94**	**$5**	**$66**	**$4**	**$42**	**$2**	**$29**	**$2**
Indirect Costs										
Research and development costs	$4	$5	$5	$6	$4	$4	$2	$3	$2	$2
Warranty on new technology	$0	$0	$0	$0	$0	$0	$0	$0	$0	$0
Warranty Step 2	$1,263	$79	$1,602	$100	$1,124	$70	$711	$45	$488	$31
Useful Life extension	$2,621	$186	$3,323	$236	$2,331	$166	$1,476	$105	$1,013	$72
Compliance program costs	$0	$0	$0	$0	$0	$0	$0	$0	$0	$0
Total Indirect Costs	**$3,888**	**$270**	**$4,931**	**$342**	**$3,458**	**$240**	**$2,189**	**$152**	**$1,502**	**$104**
Total Cost Increase ($M)	**$3,962**	**$274**	**$5,025**	**$347**	**$3,525**	**$244**	**$2,231**	**$154**	**$1,531**	**$106**

Source: ACT Research Co., LLC: Copyright 2020

Medium-Heavy Duty MY2031. We estimate (in Table 7) that the Omnibus Requirements proposed for MY2031 would cost MHD truck and engine makers an additional $1.0 billion on a national level, or $6,550 per-unit. For California, the projected $100 million cost increase equates to $17,560 per-unit. As noted above in the *Market Sizing* section, we assume a smaller diesel-powered market size in California in 2031 due to the implementation of CARB's ZEV rules. **The net present value of these costs (using the 3% and 7% discount rates) is $615 – $935 million for the MHD industry on a nationwide basis, or $3,700 – $5,800 per MHD vehicle, and $60 – $90**

million in California, or $9,900 – $15,600 per vehicle. The costs were largely similar to the estimates calculated for HHD diesel vehicles. While smaller in absolute terms, they represent similar proportional price increases.

Table 7: Cost Estimates to Meet Proposed Combined MY2031 Standards for MHD Vehicles

Medium-heavy Duty Diesel										
Social Cost Methodology				MY2031 - from MY2027 baseline					*Private Cost (not Social)*	
Costs to Develop & Build Ultra-Low-NOx products	2019 dollars		Inflation-adjusted at:		Discounted at:		Discounted at:		Discounted at WACC	
Phase 1, part 1			2%		3%		7%		10%	
	National	California	National	California	National	California	National	California	National	California
Units	158,526	5,511	158,526	5,511	158,526	5,511	158,526	5,511	158,526	5,511
Per unit costs ($)										
Direct manufacturing costs										
Engine	$0	$0	$0	$0	$0	$0	$0	$0	$0	$0
Aftertreatment	$0	$0	$0	$0	$0	$0	$0	$0	$0	$0
Vehicle + On-Board Diagnostics	$0	$0	$0	$0	$0	$0	$0	$0	$0	$0
Total Direct Costs	**$0**	**$0**	**$0**	**$0**	**$0**	**$0**	**$0**	**$0**	**$0**	**$0**
Indirect Costs										
Research and development costs	$158	$4,537	$200	$5,753	$140	$4,035	$89	$2,555	$61	$1,753
Step 2 warranty	$3,219	$7,049	$4,083	$8,940	$2,864	$6,271	$1,813	$3,970	$1,244	$2,724
Useful Life extension	$3,174	$5,978	$4,026	$7,582	$2,823	$5,318	$1,787	$3,366	$1,227	$2,310
Compliance program costs	$0	$0	$0	$0	$0	$0	$0	$0	$0	$0
Total Indirect Costs	**$6,551**	**$17,564**	**$8,308**	**$22,276**	**$5,827**	**$15,624**	**$3,689**	**$9,891**	**$2,532**	**$6,788**
Total Cost Increase per Unit	**$6,551**	**$17,564**	**$8,308**	**$22,276**	**$5,827**	**$15,624**	**$3,689**	**$9,891**	**$2,532**	**$6,788**
EOEM Costs ($M)										
Direct manufacturing costs										
Engine	$0	$0	$0	$0	$0	$0	$0	$0	$0	$0
Aftertreatment	$0	$0	$0	$0	$0	$0	$0	$0	$0	$0
Vehicle + On-Board Diagnostics	$0	$0	$0	$0	$0	$0	$0	$0	$0	$0
Total Direct Costs	**$0**	**$0**	**$0**	**$0**	**$0**	**$0**	**$0**	**$0**	**$0**	**$0**
Indirect Costs										
Research and development costs	$25	$25	$32	$32	$22	$22	$14	$14	$10	$10
Step 2 warranty	$510	$39	$647	$49	$454	$35	$287	$22	$197	$15
Useful Life warranty	$503	$33	$638	$42	$448	$29	$283	$19	$194	$13
Compliance program costs	$0	$0	$0	$0	$0	$0	$0	$0	$0	$0
Total Indirect Costs	**$1,039**	**$97**	**$1,317**	**$123**	**$924**	**$86**	**$585**	**$55**	**$401**	**$37**
Total Cost Increase ($M)	**$1,039**	**$97**	**$1,317**	**$123**	**$924**	**$86**	**$585**	**$55**	**$401**	**$37**

Source: ACT Research Co., LLC: Copyright 2020

Pre-Buy/No-Buy Analysis

Introduction. A "pre-buy" occurs when industry participants initially reject a regulation-driven change in a product, in this case heavy-duty on-highway commercial vehicles, and instead buy as much of that product as possible in the years before the new regulation takes effect. A "no-buy" occurs in the initial years after the new regulation is implemented, when product demand, while not literally zero, falls sharply. The trucking industry is naturally risk-averse and prone to avoid new regulations that may impact the reliability and operating costs of trucks, since operational reliability is so vital to industry participants' ability to survive in an historically low-margin business.

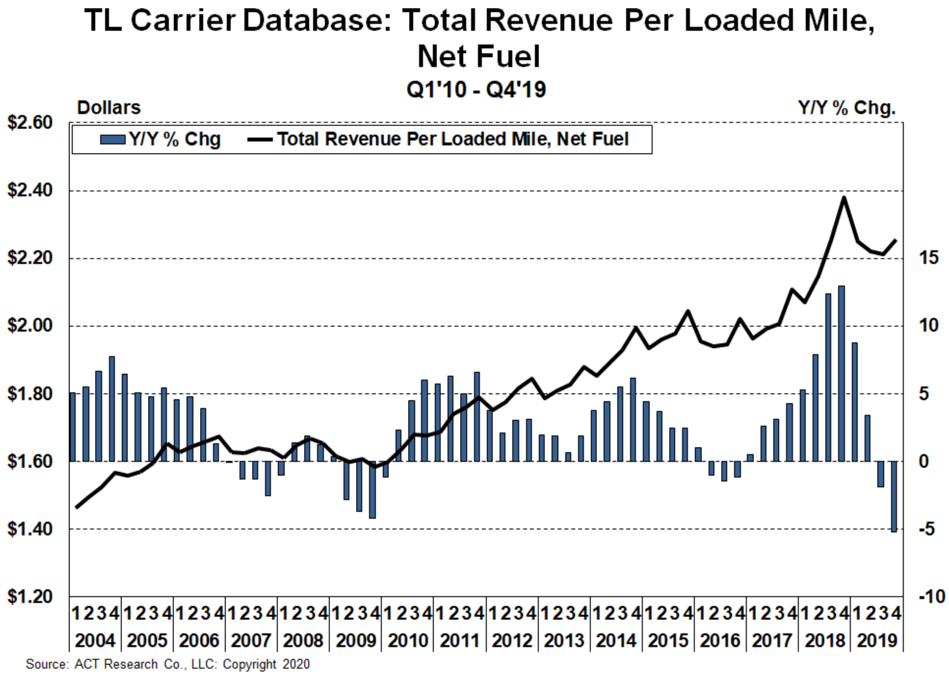
The base case of our cost study uses a hypothetical market size which takes a trailing five-year average and scales it up by a 1% CAGR. This borrows from the established assumption that freight volume per capita is very stable in the long-run, so freight grows roughly in line with population growth. It also borrows from our view that truck supply and demand always return to equilibrium, notwithstanding intermittent periods of over and under supply relative to freight demand. Based on our cost study, we estimate that HHD truck prices are likely to rise $18k-$24k (14%-18%) in MY2027, and another $8k-$13k (5%-8%) in MY2031. MHD truck prices are likely to rise $12k-$16k in MY2027, and another $4k-$6k in MY2031, with similar percentages, as a result of the proposed Omnibus Regulations.

There is not a great deal of pricing information available in the new MHD and HHD truck markets, though information on freight rates has improved significantly in recent years, so partial equilibrium analysis not very effective for the manufacturing sector, but perhaps better for the trucking industry. And since the costs of the proposed regulations will be passed to the trucking industry, it is those effects which we believe are most important to consider.

Past experience, particularly the pre-buy that occurred in 2005-2006 ahead of EPA07, demonstrates that emissions standards which significantly increase the cost and complexity of HHD tractors are likely to lead to pre-buying of equipment in the years leading up to the regulations, assuming the industry has the financial wherewithal to adjust the timing of capital expenditures. And given the lower tax rates as of 2018, we think the industry is structurally more profitable, or at least it has not been adversely impacted. Therefore, the trucking industry likely will have the ability to pre-buy in advance of the Omnibus Regulations taking effect.

Starting from the experience in 2006-2007, the trend in contract truckload rates, which fell 1.3% in 2007, has risen 3% per-year on average since then. That amounts to a 4%-type opportunity cost for the industry. (See chart below.)

Exhibit A



With that opportunity cost in mind, we believe the proposed Omnibus Regulations would precipitate the largest-ever pre-buy for medium-heavy and heavy-heavy duty trucks and tractors. The primary repercussions of a pre-buy would be two years of vehicle underproduction in 2027 and 2028 to counterbalance the likely overproduction in 2025 and 2026. While we can make a case that R&D costs are ultimately recouped over time thanks to higher vehicle prices, not all costs are recoverable. There would be significant costs for the OEMs and their employees in terms of the inefficiencies that come with a rapid ramp-up to meet an artificial demand bubble followed by a demand collapse in the period of capacity rebalancing that leads to layoffs and production cuts.

While the vehicle and engine manufacturers will have to handle major market disruptions relating to nonmarket-driven demand impacts, the HHD market has an additional constituency that likely will be severely impacted by the proposed rule-making. The anticipated pre-buy, like the one that occurred ahead of EPA'07 in 2005–2006, is likely to result in significant and unnecessary capacity additions in the HHD trucking industry. A large portion of those truckers operates on a for-hire basis and is dependent upon market rates to move freight. The lower freight rates which will inevitably result from the regulation-driven overcapacity bubble will have a significant adverse financial impact on the nation's truckers, **with an estimated impact of $6.5 – $8.6 billion at net present value.**

Pre-Buy Model. Using a multi-factor relational model based on a significant history of industry activity before and after the introduction of new emissions regulations, **we estimate (in Table 8) the industry will pre-buy 64,800 (4,200 + 60,600) additional HHD tractors and 25,300 (2,600 + 22,700) MHD vocational trucks in 2025 – 2026 ahead of the MY2027 regulations. This adds up to 90,100 total Class 8 vehicles over the two-year pre-buy. Ahead of the MY2031 standards, we estimate another pre-buy of 35,000 (4,200 + 30,700) HHD tractors and 11,600 (2,300 + 9,200) HHD vocational trucks in 2029 – 2030.** Vocational trucks are similar to MHD vehicles in that they are typically a component of a job (construction/dump/cement) and are not directly subject to market rates, so the modeled freight rate effects exclude vocational trucks. Overcapacity in MHD vocational trucks will primarily impact manufacturers who will have to lay off workers and lower supplier orders. However, in the HHD tractor market, there likely will be very significant price impacts on freight rates.

Table 8: Prebuy Size Estimates in Units and Percent

	MY2027 $ Change Op. Costs	MY2027 % Change Op. Costs	Anticipated Prebuy: 2025	Share of new Market	Anticipated Prebuy: 2026	Share of new Market
US Class 8 Tractor	$ 35,103	18.3%	4,219	2.7%	60,622	39.9%
US Class 8 Vocational	$ 35,190	14.6%	2,620	4.7%	22,667	36.9%
US Total Class 8			6,838	3.2%	83,290	39.0%
Source: ACT Research Co.,LLC: Copyright 2020						

	MY2031 $ Change Op. Costs	MY2031 % Change Op. Costs	Anticipated Prebuy: 2029	Share of new Market	Anticipated Prebuy: 2030	Share of new Market
US Class 8 Tractor	$ 12,491	6%	4,234	2%	26,717	13%
US Class 8 Vocational	$ 14,536	6%	2,344	4%	9,236	14%
US Total Class 8			6,578	3%	35,953	14%
Source: ACT Research Co.,LLC: Copyright 2020						

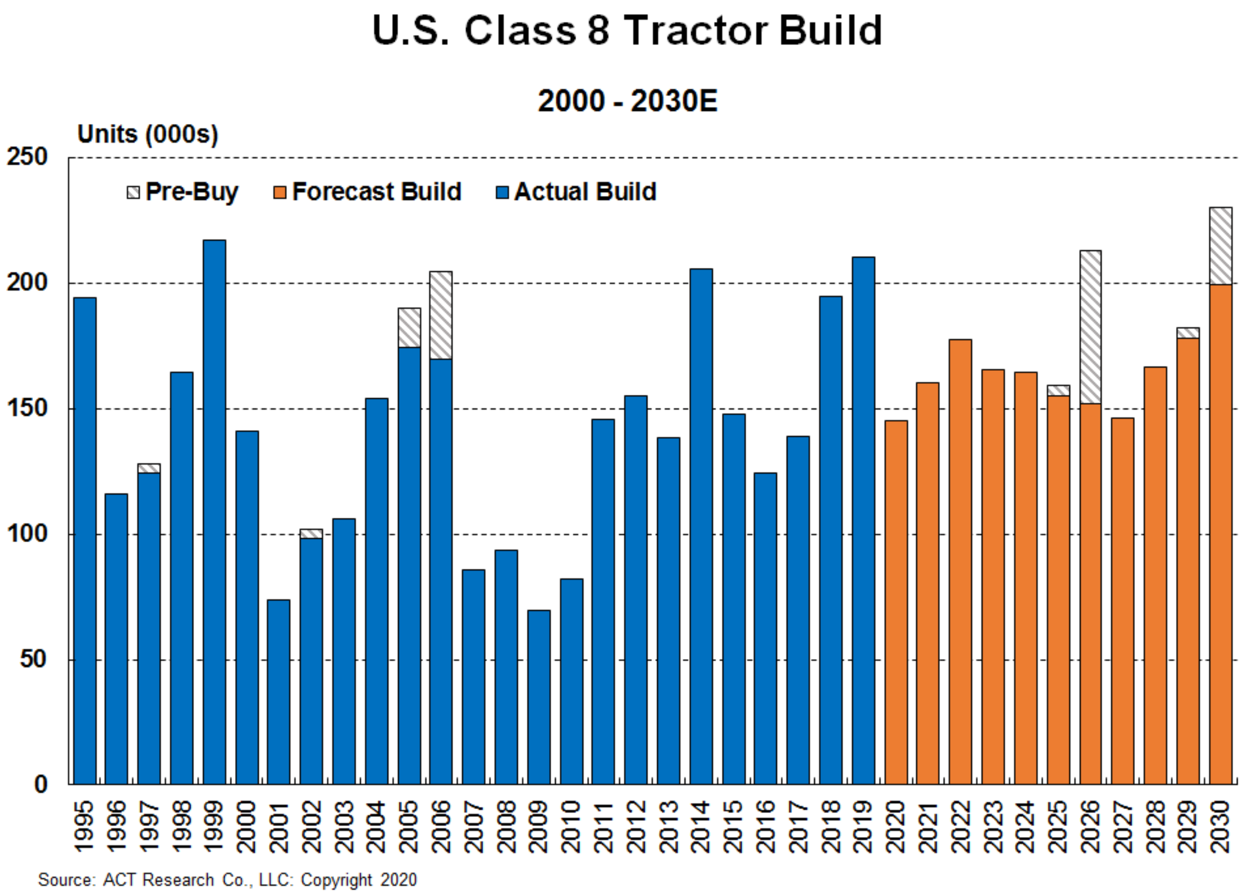
The HHD tractor pre-buy model starts with the base tractor price, adds in the 12% Federal Excise Tax (FET) and an average 8% for State and Local taxes. We then raise the sticker price by the cost of meeting the proposed standards, using $23,885 (18% of base), which we settled on because that cost increase was near the center of the range of the $30,300 per-unit value undiscounted at the 2% inflation rate, and the $17,600 per-unit value using a 7% discount rate. We taxed the $23,885 at the FET + state tax rate, added in three years of insurance at a rate of 5% of the truck cost each year, and added financing costs at an interest rate of 5% for half of the value of the vehicle. This totals about $35,000 of added upfront costs for the HHD vehicle purchaser in MY2027, and another $12,000 in MY2031. (See Table 8.)

Fuel economy considerations all play a role in the model. After considerable discussion, we included the impending fuel economy improvements associated with GHG-2 regulations in MY2027, even though most of those fuel economy improvements will be in effect prior to the Omnibus Regulations. In our cost analysis from the manufacturers' perspective, we did not include costs or benefits for the GHG-2 regulations, except as we understand the state of the market to be in MY2027. To estimate the social cost to the trucking industry, however, our model's purpose is to reflect the conditions impacting the industry in MY2027 and MY2031. We considered both the improvements in fuel efficiency and additional use of diesel emissions fluid (DEF), finding that the 4% improvement in fuel efficiency expected in MY2027 from GHG-2 regulations would more than offset a doubling of the DEF dosing rate. Moving from a 2.5% to a 5% DEF dosing rate on a 90,000 mile per-year truckload application would use 233 additional gallons per-year at a cost of about $665, but the 4% fuel efficiency improvement saves $1,300 per-year at 440 gallons in this application. We are not using those estimates as benefits relating to the Omnibus Regulations, but rather to refine our analysis of the potential magnitude of a pre-buy.

Regarding maintenance costs, some of the technology solutions anticipated for the proposed Omnibus Regulations are targeted towards improving the durability of aftertreatment systems, which could have the effect of lowering maintenance expenses in some instances. However, the overall increase in the complexity of the engine and aftertreatment systems likely will require more frequent maintenance for these trucks through their life-cycles, not less. Given the high degree of uncertainty, however, we have not included explicit estimates of maintenance expenses, except to say that there are positives and negatives from a fleet perspective, and as noted earlier in our report, the higher warranty and useful life costs are included in the estimated sticker price increases.

Tractor Pre-Buy. The sum of the multiple costs result in a "willingness to buy" factor, which is the percentage change in total cost of ownership (TCO) of the vehicle before and after the regulation. At a cost of $35,100 in MY2027, the net TCO impact is 18% of the pre-regulation purchase price. Based on historical pre-buys and assuming reasonable industry profit margins leading into the new regulatory mandates, we estimate that the 18% increase will drive an additional 3% of HHD tractor sales in 2025 (4,200 units), and a 40% pre-buy in 2026 (60,600 units). The $12,500 net TCO increase due to the proposed MY2031 standards, which amounts to an additional 6% price/TCO increase, will drive another 2% of tractor sales in 2028 (4,200 units) and an additional 15% pre-buy in 2029 (30,700 units). (See Table 8.)

Table 9: Retail Sales and Pre-Buy History and Forecast in the U.S. Class 8 Tractor Market



Freight Rate Impact. Adding these 65,000 "pre-bought" tractors into our population models, where we estimate 1.4 million HHD tractors engaged in truckload and/or less-than-truckload freight hauling, amounts to a 4.5% increase in capacity or supply into the industry. Our freight pricing models indicate that the sensitivity of truckload contract pricing is roughly -64% relative to capacity additions when modeled econometrically with demand and regulatory factors included. In other words, a 1% increase in freight-hauling capacity lowers pricing by .64%, so a 4.5% increase in capacity, as expected in this case, would lower truckload pricing by 2.9%.

Trucking Industry Sizing and Earnings Impact. According to the U.S. Census Bureau's Quarterly Services Survey, the U.S. trucking industry is on pace for $195 billion in revenue (NAICS code: 4841, General Freight Trucking) in 2019. Using a trailing 5-year industry growth rate of 3% to extrapolate to 2026, the industry should be generating $240 billion of revenue in 2026. A 2.9% pricing impact on a $240 billion segment of the economy would be a cost to aggregate trucking industry earnings of $6.9 billion on an annual basis, and it would likely last 18-24 months. Thus,

the total impact on the trucking industry would likely be $10.4 – $13.8 billion of lost earnings in 2026 – 2027. This discounts back to $6.5 - $8.6 billion in 2019 dollars at 7%.

We have focused here on the for-hire market reported on by the Census Bureau. Our estimates do not include effects on the private fleet segment of the trucking industry, which makes up just over half of the tractor fleet, but generally hauls freight for a single company. Private fleets are generally a cost center inside companies that ship goods, with few booking revenue for their services. As a result, we did not include that part of the market in estimating financial impacts.

Vocational Pre-buy. The main focus of our analysis (in Table 8) is on the tractor portion of the heavy-duty Class 8 market, since, over the past decade, tractors have represented 75% of the Class 8 vehicles sold in the US, compared to 25% for the Class 8 market's vocational segment. Significantly higher miles traveled per-year for tractors mean shorter lengths of ownership due to reliability/downtime issues as miles accrue. On the vocational side of the market, localized vocational applications (P&D, construction, government) mean fewer miles per-year and longer first-buyer ownership. And, as previously discussed, unlike the tractor market, where every vehicle is a profit center, the vocational truck is often a tool used to facilitate a non-transportation related business. Thus, there is significantly more volatility in US tractor demand from year to year compared to the vocational truck portion of the market.

In that regard, like the MHD market, we do not typically view the vocational portion of the HHD market as a candidate for pre-buying. But in terms of vocational equipment pre-buying ahead of EPA07, ACT's modeling suggests that a prebuy did occur ahead of that regulatory mandate. Vocational buyers and dealers accounted for 30% of the 92,000 units of prebuying that occurred in 2005 and 2006, or 5 percent higher than the segment's long-run market share. We have concluded that the majority of that prebuy resulted from vocational fleet buyers actively working to avoid the EPA07 emissions mandate.

Using our model, the sharp rise in vehicle costs ahead of the MY2027 mandates in this case indicates that vocational truck buyers will pre-buy approximately 26,000 units in 2025 and 2026. (See Table 8.) At $35,200 in MY2027, the net TCO impact is 15% of the pre-regulation purchase price. That includes a $24,000 price increase, plus taxes, insurance, financing and diesel emissions fluid costs. The net result is that we estimate that the increased costs will drive an additional 5% of vocational tractor sales in 2025 (2,600 units) and a 37% pre-buy in 2026 (22,700 units), which totals to a pre-buy of 25,300 units. For the MY2031 mandate step, the model projects another 4% pre-buy in 2029 (2,300 units) with an additional 14% pre-buy in 2030 (9,200 units) due to a $14,500 net TCO increase for the MY2031 proposed standards, which amounts to an additional 6% price/TCO increase. Combined, the MY2031 vocational Class 8 prebuy sums to 11,600 units.

When combined, the projected US Class 8 prebuy for trucks and tractors rises to 90,100 units ahead of the MY2027 regulatory step, with 6,800 units pulled into 2025 and 83,300 units pulled into 2026. The prebuy represents a 3% increase above modeled 2024 demand and a 39% jump above modeled levels in 2025. **For the MY2031 mandate, the model anticipates 6,600 units being pulled into 2029, and an additional pre-buy of 39,900 Class 8 units in 2030.** Prebuying as a percentage of the market is 3% in 2028 and 15% in 2029.

Sensitivity Analysis: Costs Using Pre-buy/No-buy Scenario. The tables below (Tables 10-11) provide a sensitivity analysis from the base case costs of the Omnibus Regulations (see Tables 4-7) which assumed a normalized demand environment. Having established that a normalized demand environment is very unlikely, we show below how the cost estimates change when we envision the significantly depressed post-pre-buy market in MY2027 that we think is more likely. In short, the total costs to the manufacturers fall significantly because most of the costs vary with production levels, but the per-unit costs rise because some of those costs are fixed, mainly R&D and compliance program costs.

For HHD vehicles in MY2027 (see Table 10), these industry Total Cost Increase figures are approximately 52% lower than the National costs presented in the base case discussed earlier in this report, and 53% lower on a California basis. (See Tables 4-7.) That is primarily because of a 38% lower vehicle-build forecast.

However, on a per-unit basis, the MY2027 costs are approximately 3% and 31% higher on a National and California-only basis, respectively. Those percentages are consistent across inflation and discount rates.

Table 10: Cost Estimates Under No-buy MY2027 Scenario for HHD Vehicles

Heavy-heavy Duty Diesel										
Social Cost Methodology				**MY2027 - from MY2018 baseline**					*Private Cost (not Social)*	
Costs to Develop & Build Ultra-Low-NOx products	2019 dollars		Inflation-adjusted at:		Discounted at:		Discounted at:		Discounted at WACC	
			2%		3%		7%		10%	
	National	California	National	California	National	California	National	California	National	California
Units	175,004	10,763	175,004	10,763	175,004	10,763	175,004	10,763	175,004	10,763
Per unit costs ($)										
Direct manufacturing costs										
Engine	$3,157	$3,833	$3,699	$4,491	$2,920	$3,545	$2,153	$2,614	$1,675	$2,034
Aftertreatment	$4,589	$6,209	$5,376	$7,274	$4,244	$5,742	$3,129	$4,234	$2,434	$3,294
Vehicle + On-Board Diagnostics	$176	$1,990	$206	$2,331	$163	$1,840	$120	$1,357	$93	$1,056
Total Direct Costs	**$7,921**	**$12,031**	**$9,281**	**$14,097**	**$7,327**	**$11,128**	**$5,402**	**$8,204**	**$4,203**	**$6,383**
Indirect Costs to Manufacturers										
Research and development costs	$3,687	$52,808	$4,319	$61,873	$3,410	$48,843	$2,514	$36,011	$1,956	$28,017
Warranty on new technology	$1,844	$2,070	$2,161	$2,426	$1,706	$1,915	$1,258	$1,412	$978	$1,098
Warranty Step 2	$3,311	$3,827	$3,880	$4,484	$3,063	$3,539	$2,258	$2,609	$1,757	$2,030
Useful Life extension	$9,451	$11,283	$11,074	$13,220	$8,742	$10,436	$6,445	$7,694	$5,014	$5,986
Compliance program costs	$261	$4,223	$306	$4,948	$241	$3,906	$178	$2,880	$138	$2,241
Total Indirect Costs	**$18,554**	**$74,212**	**$21,739**	**$86,951**	**$17,161**	**$68,640**	**$12,653**	**$50,606**	**$9,844**	**$39,373**
Cost Increase per Unit ($)	**$26,476**	**$86,243**	**$31,020**	**$101,048**	**$24,488**	**$79,768**	**$18,054**	**$58,811**	**$14,047**	**$45,756**
EOEM Costs ($M)										
Direct manufacturing costs										
Engine	$552	$41	$647	$48	$511	$38	$377	$28	$293	$22
Aftertreatment	$803	$67	$941	$78	$743	$62	$548	$46	$426	$35
Vehicle + On-Board Diagnostics	$31	$21	$36	$25	$28	$20	$21	$15	$16	$11
Total Direct Costs	**$1,386**	**$129**	**$1,624**	**$152**	**$1,282**	**$120**	**$945**	**$88**	**$735**	**$69**
Indirect Costs										
Research and development costs	$645	$568	$756	$666	$597	$526	$440	$388	$342	$302
Warranty on new technology	$323	$22	$378	$26	$299	$21	$220	$15	$171	$12
Warranty Step 2	$579	$41	$679	$48	$536	$38	$395	$28	$307	$22
Useful Life extension	$1,654	$121	$1,938	$142	$1,530	$112	$1,128	$83	$878	$64
Compliance program costs	$46	$45	$53	$53	$42	$42	$31	$31	$24	$24
Total Indirect Costs	**$3,247**	**$799**	**$3,804**	**$936**	**$3,003**	**$739**	**$2,214**	**$545**	**$1,723**	**$424**
Total Cost Increase ($M)	**$4,633**	**$928**	**$5,429**	**$1,088**	**$4,285**	**$859**	**$3,160**	**$633**	**$2,458**	**$492**

Source: ACT Research Co., LLC: Copyright 2020

For MY2031 (see Table 11), and calculated off the MY2027 baseline, the per-unit costs rise 4% and 5%, respectively, for the National and California-only programs under the lower no-buy demand scenario. Those respective percentage increases are closer together because the MY2031 costs are largely variable outside of R&D. On an aggregate basis, the lower vehicle-production assumptions would reduce the total costs of the program by 28% for both a National and a California program, due to the 32% lower vehicle-build forecast.

Table 11: Cost Estimates Under No-buy MY2031 Scenario for HHD Vehicles

Heavy-heavy Duty Diesel										
Social Cost Methodology				MY2031 - from MY2027 baseline					*Private Cost (not Social)*	
Costs to Develop & Build Ultra-Low-NOx products	2019 dollars		Inflation-adjusted at:		Discounted at:		Discounted at:		Discounted at WACC	
			2%		3%		7%		10%	
	National	California	National	California	National	California	National	California	National	California
Units	182,540	10,317	182,540	10,317	182,540	10,317	182,540	10,317	182,540	10,317
Per unit costs ($)										
Direct manufacturing costs										
Engine	$0	$0	$0	$0	$0	$0	$0	$0	$0	$0
Aftertreatment	$290	$302	$367	$383	$258	$269	$163	$170	$112	$117
Vehicle + On-Board Diagnostics	$0	$0	$0	$0	$0	$0	$0	$0	$0	$0
Total Direct Costs	**$290**	**$302**	**$367**	**$383**	**$258**	**$269**	**$163**	**$170**	**$112**	**$117**
Indirect Costs to Manufacturers										
Research and development costs	$16	$313	$21	$397	$15	$279	$9	$176	$6	$121
Warranty on new technology	$0	$0	$0	$0	$0	$0	$0	$0	$0	$0
Warranty Step 2	$4,921	$5,512	$6,241	$6,991	$4,377	$4,903	$2,771	$3,104	$1,902	$2,130
Useful Life extension	$10,208	$12,940	$12,946	$16,411	$9,080	$11,510	$5,748	$7,287	$3,945	$5,001
Compliance program costs	$0	$0	$0	$0	$0	$0	$0	$0	$0	$0
Total Indirect Costs	**$15,145**	**$18,765**	**$19,208**	**$23,799**	**$13,472**	**$16,692**	**$8,528**	**$10,567**	**$5,853**	**$7,252**
Cost Increase per Unit ($)	**$15,435**	**$19,068**	**$19,575**	**$24,182**	**$13,730**	**$16,961**	**$8,692**	**$10,737**	**$5,965**	**$7,369**
EOEM Costs ($M)										
Direct manufacturing costs										
Engine	$0	$0	$0	$0	$0	$0	$0	$0	$0	$0
Aftertreatment	$53	$3	$67	$4	$47	$3	$30	$2	$20	$1
Vehicle + On-Board Diagnostics	$0	$0	$0	$0	$0	$0	$0	$0	$0	$0
Total Direct Costs	**$53**	**$3**	**$67**	**$4**	**$47**	**$3**	**$30**	**$2**	**$20**	**$1**
Indirect Costs										
Research and development costs	$3	$3	$4	$4	$3	$3	$2	$2	$1	$1
Warranty on new technology	$0	$0	$0	$0	$0	$0	$0	$0	$0	$0
Warranty Step 2	$898	$57	$1,139	$72	$799	$51	$506	$32	$347	$22
Useful Life extension	$1,863	$133	$2,363	$169	$1,657	$119	$1,049	$75	$720	$52
Compliance program costs	$0	$0	$0	$0	$0	$0	$0	$0	$0	$0
Total Indirect Costs	**$2,765**	**$194**	**$3,506**	**$246**	**$2,459**	**$172**	**$1,557**	**$109**	**$1,068**	**$75**
Total Cost Increase ($M)	**$2,817**	**$197**	**$3,573**	**$249**	**$2,506**	**$175**	**$1,587**	**$111**	**$1,089**	**$76**

Source: ACT Research Co., LLC: Copyright 2020

Dealer Pre-buy. While we have discussed truckers as the primary drivers of pre-buying, there is another group that is also likely to contribute to pre-buying activity ahead of the MY2027 standard — truck dealers. Based on the experience ahead of EPA'07, we would expect that U.S. MHD and HHD commercial vehicle dealers would likely increase inventory levels aggressively in advance of the proposed MY2027 regulations. Dealers' ability to add to stock, however, would largely be determined by the availability of manufacturers' production capacity. Dealers' pre-buy decisions would be based on several factors:

> First, is the cost of pre- versus post-mandate vehicles. With the sharply higher costs likely for the MY2027 vehicles, having lower priced units in inventory should facilitate dealer sales for several months into the post-mandate period.
>
> Second, given the risks that early post-mandate purchasers might face with respect to the reliability of early post-mandate vehicles, most truckers would prefer to let someone else act as the beta-tester for real-world usage. Dealers carrying pre-mandate

inventories could provide their risk-averse customers with a competitive edge early in the post-mandate period.

Looking back to the last major pre-buy in 2006, MHD and HHD vehicle dealers both added to inventories over the course of that year. Based on ACT Research data collection, MHD inventory levels rose from 49,500 units at the end of December 2005, to 70,500 units at the end of 2006. A baseline 6% year to year increase in MHD Classes 5-7 retail sales in the U.S. does not explain the 42% inventory increase across 2006.

Reviewing changes to HHD vehicle inventories ahead of EPA07, from December 2005 to January 2007, U.S. Class 8 inventories rose from 42,200 units to 54,600 units, a 29% increase compared to a 12% increase in U.S. Class 8 retail sales from 2005 to 2006. Arguably the HHD dealer inventory pre-buy should have been larger in 2006, but final demand from trucking companies in the U.S. and Canada pushed the North American Class 8 manufacturing to unprecedented levels. In 2006, total North American Class 8 production rose to 376,000 units, 31,000 units higher than the second-best year ever, 2019.

Thus, we suspect that, as was the case in 2006, it will not be a lack of desire on the part of dealers to add inventory that limits Class 8 inventory-building ahead of the MY2027 regulation. Rather, it will be strong purchasing demand on the part of truck fleet operators that will limit dealers' ability to acquire and maintain those stocks.

Conclusions. The tables set forth below summarize the results of our cost study.

Table 12: Aggregate Costs, Discounted to NPV at 7%

	National			California		
Dollars in billions	MY2027	MY2031	Total	MY2027	MY2031	Total
Manufacturing Costs	$6.3	$2.8	$9.1	$1.08	$0.21	$1.29
Pre-buy / No-buy Costs	$7.6	$0.0	$7.6	NA	NA	NA
Grand Totals for HHD and MHD	**$13.9**	**$2.8**	**$16.7**	**$1.08**	**$0.21**	**$1.29**
Dollars per unit						
Medium-heavy duty	$11,752	$3,689	$15,441	$41,474	$9,891	$51,365
Heavy-heavy duty	$17,610	$8,352	$25,963	$47,686	$10,219	$57,905
Grand Totals for HHD and MHD	**$15,429**	**$6,616**	**$22,044**	**$45,607**	**$10,131**	**$55,738**

Our results show that on a nationwide base, using a 7% discount rate, the Omnibus Regulations will yield per-vehicle cost increases for HHD vehicles totaling $26,000 ($17,600 in 2027, and $8,400 in 2031), and per-vehicle cost increases for MHD vehicles totaling $15,400 ($11,800 in 2027, and $3,700 in 2031). The aggregate costs to the industry will be $16.7 billion ($13.9 billion in 2027, and $2.8 billion in 2031). This consists of $9.1 billion of manufacturing costs ($6.3 billion in 2027, and $2.8 billion in 2031) and $7.6 billion of pre-buy/no-buy costs (all focused on 2027) on the trucking industry.

On a California-only basis, our results show, again using a 7% discount rate, that the Omnibus Regulations will yield per-vehicle price increase for HHD vehicles totaling $57,900 ($47,700 in 2027, and $10,200 in 2031), and per-vehicle price increases for MHD vehicles totaling $51,400 ($41,500 in 2027, and $9,900 in 2031). The aggregate cost to the vehicle and engine manufacturing industry will be $1.35 billion ($1.14 billion in 2027, and $0.22 billion in 2031).

All in, the aggregate cost to the vehicle and engine manufacturing industry from the Omnibus Regulations, not including the additional costs to vehicle purchasers and operators would be $9.1 billion, and the lost earnings for the trucking industry would be $7.6 billion, bringing the total cost to $17.1 billion. Those very significant cost impacts call into question whether the Omnibus Regulations could be cost-effective, especially on a nationwide basis.

Exhibit B

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Potential Air Quality Benefits of a 90%/50% Reduction in NO_x Emissions from New Heavy-Duty On-Highway Vehicles

– Technical Details of Analysis and Assumptions

Prepared for the Truck and Engine Manufacturers Association

April 2020

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Contents

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List of Acronyms

ACE	Affordable Clean Energy
BCA	Benefit-Cost Analysis
BenMAP	Benefits Mapping and Analysis Program
CAMx	Comprehensive Air Quality Model with Extensions
C-R	Concentration-Response
EMA	Truck and Engine Manufacturer's Association
EPA	Environmental Protection Agency
FTP	Federal Test Procedure
GVWR	Gross Vehicle Weight Rating
HDOH	Heavy-Duty On-Highway
HHD	Heavy Heavy-Duty Vehicle; Class 8a and 8b Trucks (GVWR > 33,000 lbs)
HHDDV	Heavy Heavy-Duty Diesel Vehicle
LHD<=14k	Light Heavy-Duty Vehicle; Class 2b Trucks with 2 Axles and at least 6 Tires or Class 3 Trucks (8,500 lbs < GVWR <= 14,000 lbs)
LHD45	Light Heavy-Duty Vehicle; Class 4 and 5 Trucks (14,000 lbs < GVWR <= 19,500 lbs)
LHDDV	Light Heavy-Duty Diesel Vehicle
LML	Lowest Measured Level
MHD	Medium Heavy-Duty Vehicle; Class 6 and 7 Trucks (19,500 lbs < GVWR <= 33,000 lbs)
MHDDV	Medium Heavy-Duty Diesel Vehicle
MOVES2014	Motor Vehicle Emission Simulator 2014
NAAQS	National Ambient Air Quality Standards
NERA	NERA Economic Consulting
NOx	Nitrogen Oxides
OMB	Office of Management and Budget
$PM_{2.5}$	Fine Particulate Matter (that have a diameter of less than 2.5 micrometers)
RIA	Regulatory Impact Analysis

I. Introduction

This report provides a description of the data, assumptions and modeling that NERA conducted in its analysis for the Engine and Truck Manufacturers Association (EMA) of the potential per-truck air quality benefits of a possible tightening of the NO_x emissions standard for heavy-duty on-highway (HDOH) trucks. This report serves as a technical supplement to a separate NERA report subtitled *Conceptual Summary of Methods and Key Results* (hereafter called the "Summary Report") that provides a policy-oriented discussion of the purpose of the analysis and summarizes key results. In addition to documenting the analysis steps in more technical detail, this report provides a more disaggregated view of the key results. We recommend that one first read the Summary Report, as that contains more general background on the context for this analysis and its policy implications than what is found in this technical documentation.

II. Objective of This Analysis

As discussed in the accompanying Summary Report for this study, past practice of the U.S. Environmental Protection Agency (EPA or the Agency) in implementing Clean Air Act provisions regarding truck emissions standards suggests that any proposal for a tightening of those standards will need to have estimated benefits that exceed its estimated costs. That is usually demonstrated though a benefit-cost analysis (BCA) that is documented in a regulatory impact analysis (RIA) that the Agency must prepare for every major rulemaking. The approach that EPA typically follows in RIAs to estimate national health benefits of regulations affecting ambient air quality such as fine particulate matter ($PM_{2.5}$) and ozone includes several steps:

A. Estimating the incremental emission reductions from implementation of the regulation (and their geographical locations);
B. Estimating the ambient ozone and $PM_{2.5}$ changes across the U.S. as a result of the reduction in emissions;
C. Estimating the population-wide health risk improvements from lower ambient ozone and $PM_{2.5}$ concentrations; and
D. Estimating the societal value in dollars of the estimated health risk improvements – which are referred to as the potential "benefits" of the regulation.

In RIAs, those benefit calculations are typically carried out for a specific future calendar year (usually when the regulation in question is fully implemented) and are compared to estimates of the annualized costs at that point in time.[1] That is a complex and resource-intensive type of analysis that requires specific assumptions about the evolution of markets affected by the regulation (such as the projected future demand for trucking services). Without knowledge of those baseline assumptions, and which specific year will be analyzed, it is not possible to approximate the specific benefits estimates that will be reported in a future RIA. Even if this could be done, the results would provide little insight without a comparable estimate of the total annualized regulatory costs in that particular year – also a complex calculation. However, it is important to develop some rough understanding of the incremental lifecycle cost of a new truck that is likely to pass a RIA's benefit-cost test before anchoring a rulemaking process around a particular degree of stringency. A scoping analysis is therefore valuable to undertake in the

[1] Less frequently, RIAs compute benefits and costs as present values over the duration of the policy implementation period. The analysis we describe in this report is relevant to that type of benefit-cost comparison as well.

preliminary stage of rulemaking, before any specific new standard levels are ready to be proposed. NERA's analysis, documented here, was developed for use in such a scoping exercise.

In developing a simpler analysis method that could produce such scoping-level insights, NERA noted that preliminary information on a new standard's potential cost will be available in the form of its impact on the lifecycle cost per new truck. We also note that if the annual benefits of that new standard will be able to pass a BCA in any future year, then the benefits that each individual truck is likely to provide over its operational lifespan also will need to exceed the incremental costs of that truck, or, at least, that this net benefit condition will be achieved on average over all new trucks. Thus, NERA has prepared an initial scoping analysis that estimates per-truck air quality benefits, focusing on projected benefits that would be attained by trucks sold in 2027, the first year that the anticipated standard would be binding. Thus, we have developed estimates of the present value of benefits over the operating life of an average new truck purchased in 2027 that meets a hypothetical 90% reduction in the NO_x FTP emissions standard. Those per-truck benefits estimates can then be compared to per-truck compliance costs to obtain preliminary insight on whether that particular standard is likely to pass a full BCA.

We emphasize that the estimates we have made in this analysis reflect an effort to anticipate what the Agency would estimate if it applies its own usual assumptions and analysis methodologies. That is, we have used analysis input assumptions that we believe are within the range of those that EPA would likely use. Of course, we do not know what may arise with updated EPA models, data, and input assumptions, but we have sought out the most recent studies and documents on air pollutants that EPA has released. Our estimates are nevertheless subject to revision as more up-to-date information is released. Were we to undertake this type of benefits analysis without regard to what EPA is expected to do, it is likely that we would utilize different methods and assumptions.

III. Overview of Methodology

The process by which we estimate per-truck benefits is summarized in this section. The remaining sections of this report then describe the data, assumptions and models we have used for each step of the process.

First, we calculate the tons of NO_x emissions reductions over time from new trucks that meet the tighter NO_x standard, if purchased in 2027. (We assume all model year 2027 trucks will fully meet the hypothetical 90% FTP standard reduction, which, based on assumptions provided by EMA, will yield 50% reductions in in-use emissions.) Recognizing that some of the new trucks will operate longer than others, we consider the average tons across all new trucks expected to be purchased in 2027 for each year over a potential life of up to 30 years (*i.e.*, through 2057). That calculation is carried out for each of the eight truck types covered by the assumed standard.[2]

Next, the per-truck emissions reductions in each future year are translated into a dollar estimate of each year's health benefits using a simple "reduced-form" method in which the precursor (*e.g.*, NO_x) emissions changes are multiplied by an estimated "benefit-per-ton" value. The result of this methodology is a time line from 2027 through 2057 of annual benefits per truck in each year of the average 2027-vintage truck's operating life.

[2] These eight truck types correspond to regulatory class IDs - 41 (LHD <=14k), 42 (LHD45), 46 (MHD), 47 (HHD), 48 (Urban Bus) and SCCVTypeIDs – 9(LHDDV), 10(MHDDV), 11(HHDDV), 12(Buses) per EPA's emissions inventory model (MOVES2014) documentation (https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P100O7VJ.pdf)

That stream of benefits then is discounted to obtain the present value of benefits per truck for each of the eight truck types. Those eight values are combined into a single sales-weighted average benefit-per-truck estimate.[3] Consistent with OMB and EPA guidance, we provide benefit-per-truck estimates that are calculated using annual discount rates of 3% and 7%. Those values represent our scoping-level estimate of the average lifecycle benefits per truck; they can then be compared to estimates of the incremental per-truck compliance cost to determine whether that anticipated standard is likely to pass a benefit-cost test after a more detailed BCA.

Finally, we calculate how these per-truck benefits are affected by changing the allowed extent of extrapolation from original health effects studies, following an approach that the Agency introduced in a 2019 RIA (EPA, 2019a) which we refer to here as "confidence-weighting."

IV. Calculation of Reduction in Tons Emitted

To obtain estimates of the tons of NO_x reduced per truck, we relied on EPA's mobile source emissions model, MOVES2014. Those calculations were done by truck type and by state for each state of the conterminous U.S. states (excluding the District of Columbia). We used the MOVES2014 data to estimate how long the average truck purchased in 2027 is expected to continue to operate, and to quantify the average operational characteristics of the still-operating trucks as a function of truck age.[4]

Specifically, for each of the eight heavy-duty truck types, we tracked a set of 100 new hypothetical vehicles purchased in 2027 and used the MOVES2014 assumptions regarding the percent of vehicles surviving through each of the next 30 years, the average miles the surviving trucks are driven in each year (which is age-dependent), and their associated baseline (current standard) NO_x emissions.[5] Each year's reduction in tons of NO_x per truck was then calculated as a 50% reduction from the respective year's baseline NO_x emissions (*i.e.*, the sum of baseline NO_x emissions from all operational modes), divided by the number of vehicles surviving in that year. This computation was carried out in each year of the truck's assumed operational life to obtain tons of NO_x reduced per truck by year.

Figure 1 illustrates the resulting estimate of reduction in NO_x emissions for an average model-year 2027 truck in each year of its operational life.[6] Those reductions decline as the trucks age because in each year some of the trucks are removed from service, and trucks that are still in service are used less intensively as they age. The estimated reduction in NO_x emissions per "statistical" vehicle ranges from a low of 0.004 tons at age 30 to a high of 0.054 tons at age 4.

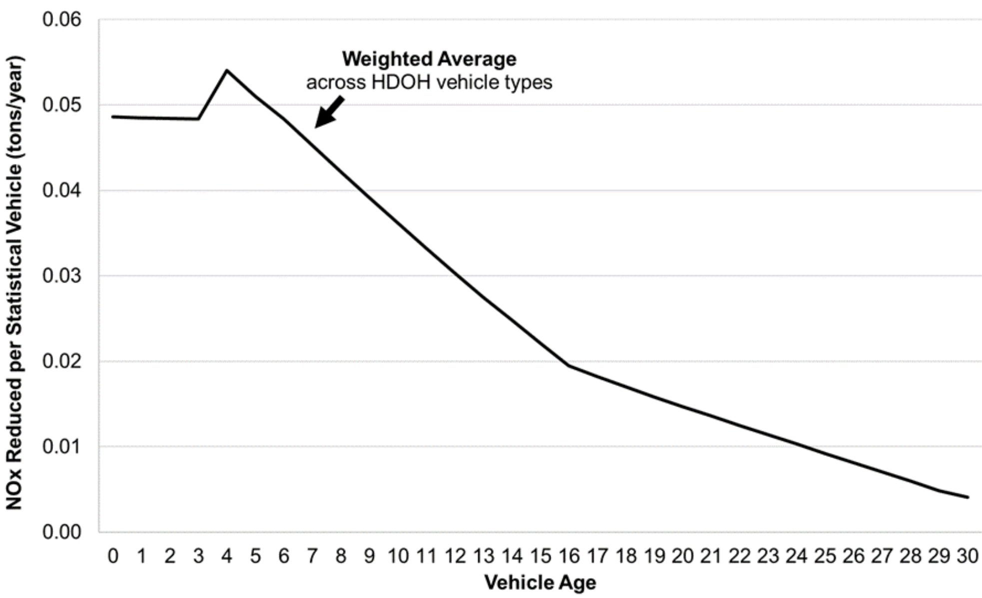
[3] We weighted the present value estimate of the per-truck benefit obtained for each of the eight truck types by the new vehicle sales in 2027 for each of the truck types projected in MOVES2014.

[4] Since the projections for on-road activity and associated baseline NO_x emissions in MOVES2014 extend only until 2050, when the trucks would be 23 years old, we based the survival rates of model-year 2027 trucks to ages of 24 through 30 years on the survival rates to each of those ages assumed in MOVES2014 for model-year 2020 trucks.

[5] The baseline NO_x emissions for each HDOH truck analyzed were calculated for each of the operational modes (running exhaust, start exhaust, extended idle exhaust, and auxiliary power exhaust) which were then summed up to yield the total baseline NO_x emissions. The baseline emissions from running exhaust were calculated using running exhaust emission rates (specified in units of grams of NOx/hr) and the number of hours the truck was operating in running exhaust mode. The baseline emissions from the other operational modes – start exhaust, extended idle exhaust, and auxiliary power exhaust – were calculated using their respective emissions rates (specified in units of grams of NOx/vehicle) and the number of vehicles operating in that year.

[6] The weights used to compute the average across the different HDOH vehicle types analyzed are the projected new vehicle sales for each of the truck types from MOVES2014 in 2027.

Figure 1: NO_x Emissions Reduced per Statistical Vehicle (Average per Year per Vehicle)



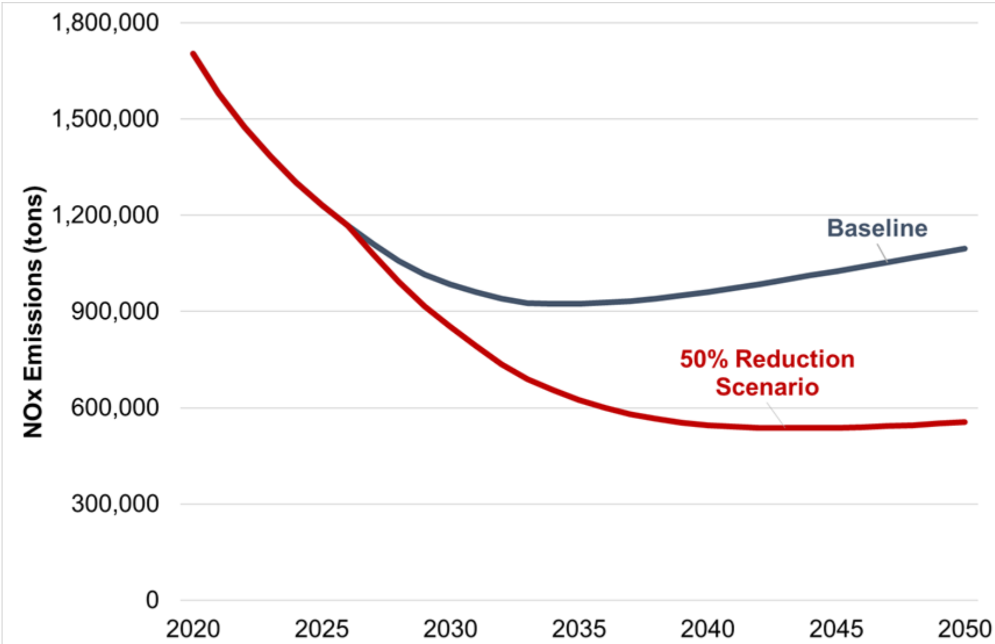
We also used MOVES2014 to estimate the aggregate reductions in NO_x emissions across the lower-48 states that would result from implementation of the tighter NO_x standard to every model year from 2027 through 2050, the final year for which MOVES2014 has NO_x emissions projections. That result could be of use if one were to conduct an analysis of benefits for specific future years rather than on a per-truck basis, the focus of our scoping analysis.

To compute the total annual tons of reduction over time, we extracted projected baseline NO_x emissions from MOVES2014 for each of the eight truck-types and all operational modes by state and by year from 2020 through 2050. To calculate the reductions in NO_x emissions, we reduced the baseline emissions across all the eight truck types by 50% in each year from 2027 onwards (where 2027 is the year in which the tighter NO_x standard is assumed to be implemented).[7]

The aggregated results are shown in Figure 2, while the results for each individual state are provided in Appendix A. The total baseline emissions across the U.S. for the eight HDOH truck types analyzed are projected to reach about 1.1 million tons by 2050, while emissions under the assumed scenario (*i.e.*, with implementation of a 90% tighter NO_x FTP standard that provides 50% reduction in in-use emissions) are projected to reach about 0.5 million tons by 2050.

[7] To keep the analysis simple, we did not apply any phase-in period for the standard. However, the effect of the standard (a 50% reduction in in-use emissions across the entire fleet), does take time to emerge as the standard is not applied to trucks purchased prior to 2027. Those pre-2027 trucks are assumed to remain in the fleet without any changes in their baseline operational or turnover assumptions.

Figure 2: Baseline and Scenario Emissions Across All HDOH Truck Categories



V. Development of Benefit-per-Ton Values and Benefit-per-Truck Estimates

A benefit-per-ton value measures the projected health benefits associated with projected changes in precursor emissions (*e.g.*, NO_x). The approach typically employed to compute those estimates involves running specific projected precursor emission changes through a full air quality fate-and-transport model (*e.g.*, CAMx) to project spatial changes in the relevant ambient pollutant concentrations. Those pollutant concentration changes are then provided as input to a demographic health risk analysis model (*e.g.*, BenMAP), along with specific assumptions about the concentration-response (C-R) relationship and social value per health effect incident to produce total monetized benefits. Those total benefits are then divided by the assumed change in tons of the precursor emission to yield a benefit-per-ton estimate stated in dollars.

This is called a "reduced-form" benefits estimate. The Agency and other groups often approximate total benefits of a potential emissions-reduction action by simply multiplying an available (and relevant) benefit-per-ton value by the number of tons of emissions reduction associated with that action. While subject to heightened uncertainty and inaccuracy, this approach avoids the great time and cost of conducting the air quality modeling step. We do not suggest that EPA will use this reduced-form approach in its own RIA for a future HDOH rulemaking, but we consider it a reasonable approach for the type of scoping-level approximation of benefits per truck that is the objective of our analysis.

While EPA has already published a number of such "reduced-form" benefit-per-ton estimates, we chose to derive our own estimates. By computing them ourselves, we can perform a wide range of sensitivity analyses that would not be possible using those published by others. For example, in our analysis, we (a) apply more up-to-date assumptions relating to baseline ambient pollutant concentrations;[8] (b) derive and explore the implications of more geographically disaggregated benefit-per-truck estimates; (c) use newer and different C-R assumptions that the Agency might use in its future benefits analyses; and (d) provide a range of benefit-per-truck estimates that vary in the extent to which they rely on extrapolation outside of the range of data supporting the original estimation of the C-R coefficients being applied.

We had to use different data sources to develop our estimates for ozone and $PM_{2.5}$. The rest of this section therefore describes the methods and the data that we used to compute our benefit-per-ton and associated benefit-per-truck estimates for ozone and $PM_{2.5}$ separately. It also provides state-specific detail to supplement the more aggregated estimates presented in the accompanying Summary Report. All of the results reported in this section give full weight to risk estimates from exposures as low as zero and make no adjustment for declining confidence associated with extrapolation of the C-R relationship to concentrations at the low end of the range of observations in the original epidemiological study. Our method for assessing the quantitative sensitivity to alternative limits on the degree of such extrapolation is described in Section VI of this report.[9]

A. $PM_{2.5}$ Calculations

To develop our "reduced-form" benefit-per-ton estimates for $PM_{2.5}$, we relied upon air quality modeling used to produce a set of mobile-source benefit-per-ton estimates reported in Wolfe *et al.* (2018). That study was of particular relevance to our analysis because it provided $PM_{2.5}$ benefit-per-ton estimates specifically due to NO_x emissions from HDOH trucks.[10] The paper reported average national and regional ("East" and "West") benefit-per-ton estimates, using a baseline $PM_{2.5}$ concentration grid and associated baseline NO_x emissions projected to occur in 2025. The benefit-per-ton estimates reported in the paper are calculated using two C-R functions – from Krewski *et al.* (2009) and Lepeule *et al.* (2012) – and using BenMAP's demographic assumptions for the year 2025.

EPA provided NERA with the BenMAP grids of 2025 HDOH nitrate contributions and the associated NO_x emissions (by state) employed by Wolfe *et al*. Using those data and the same C-R relationships, NERA ran the BenMAP model to confirm we could replicate the nitrate benefit-per-ton estimates due to HDOH trucks, both at the national and the regional level.

To better understand the degree of potential variation in such values on a geographic basis, NERA then used BenMAP and those same air quality and emissions data to develop benefit-per-ton estimates on a more disaggregated basis, generally state by state (which was the smallest disaggregation available for the emissions data.) However, recognizing that much of the ambient $PM_{2.5}$ in very small states would be attributable to emissions in surrounding states, several of the smallest Eastern states were aggregated into subregions about the size of the larger states.[11]

[8] For our analysis, we used 2035 baseline ozone and $PM_{2.5}$ grids from a recent air RIA (EPA, 2019a), which were the BenMAP inputs with the most up-to-date air quality modeling that we were able to identify in the public domain. The concentrations in these grids also are broadly reflective of the concentrations of ozone and $PM_{2.5}$ projected to occur in the years during which the tighter standard would be having most of its incremental impact (*i.e.*, in the 2030s and 2040s).

[9] The case for this latter type of sensitivity analysis, which we call "confidence weighting," is explained in more detail in the accompanying Summary Report.

[10] The species of $PM_{2.5}$ associated with NO_x precursor emissions is particulate nitrate.

[11] The two multi-state regions are called North East and Mid-Atlantic. The North East region comprises Connecticut, Massachusetts, New Hampshire, New York, Rhode Island and Vermont. The Mid-Atlantic aggregate region comprises Delaware, Maryland, New Jersey, Pennsylvania, Virginia and West Virginia. The benefit-per ton-estimates for these aggregate

Using the Krewski C-R coefficient that Wolfe *et al.* used, we found much greater geographic variation in the benefit-per ton-estimates than was apparent from the values for the two aggregate regions in that study. This variation is illustrated for our year-2050 estimates as a map in Figure 3, and as a population-weighted cumulative distribution in Figure 4.[12] State-specific estimates range from less than $1,000 per ton to more than $20,000 per ton (2019$) around a national average of $8,000 per ton.[13] This range primarily reflects variations in population densities, and also regional differences in the amount of change in ambient $PM_{2.5}$ per ton of HDOH NO_x emissions. The values in these figures are based on year-2050 demographic assumptions, but the variation from state to state is very similar for other demographic years. The numerical values estimated for the 2030, 2040, and 2050 demographic assumptions are provided in Appendix B.

regions are calculated by the dividing the aggregate benefits for the region by the aggregate NO_x emissions reduction for the region.

[12] We employed a C-R coefficient for all-cause mortality of 0.0058, based on a relative risk of 1.06 per 10 µg/m^3 change in $PM_{2.5}$ reported in that report's Commentary Table 4 on p. 126.

[13] These estimates apply year-2050 demographic conditions, whereas Wolfe *et al.* applies year-2025 demographic assumptions, which produce lower per-ton values. Also, these are stated in 2019 real dollars, whereas Wolfe *et al.* states its estimates in 2015 real dollars, which also results in lower numerical values. As noted earlier, our analysis methods do replicate the estimates reported Wolfe *et al.* when we apply the same demographic assumptions and state the results in same-year real dollars.

Figure 3: Map of $PM_{2.5}$-Only Benefits per Ton by State Using the Krewski *et al.* (2009) C-R Coefficient (2050)

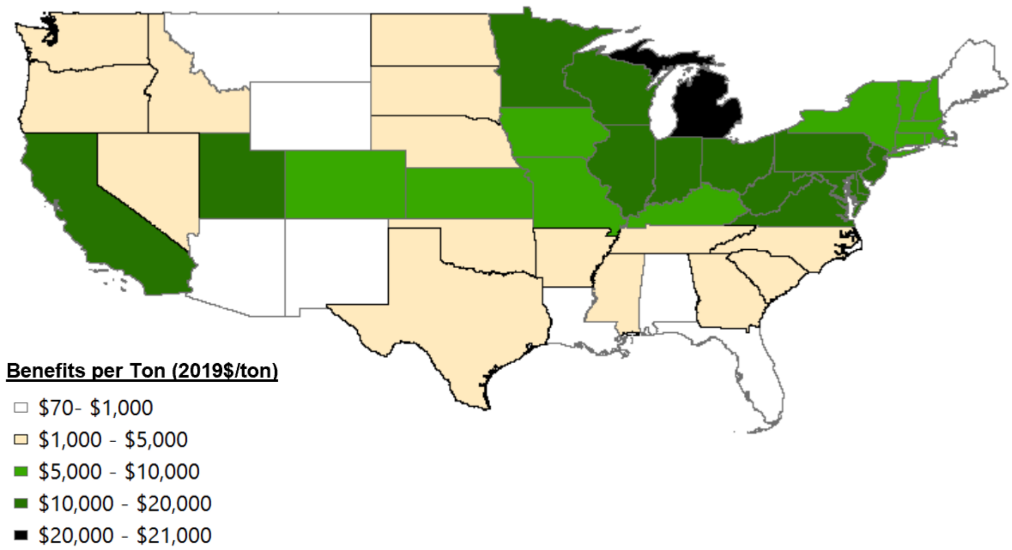
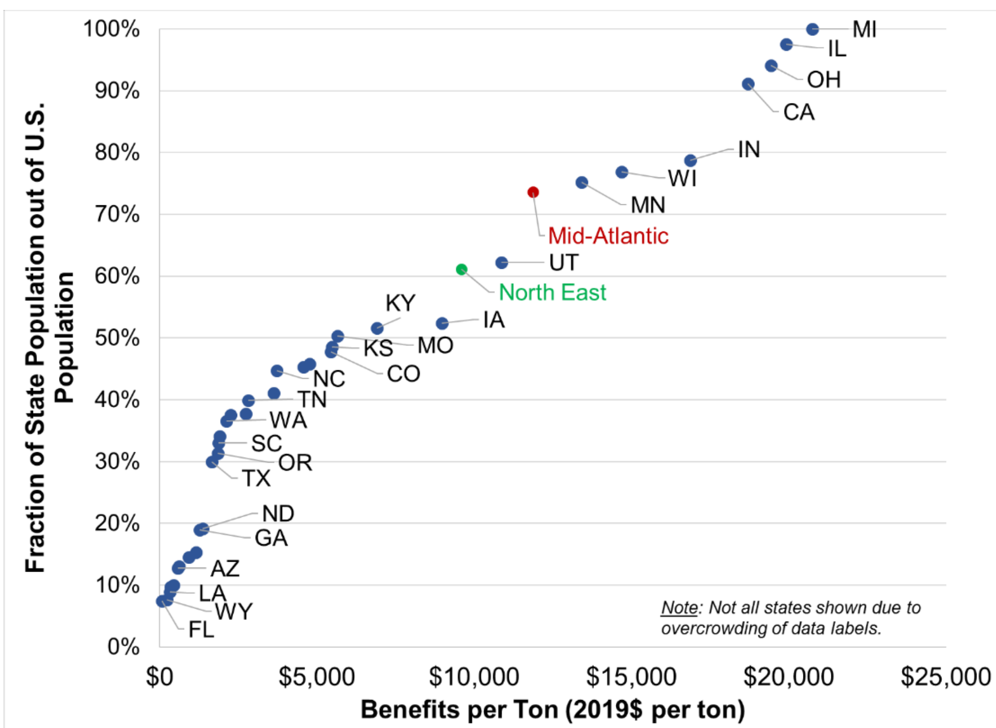


Figure 4: Cumulative Distribution of $PM_{2.5}$-Only Benefits per Ton by State Using the Krewski *et al.* (2009) C-R Coefficient (2050)



Like Wolfe *et al.*, we estimate a range for the $PM_{2.5}$ benefits-per-ton using two alternative C-R relationships for mortality risk. Rather than use the same two C-R relationships that Wolfe *et al.* used, we chose to update those inputs to reflect what one might expect the Agency to use in a future RIA. To decide on the assumptions that would drive the lower and higher ends of the range, NERA reviewed EPA's recent Policy Assessment for $PM_{2.5}$ (EPA, 2020). That document contains all-cause mortality risk estimates that range from one that is much lower than that obtained using the C-R relationship from Krewski *et al.* (2009) to one that is much higher, based on a new study by Di *et al.* (2017). Given the widespread use of Krewski *et al.* in Agency risk analyses up until the current Policy Assessment, and given the fact that it is not as low as the lowest estimate in the Policy Assessment, we chose to be conservative and rely on the C-R relationships from Krewski *et al.* (2009) at the lower end. We chose to rely on the Di *et al.* (2017) study at the higher end.[14]

Consistent with EPA practice for long-term $PM_{2.5}$ benefits calculations, we applied EPA's standard twenty-year segmented cessation lag (EPA, 2004) to both the lower and higher end estimates.[15] As noted above, our year-2050 national average benefit per ton of HDOH NO_x emissions is about $8,000 (2019$); the same estimate calculated using the Di *et al.* (2017) C-R relationship is about $10,000 per ton (2019$). The geographic variation around that average is presented in Figure 5 and Figure 6 on the next page, and is very similar to that using Krewski *et al.* Numerical values behind these figures, and for 2030 and 2040 are also provided in Appendix B.

[14] We employed a C-R coefficient for all-cause mortality of 0.0087, based on a relative risk of 1.084 per 10 µg/m^3 change in $PM_{2.5}$ (Single pollutant analysis) from Di *et al.* (2017), Table 2 (p. 2518). That C-R relationship applies to people ages 65 years or older, and our BenMAP calculations have used this older population when applying the Di *et al.* coefficient.

[15] This structure assumes a 30% reduction in premature mortality in the first year, a 50% reduction over years 2 through 5 and a 20% reduction over years 6 through 20 after the reduction in $PM_{2.5}$ concentration.

Figure 5: Map of $PM_{2.5}$-Only Benefits per Ton by State Using the Di *et al.* (2017) C-R Coefficient (2050)

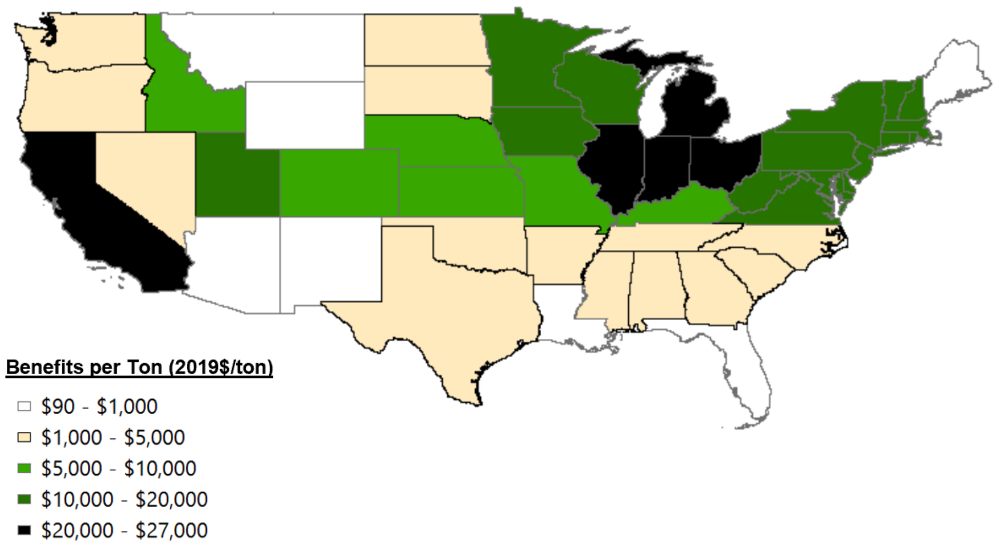
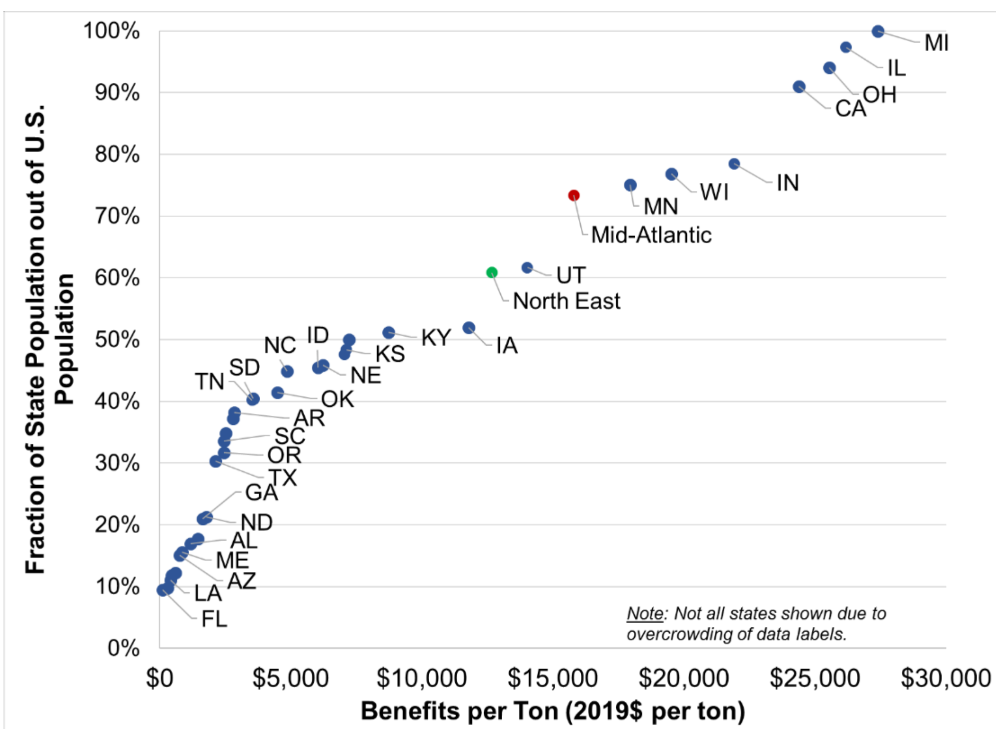


Figure 6: Cumulative Distribution of $PM_{2.5}$-Only Benefits per Ton by State Using the Di *et al.* (2017) C-R Coefficient (2050)



As explained in the prior section, our estimates of the *per-truck* benefits apply our estimates of benefits per ton in each year from 2027 through 2057[16] to our estimates of the per-truck tons of reduction each respective year, and take a present value of that stream of annual values. Figure 7 and Figure 8 below present the maps and cumulative distributions, respectively, of $PM_{2.5}$ benefit-per-truck estimates computed using the Krewski *et al.* (2009) epidemiological study and applying a 3% discount rate. Figure 9 and Figure 10 present the same information using instead the Di *et al.* (2017) epidemiological study (also applying a 3% discount rate). The national average $PM_{2.5}$ estimates (for a 3% discount rate) are $4,580 per truck based on the Krewski *et al.* study and $5,540 per truck based on the Di *et al.* study. As with the distributions presented in Figure 4 and Figure 6, the states with the highest benefit-per-truck estimates are in the Midwest and California.

The corresponding maps and distributions for the $PM_{2.5}$ benefit-per-truck estimates computed using a 7% discount rate are presented in Appendix C. For each state, those benefits estimates are about 25% lower than their respective 3% discount rate estimates, leaving the geographical variations much the same as presented in the figures below.

[16] For each year's specific benefit-per-ton value, we interpolated linearly between our 2030 and 2050 per-ton values. We considered this a reasonable approximation for our scoping analysis. However, we note that use of a more refined interpolation that incorporates year-2040 values appears to increase per-truck benefits estimates by less than 5%.

Figure 7: Map of $PM_{2.5}$-Only Benefits-per-Truck by State Using the Krewski *et al.* (2009) C-R Coefficient, 3% Discount Rate

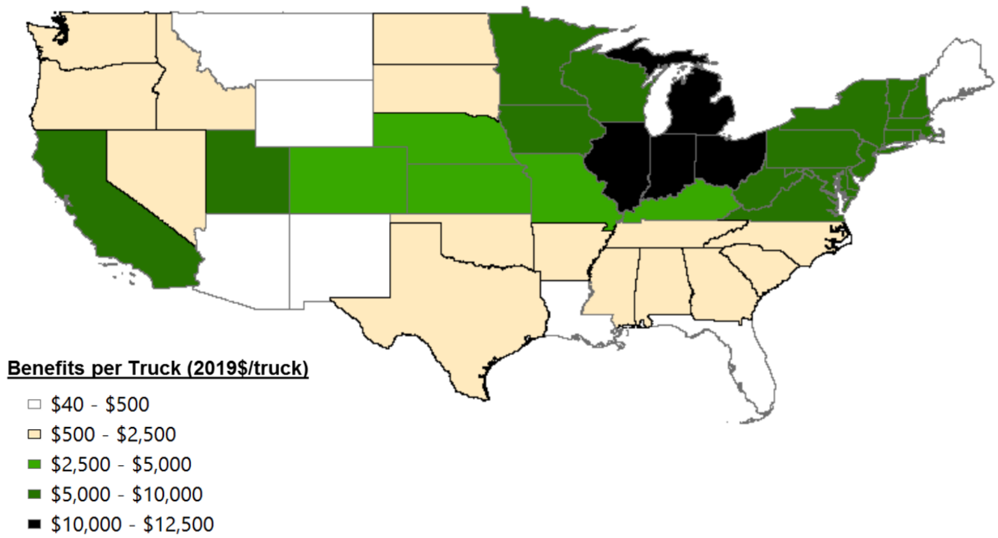


Figure 8: Cumulative Distribution of $PM_{2.5}$-Only Benefits-per-Truck by State Using the Krewski *et al.* (2009) C-R coefficient, 3% Discount Rate

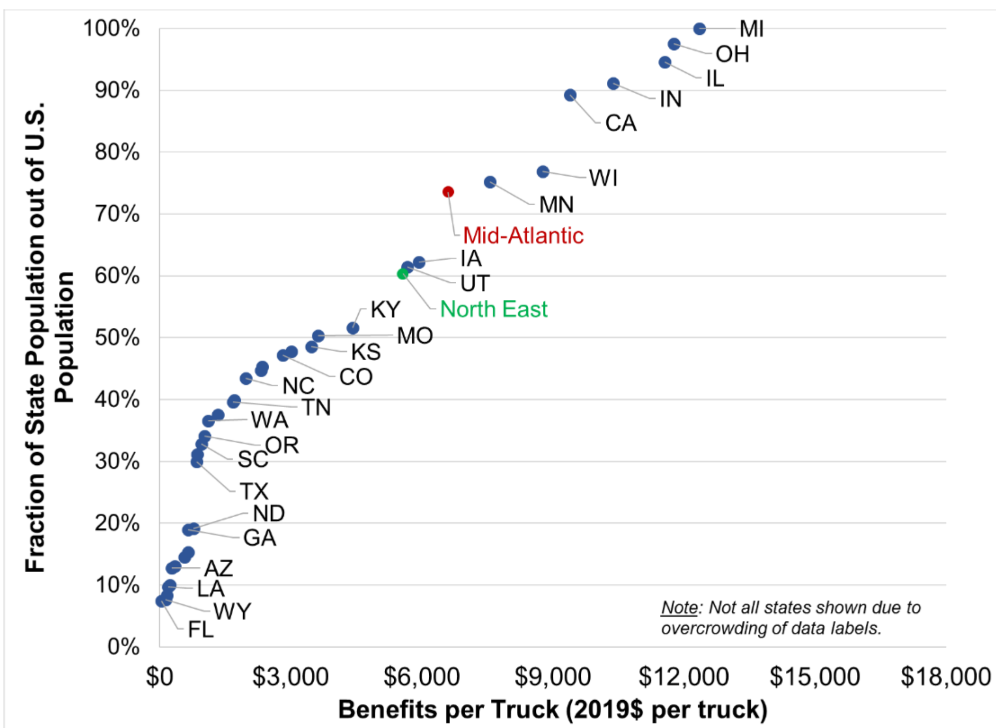


Figure 9: Map of $PM_{2.5}$-Only Benefits-per-Truck by State Using the Di *et al.* (2017) C-R Coefficient, 3% Discount Rate

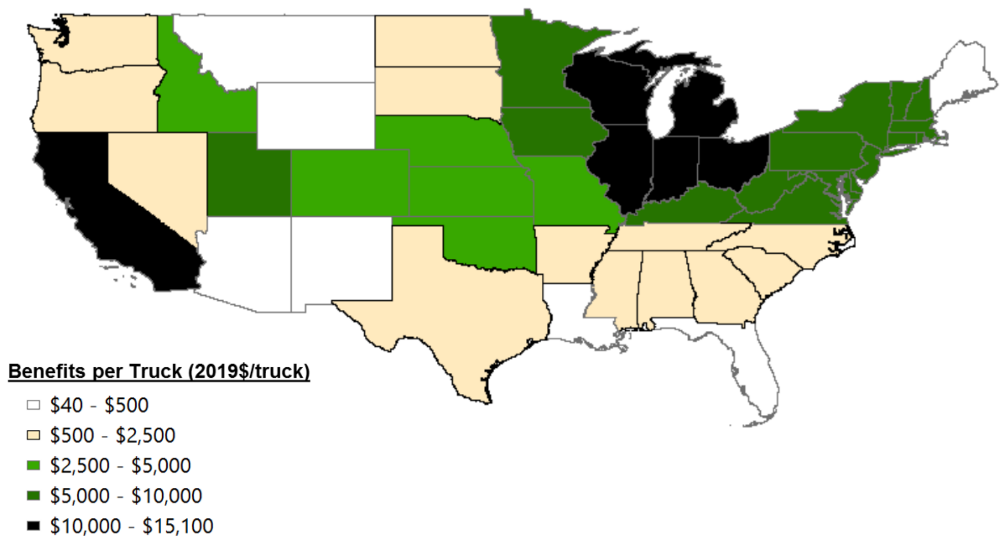
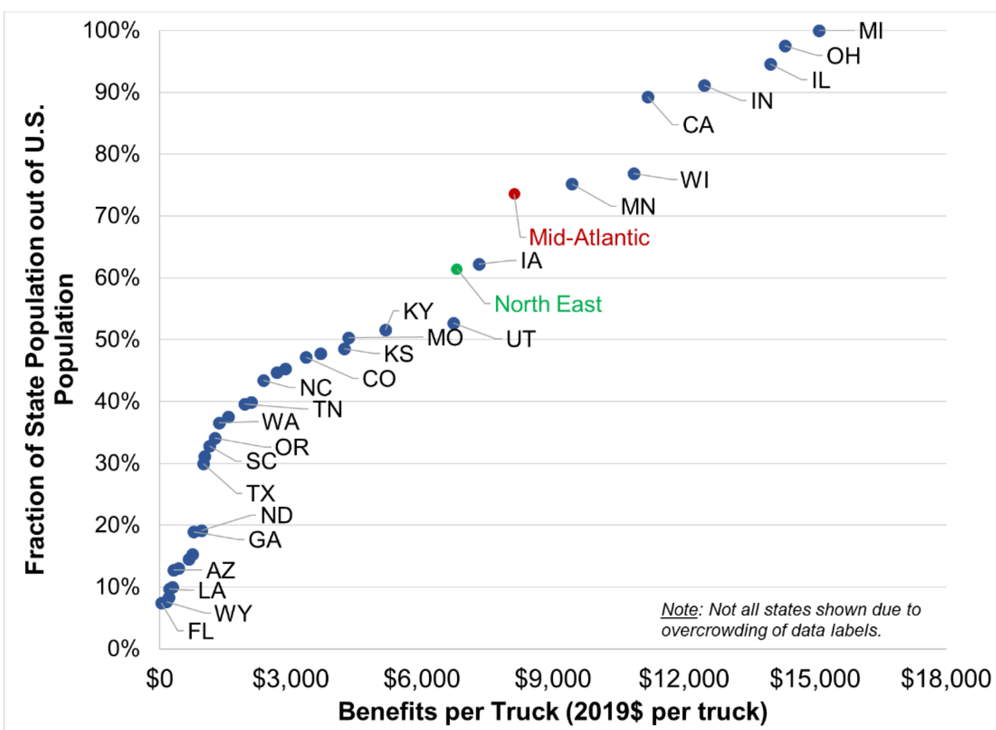


Figure 10: Cumulative Distribution of $PM_{2.5}$-Only Benefits-per-Truck by State Using the Di *et al.* (2017) C-R Coefficient, 3% Discount Rate



B. Ozone Calculations

Wolfe *et al.* (2018) does not provide any benefit-per-ton estimates for ozone. Also, there appears to be only one example among EPA's past RIAs that used the "reduced-form" benefit-per-ton methodology for ozone – the RIA for the Clean Power Plan (EPA, 2015a). Because those estimates were based on NO_x reductions from electricity generating units, which have a very different geographic distribution than vehicle emissions, they are not relevant for use in our HDOH benefits scoping analysis. All of the other past RIAs we reviewed that contained estimates of ozone-related health benefits had based those estimates on full-scale US-wide air quality modeling of the specific emissions reductions projected for that regulation. One can develop a rough estimate of the average benefit-per-ton *implied* in those remaining RIAs by dividing the RIA's estimate of total benefits by its estimated tons of NO_x emissions reductions.

Of those remaining RIAs, the one that is most relevant to an HDOH NO_x reduction regulation is the RIA for the Tier 3 Light-Duty Vehicle standards from 2014 (EPA, 2014a). We find that the approximate national-average ozone benefit per ton implied in that RIA (stated in 2019$) ranges from about $3,800 per ton when using an all-cause mortality C-R relationship from Bell *et al.* (2004) to about $17,300 per ton when using an all-cause C-R relationship from Levy *et al.* (2005). A more relevant but older RIA is that for the prior HDOH NO_x emissions rulemaking (EPA, 2000). Its implied national average ozone benefit per ton was $824 (2019$). That estimate was based on a C-R function for hospital admissions rather than mortality. Clearly there is a wide range, but none of those estimates reflects the Agency's current thinking about ozone-related health risks that could be viewed as a likely basis for ozone benefits calculation in a future RIA. Below we describe how we developed our own reduced-form estimates for ozone benefits, and their implications for per-truck benefits.

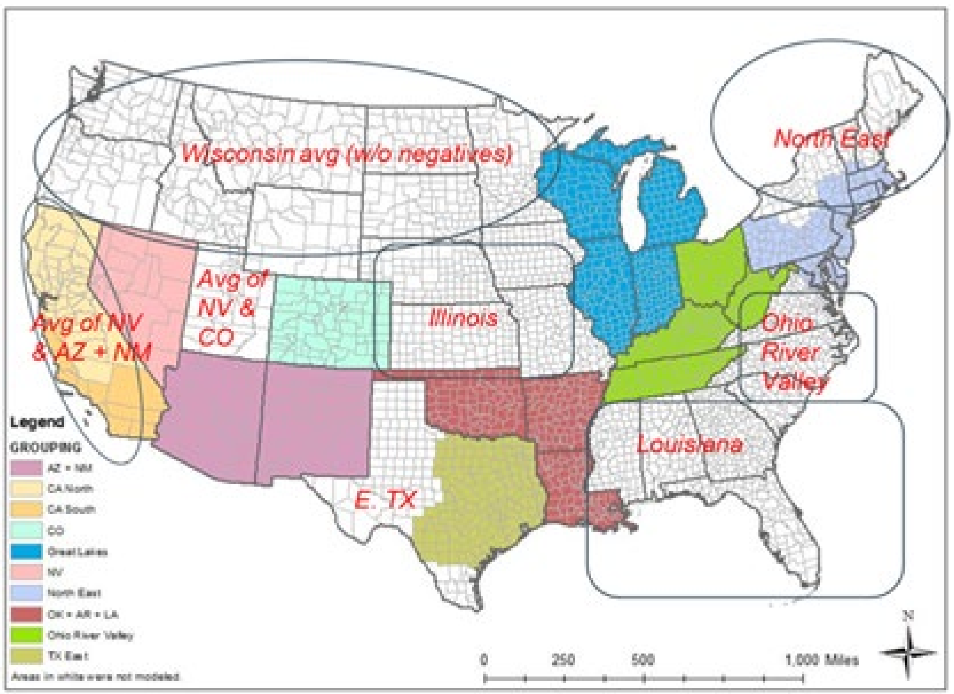
EPA's current draft Policy Assessment for ozone (EPA, 2019c) does not provide epidemiology-based risk calculations for any health effect, and it specifically casts doubt on ozone's potential mortality risk. This suggests that a future RIA might not attribute any mortality benefits to ozone reductions. In the spirit of providing an upper and lower value, however, we decided to employ a coefficient for respiratory mortality from Zanobetti and Schwartz (2008) as our higher (*i.e.*, non-zero) estimate. This choice reflects the fact that EPA did cite several epidemiological studies addressing respiratory health risks in an appendix of the draft ozone Policy Assessment; of those cited, Zanobetti and Schwartz provided the clearest option for a C-R coefficient specifically for respiratory mortality risk.[17]

Also challenging to this part of our analysis was a lack of a specific grid of ambient ozone concentrations associated with a specific quantity of tons of NO_x emissions, such as was available for $PM_{2.5}$ from the Wolfe *et al.* study. We instead had to rely on less nationally comprehensive results from prior air quality modeling sensitivity cases that had been prepared for the 2015 Ozone RIA (EPA, 2015b). For that RIA, EPA conducted several sensitivity runs with CAMx for specific regions of the U.S. that the Agency had projected would need to make NO_x reductions to attain an ozone NAAQS down to 65 ppb. Some of those sensitivity runs simulated the ambient ozone impacts of "across-the-board" 50% reductions in anthropogenic NO_x emissions, which thus, at least in part, included mobile source emissions reductions. We consider those specific sensitivity runs to be the most relevant for our analysis. They had been run for eight U.S. regions, identified by the colored areas (excluding the two in California) in Figure 11, which is

[17] We employ a C-R coefficient for respiratory mortality of 0.00054, based on a relative risk of 1.0054 per 10 ppb change in 8-hr ozone from the 0-day lag model reported in Zanobetti and Schwartz (2008), Table 1, p. 186.

copied from EPA (2015b).[18] The outputs of those sensitivity runs that were reported in a technical support document spreadsheet (EPA, 2015c) were ozone design values at each existing monitor across the U.S. for the base case and for each of the sensitivity cases and the NO_x emissions changes between the two cases. Following guidance in that document, we used those outputs to calculate "ozone response factors" for each of the sensitivity cases by dividing the projected change in the ozone design value at each monitor across the U.S. by the tons of NO_x emissions reduction assumed for that case.

Figure 11: Basis for Estimating Ozone Response Factors for Each State
(Source: EPA (2015b), Figure 2-2, with red font text added by NERA, as explained in text.)



Note: For northern states west of WI, "Wisconsin avg (w/o negatives)" means that monitors in WI with a negative response factor were not included in the average estimated for these states. Negative values imply local ozone formation is VOC-limited, which does occur in parts of WI (near the lake), but which we assume does not occur in northern states west of WI.

For each state where emissions were reduced in one of the eight relevant sensitivity runs, we extracted the ozone response factors for all the monitors in that state and adopted the simple average of those values as our analysis's assumption for that state's change in ambient ozone due to a ton of NO_x emitted by HDOH trucks in that state.

Although EPA's data provided response factors for all monitors throughout the entire U.S., we did not use response factor data for monitors that were not within the region for which emissions had been cut.[19] For areas of the U.S. that were not included in any of EPA's sensitivity cases (*i.e.*, the white areas in Figure 11), we adopted an average ozone response factor from one of the modeled regions, selecting a region that we judged to have relatively similar ozone forming attributes (*e.g.*, temperature, sunlight, *etc.*). For

[18] None of the sensitivity cases run for the two California regions involved the 50% across-the-board NO_x reductions that we considered relevant for our analysis.

[19] We did confirm that response factors for monitors outside of the region of the simulated emissions reductions were generally very much smaller than those for monitors within the region.

example, for Missouri, we used an ozone response factor (*i.e.*, the average ppb change in Missouri per ton of NO_x emitted in Missouri) that was the same as EPA's modeling indicated for Illinois. The red text on Figure 11 identifies the assignments we made for each of those areas that were not included in one of EPA's sensitivity cases.[20] The state-specific values of our resulting set of ozone response factors are provided in Appendix D.

We multiplied our state-specific ozone response factors by the state-specific NO_x emission reductions that we also estimated (as described in Section IV, and reported in Appendix A) to obtain rough estimates of projected changes in ozone design values expected to occur in each state with the implementation of the hypothetical tighter HDOH NO_x standard. We further assumed that changes in average seasonal ozone concentrations would be equal to the estimated changes in design values that was the basis of our estimates of ozone response factors.[21] Using BenMAP, we applied those estimates of absolute changes in ambient ozone to the baseline ozone levels in every 12-km grid cell in each respective state to compute ozone benefit-per-ton estimates. As noted above, we used a C-R relationship for acute respiratory mortality risk during the summer months (June – August) estimated by a multi-city study and reported in Zanobetti and Schwartz (2008).[22] Those calculations were carried out for the U.S. and by state for 2030, 2040, and 2050. The benefit-per-ton estimates obtained for the U.S. and by state are provided in Appendix B, with the year-2050 estimates summarized below.

Our estimate of the national average ozone benefit per ton for 2050 is $795 per ton (2019$).[23] Figure 12 and Figure 13 present the state-specific results, which show California far higher than any other state: about $5,250 per ton – more than 6 times the U.S. average. If California is removed from the data, the average for the remaining 47 states is about $400 per ton.

Figure 14 and Figure 15 graph the *per-truck* benefit estimates when applying a 3% discount rate. The national average ozone benefit-per-truck estimate is $390 per truck (2019$). California's estimate is $2,570 per truck, while the average for Rest of U.S. is $210 per truck. The corresponding maps and distributions for the ozone benefit-per-truck estimates computed using a 7% discount rate are presented in Appendix C. For each state, those estimates are about 25% lower than the respective 3% discount rate estimates.

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[20] Because the sensitivity cases for California were not appropriate for our analysis needs, we made an assignment for California too, as identified in red font in the figure.

[21] We surmise that this assumption causes our analysis to overstate the projected changes in ozone in most locations, as it is quite likely that absolute changes in average ozone will be smaller than absolute changes in the highest levels of ozone. If so, this also means that our benefit-per-truck estimates for ozone will be overstated. As those estimates have turned out quite small even if they may be overstated due to this assumption, we have not attempted to further refine the assumption or to conduct sensitivity analyses for it.

[22] Consistent with EPA's methods for estimating risk from ozone exposures measured only during ozone-season months, our benefits calculations are for June through August. An adjustment factor of 0.25 was applied to BenMAP's year-round counts of avoided respiratory mortality. This factor reflects the fraction of the days in the year covered by those months.

[23] This is low compared to the ozone benefit-per-ton values implied in the Tier 3 Light-Duty Vehicle Standards RIA (EPA, 2014a). The primary reason for the large reduction is that our benefits calculations are for respiratory mortality only, whereas the 2014 RIA used C-R relationships for all-cause mortality, which the Agency now views as not likely causal. We also suspect (but cannot confirm) that the 2014 RIA applied a seasonal C-R relationship to mortality risk across the entire year. The Agency did not make such an extrapolation in its Health Exposure and Risk Assessment for that ozone NAAQS review (EPA 2014b).

Figure 12: Map of Ozone-Only Benefits per Ton by State (2050)

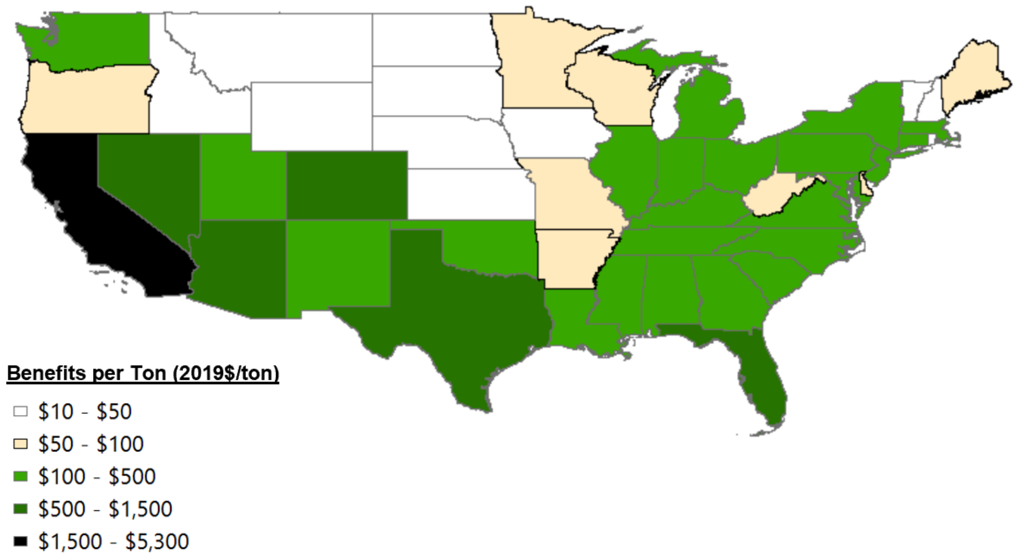


Figure 13: Cumulative Distribution of Ozone-Only Benefits per Ton by State (2050)

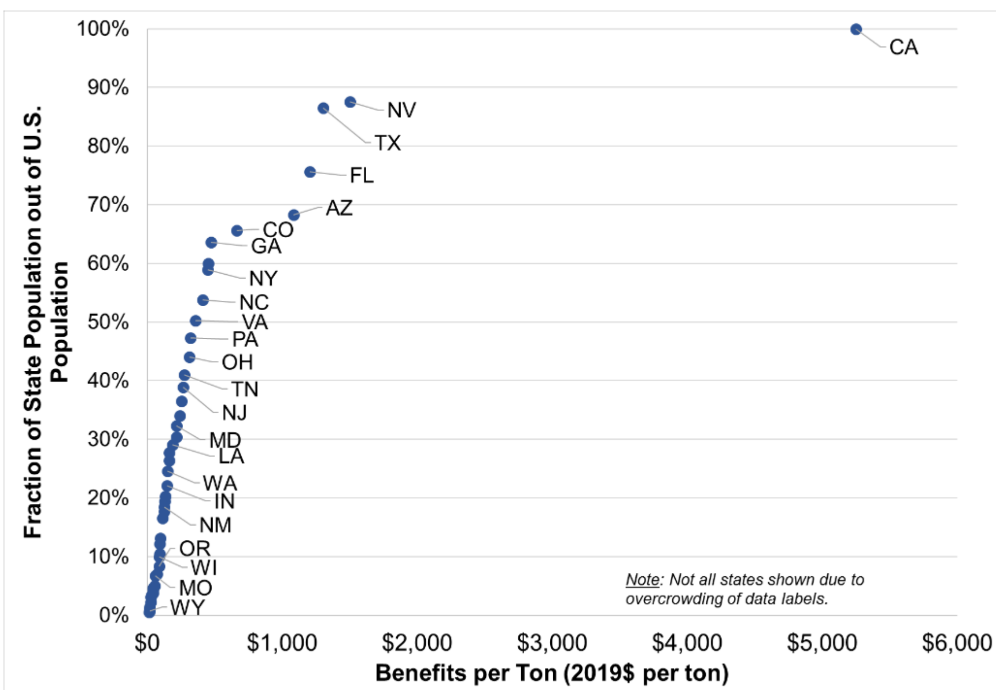


Figure 14: Map of Ozone-Only Benefits-per-Truck by State, 3% Discount Rate

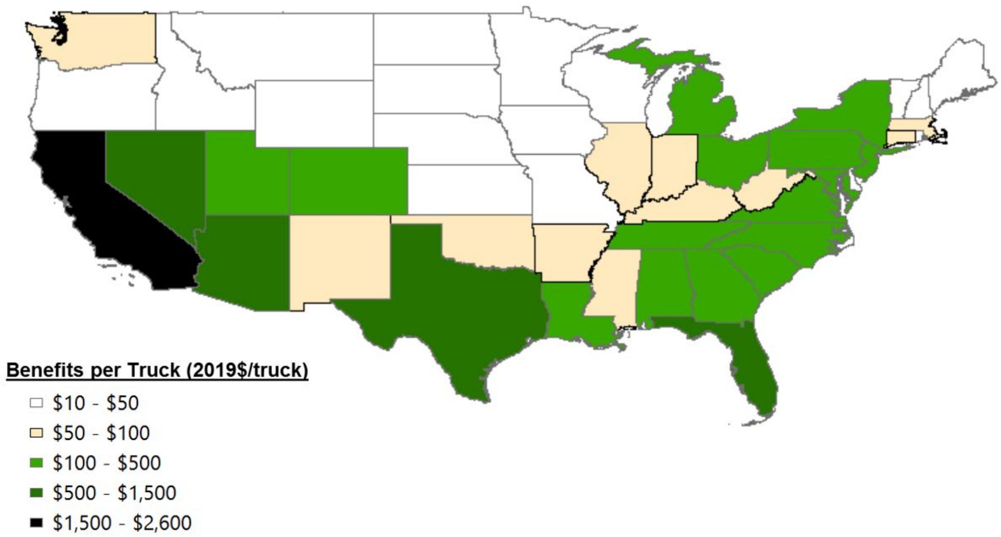
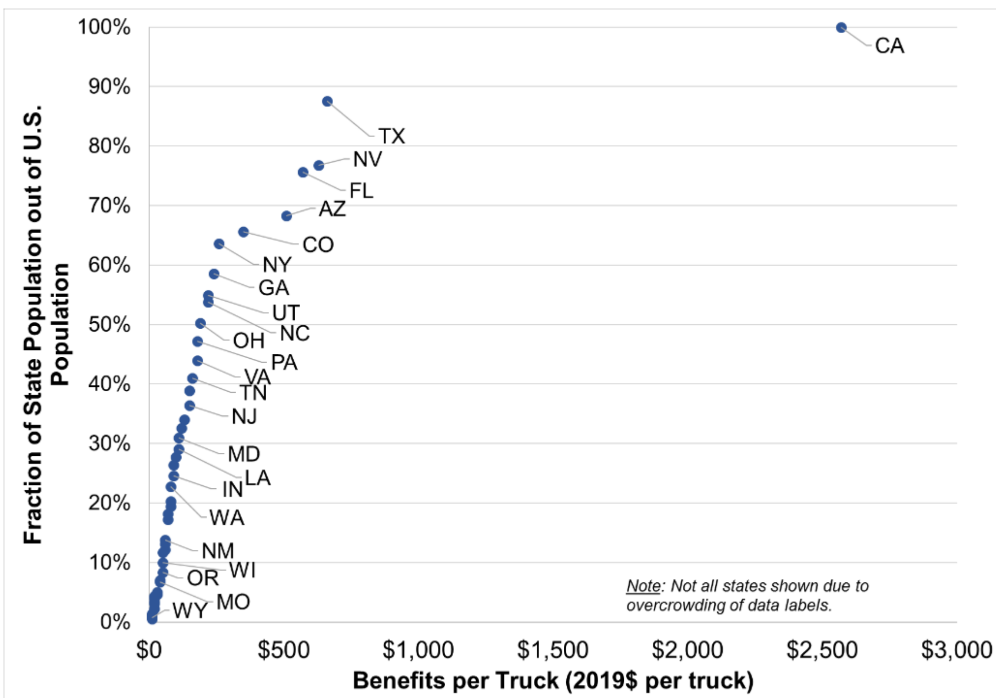


Figure 15: Cumulative Distribution of Ozone-Only Benefits-per-Truck by State, 3% Discount Rate



VI. Benefit-per-Truck Estimates with Varying Confidence Levels

An important input that drives the benefit-per-ton estimates and thus the benefit-per-truck estimates is the C-R coefficient, which is an assumption about the increase in health risk per unit change in ozone and $PM_{2.5}$ concentration. That assumption is usually based on a statistically-derived association reported in one of many existing epidemiological papers. There are significant scientific uncertainties introduced when using these statistical associations to predict risks under different population and air quality conditions than those analyzed in the papers, since it involves extrapolation outside the range of observed exposures. The accompanying Summary Report of our analysis provides a detailed explanation of this concern with extrapolation in benefits analyses.[24] It also discusses an approach introduced by EPA in a recent RIA (EPA, 2019a) to quantify the sensitivity of benefits estimates to various amounts of limitations on the amount of extrapolation allowed in their computation, which we have applied to the benefit-per-truck estimates of our scoping analysis.

We provide alternative estimates of benefits per truck associated with varying levels of extrapolation-related confidence. Estimates at the "more confident" end of the spectrum exclude benefits calculated to occur in areas with projected baseline concentrations below the 25th percentile of the range of observations in the original C-R estimation data.[25] Estimates at the "less confident" end of the spectrum make no exclusions at all, allowing extrapolation of the C-R relationship even where projected baseline concentrations are lower than the lowest measured level (LML) in the original epidemiological study.[26] Estimates that fall between these two ends of the spectrum exclude benefits that are in areas with projected baseline concentrations that are below percentile levels lower than the 25th percentile of the pollutant observations in the original study (such as the 1st, 5th, 10th percentiles of the original study's observed exposure levels).

To apply this method, two sets of data are needed. First, the relevant baseline concentrations associated with the regulation's benefits, C_b, must be identified. Second, the concentrations associated with each selected population-weighted percentile p in the original epidemiological study must be obtained. These values are denoted C_p, which we apply for p=0, 1st, 5th, 10th, and 25th percentiles. The estimated benefits are placed into bins according to the baseline concentration level, C_b, from which they have been computed. Total benefits associated with each percentile level p are then recomputed by summing up benefits in only those bins with baseline concentrations $C_b \geq C_p$. This results in gradually declining benefits-per-ton estimates as the percentile cut-off rises – implying greater confidence that the benefits included in the computation are not the result of speculative extrapolation outside of the range of observed exposures.

An appropriate set of baseline exposures would be those projected to be in effect during the time period when the new regulation is taking effect. For our analysis, that would be from 2027 through 2057. The most relevant air quality projections usable in BenMAP that we could identify in the public domain are those prepared for the RIA for the final Affordable Clean Energy (ACE) regulation, which include projected $PM_{2.5}$ and ozone levels nationally for the years 2025, 2030, and 2035. We obtained those BenMAP air quality grids from EPA. We chose to use the 2035 projections for our analysis, as most of the per-truck benefits occur in the years 2027 through 2040, although about 20% do occur after 2040, when baseline exposures will probably be lower still.

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[24] See Section IV of that Summary Report.

[25] Consistent with EPA's confidence spectrum, we consider levels up to the 25th percentile of the original data set.

[26] The Agency uses the acronym LML to denote the 0th percentile of the distribution of exposures in the original study.

For each of the three C-R relationships that we use in our scoping analysis, we obtained the concentrations associated with each percentile (*i.e.*, the C_p values) from the respective original study. For example, we use the population-weighted exposure distribution from Krewski *et al.* (2009) to develop the values of C_p for our lower $PM_{2.5}$ benefit-per-truck estimates, and we use the distribution of $PM_{2.5}$ exposures in the Di *et al.* (2017) study to develop confidence-weighting adjustments for our higher $PM_{2.5}$ benefit-per-truck estimates. The concentration levels at each percentile from the Krewski *et al.* study are reported in the ACE RIA, but we confirmed them from Table 1 of the original paper. The percentiles in the Di *et al.* study are available in supplemental materials to the original paper but are more precisely listed in a $PM_{2.5}$ docket entry (EPA, 2019b). We use information on the distribution of city-specific average ozone concentrations reported in Table 2 of the online supplement to Zanobetti and Schwartz (2008) study.

For each of the three epidemiological studies we have relied upon, Tables 1 through 3 below identify (in the first row) the ambient concentration levels (C_p values) for each of the above percentile cut-off levels that we have used to explore sensitivities to extrapolation-related confidence weighting. The second row of each table identifies the percentage of the respective study's total avoided premature statistical deaths that lie *within* each alternative confidence range. (These sum to 100% across the row.) The last two rows of each table report the benefit-per-truck values associated with each confidence level when applying, respectively, a 3% and 7% discount rate to the present value calculation. The first column in each table reports the national average estimates unadjusted for confidence (which we reported in the previous section), while the values in the columns to the right show the estimates that have increasingly higher confidence, up to the point where only benefits in areas with exposures at or above the 25th percentile of the original epidemiological study are included.

Table 1 presents the $PM_{2.5}$ benefit-per-truck estimates calculated using Krewski *et al.* (2009). It shows that only about 3% of the benefits are projected to occur in locations that have exposures greater than the 25th percentile of all the exposures in the epidemiological study. Thus, the unadjusted estimate of $4,580 per truck that was reported in the prior section of this report declines to $160 per truck at the "more confident" end of the spectrum.[27] If we were to use the 10th percentile as a less conservative confidence cut-off, the associated benefit-per-truck estimate would be $360 with about 8% of the benefits projected to occur in locations that have exposures greater than the 10th percentile of all the study exposures.[28] As before, estimates computed using a 7% discount rate are about 25% lower than the respective 3% discount rate.

Table 2 presents the corresponding $PM_{2.5}$ benefit-per-truck estimates calculated using Di *et al.* (2017). The uncertainty due to extrapolation is much less pronounced than in Table 1 because the distribution of exposures that were observed in the Di *et al.* study is lower than that observed in the Krewski *et al.* study. For example, about 14% of our unadjusted benefits are projected to occur in locations with exposures greater than the 25th percentile of Di *et al.*'s study, compared to only 3% in the case of Krewski *et al.* Thus, we can see that the benefit-per-truck estimates decline less when moving from the "less confident" to the "more confident" end of the benefits scale, with the unadjusted estimate (for a 3% discount rate) of $5,540 per truck declining to $780 per truck. At the 10th percentile confidence cut-off, the Di *et al.* study

[27] The benefit-per-truck estimate of $160 is calculated by multiplying the confidence un-adjusted estimate with the fraction of benefits that can be attributed to locations with exposures greater than the 25th percentile of the study exposures: 3%*$4,580.

[28] 8% is computed as the sum of the percentages of the total deaths that can be attributed to locations with exposures greater than the 25th percentile of the study exposures (i.e. the sum of the last two columns): 4.3%+3.4% with the corresponding estimate of $360 computed as: 8%*$4,580.

(for the 3% discount rate) is $3,180 per truck, more than eight times greater than the corresponding Krewski *et al.* estimate.

Table 3 presents the ozone benefit-per-truck estimates calculated using Zanobetti and Schwartz (2008). The pattern observed in the drop-off of the benefit-per-truck estimates is significantly different from that for $PM_{2.5}$. The unadjusted estimate of $390 per truck remains unchanged through the 5th percentile confidence cut-off because almost none of the U.S. is projected to have ozone concentrations below 23.4 ppb in our baseline air quality grid, even though Zanobetti and Schwartz data indicate that about 5% of the cities in their study had lower average ozone levels.[29] The confidence-weighted ozone benefit estimate declines to $180 per truck at the highest confidence end of the spectrum with 46% of our estimated ozone benefits projected to occur in locations with exposures above the 25th percentile of all the cities observed in the original Zanobetti and Schwartz study.

[29] We have no explanation for such a discrepancy at this time, which seems surprising given that our estimates of baseline exposure are more disaggregated than those of Zanobetti and Schwartz's observations (12-km grid resolution *vs.* city-wide averages) and they occur later in time (2035 *vs.* 1989-2000) when tighter ozone standards will be in place.

Exhibit B

Table 1: Avoided Premature Statistical Deaths (%) and National $PM_{2.5}$ Benefit-per-Truck Estimates (2019$/truck) by Confidence Level Using Krewski *et al.* (2009) Epidemiology Study and Applying 3% and 7% Discount Rates

	Below LML (<5.8)	LML to 1st Percentile (≥5.8 & <6.7)	1st to 5th Percentile (≥6.7 & <8.8)	5th to 10th Percentile (≥8.8 & <10.2)	10th to 25th Percentile (≥10.2 & <11.8)	25th Percentile & Above (≥11.8)
Avoided Premature Statistical Deaths (%)						
National	9%	16%	56%	11%	4%	3%
Benefit-Per-Truck (2019$/truck)						
3% Discount Rate	$4,580	$4,150	$3,440	$870	$360	$160
7% Discount Rate	$3,430	$3,110	$2,570	$650	$270	$120

LML = Lowest Measured Level, meaning the minimum observed $PM_{2.5}$ concentration in the original epidemiological study

Exhibit B

Table 2: Avoided Premature Statistical Deaths (%) and National $PM_{2.5}$ Benefit-per-Truck Estimates (2019$/truck) by Confidence Level Using Di *et al.* (2017) Epidemiology Study and Applying 3% and 7% Discount Rates

	Below LML (<0.02)	LML to 1st Percentile (≥0.02 & <3)	1st to 5th Percentile (≥3 & <6.2)	5th to 10th Percentile (≥6.2 & <7.3)	10th to 25th Percentile (≥7.3 & <9.1)	25th Percentile & Above (≥9.1)
Avoided Premature Statistical Deaths (%)						
National	0%	0%	15%	27%	43%	14%
Benefit-Per-Truck (2019$/truck)						
3% Discount Rate	$5,540	$5,540	$5,540	$4,680	$3,180	$780
7% Discount Rate	$4,130	$4,130	$4,130	$3,490	$2,370	$580

LML = Lowest Measured Level, meaning the minimum observed $PM_{2.5}$ concentration in the original epidemiological study

Table 3: Avoided Premature Statistical Deaths (%) and National Ozone Benefit-per-Truck Estimates (2019$/truck) by Confidence Level Using Zanobetti and Schwartz (2008) Epidemiology Study and Applying 3% and 7% Discount Rates

Less confident **More confident**

	Below LML (<15.1)	LML to 1st Percentile (=15.1)	1st to 5th Percentile (>15.1 & <23.4)	5th to 10th Percentile (≥23.4 & <35.6)	10th to 25th Percentile (≥35.6 & <44.0)	25th Percentile & Above (≥44.0)
Avoided Premature Statistical Deaths (%)						
National	0%	0%	0%	17%	37%	46%
Benefit-Per-Truck (2019$/truck)						
3% Discount Rate	$390	$390	$390	$390	$330	$180
7% Discount Rate	$290	$290	$290	$290	$240	$130

LML = Lowest Measured Level, meaning the minimum observed ozone concentration in the original epidemiological study

As illustrated previously, significant differences exist between the projected concentrations in California and the Rest of U.S., which points to the existence of different patterns in the decline of the benefit-per-truck estimates moving from the "less confident" to the "more confident" end of the benefits estimates scale.[30] Tables 4 through 6 present the benefit-per-truck separately for California and Rest of the U.S. in the same format as that presented above for the national estimates. These tables show that California benefit-per-truck estimates decrease at a slower rate than the Rest of the U.S estimates do, which further widens the significant disparities that were noted in the unadjusted estimates in the prior section.

Table 4 presents the $PM_{2.5}$ benefit-per-truck estimates calculated using Krewski *et al.* (2009) for these two regions. The 3% confidence unadjusted estimate declines from $9,390 to $1,600 per truck for California, while it declines from $4,190 to $20 per truck for the Rest of the U.S. While the estimates for California are about 2 times higher than those for the Rest of the U.S. at the "less confident" end of the spectrum, they are more than 80 times higher at the "more confident" end. About 17% of the benefits in California are projected to occur in locations with baseline concentrations greater than the 25th percentile of the original study; in contrast, the corresponding fraction for benefits estimates across the Rest of the U.S. is less than 1%.

Table 5 presents the corresponding $PM_{2.5}$ benefit-per-truck estimates calculated using Di *et al.* (2017). A somewhat lesser rise in disparity with increasing confidence level is observed, but it is still pronounced. The estimates for California are again about 2 times higher than those for the Rest of the U.S. for the "less confident" estimates, but widen to about 30 times higher at the "more confident" end of the benefits estimate spectrum.

Table 6 presents the ozone benefit-per-truck estimates calculated using Zanobetti and Schwartz (2008) for the two regions. In this case – compared to the $PM_{2.5}$ estimates – a larger disparity in the estimates for the two regions is observed at the "less confident" end of the spectrum, but less at the "more confident" end of the spectrum. That is, the California benefit-per-truck estimates are about 12 times higher than those for the Rest of the U.S. before confidence-weighting, and are about 22 times higher at the other end of the confidence-weighting spectrum.

[30] The Rest of U.S. region includes all states across the conterminous U.S. except for California.

Table 4: Avoided Premature Statistical Deaths (%) and $PM_{2.5}$ Benefit-per-Truck Estimates (2019$/truck) for California and Rest of U.S. by Confidence Level Using Krewski *et al.* (2009) Epidemiology Study and Applying 3% and 7% Discount Rates

	Below LML (<5.8)	LML to 1st Percentile (≥5.8 & <6.7)	1st to 5th Percentile (≥6.7 & <8.8)	5th to 10th Percentile (≥8.8 & <10.2)	10th to 25th Percentile (≥10.2 & <11.8)	25th Percentile & Above (≥11.8)
Avoided Premature Statistical Deaths (%)						
California	4%	5%	24%	27%	23%	17%
Rest of U.S.	11%	18%	63%	8%	<1%	<1%
Benefit-Per-Truck (2019$/truck)						
3% Discount Rate						
California	$9,390	$9,050	$8,530	$6,300	$3,760	$1,600
Rest of U.S.	$4,190	$3,750	$3,000	$360	$30	$20
7% Discount Rate						
California	$6,920	$6,670	$6,290	$4,650	$2,770	$1,180
Rest of U.S.	$3,140	$2,810	$2,250	$270	$20	$10

LML = Lowest Measured Level, meaning the minimum observed $PM_{2.5}$ concentration in the original epidemiological study

Table 5: Avoided Premature Statistical Deaths (%) and $PM_{2.5}$ Benefit-per-Truck Estimates (2019$/truck) for California and Rest of U.S. Using Di *et al.* (2017) Epidemiology Study and Applying 3% and 7% Discount Rates

	Below LML (<0.02)	LML to 1st Percentile (≥0.02 & <3)	1st to 5th Percentile (≥3 & <6.2)	5th to 10th Percentile (≥6.2 & <7.3)	10th to 25th Percentile (≥7.3 & <9.1)	25th Percentile & Above (≥9.1)
Avoided Premature Statistical Deaths (%)						
California	0%	0%	5%	11%	25%	60%
Rest of U.S.	0%	0%	18%	31%	47%	4%
Benefit-Per-Truck (2019$/truck)						
3% Discount Rate						
California	$11,160	$11,160	$11,160	$10,620	$9,430	$6,660
Rest of U.S.	$5,080	$5,080	$5,080	$4,180	$2,620	$210
7% Discount Rate						
California	$8,180	$8,180	$8,180	$7,780	$6,910	$4,880
Rest of U.S.	$3,790	$3,790	$3,790	$3,120	$1,950	$160

LML = Lowest Measured Level, meaning the minimum observed $PM_{2.5}$ concentration in the original epidemiological study

Table 6: Avoided Premature Statistical Deaths (%) and Ozone Benefit-per-Truck Estimates (2019$/truck) for California and Rest of U.S. by Confidence Level Using Zanobetti and Schwartz (2008) Epidemiology Study and Applying 3% and 7% Discount Rates

	Below LML (<15.1)	LML to 1st Percentile (=15.1)	1st to 5th Percentile (>15.1 & <23.4)	5th to 10th Percentile (≥23.4 & <35.6)	10th to 25th Percentile (≥35.6 & <44.0)	25th Percentile & Above (≥44.0)
Avoided Premature Statistical Deaths (%)						
California	0%	0%	0%	12%	30%	58%
Rest of U.S.	0%	0%	0%	22%	46%	32%
Benefit-Per-Truck (2019$/truck)						
3% Discount Rate						
California	$2,570	$2,570	$2,570	$2,570	$2,250	$1,490
Rest of U.S.	$210	$210	$210	$210	$160	$70
7% Discount Rate						
California	$1,890	$1,890	$1,890	$1,890	$1,660	$1,100
Rest of U.S.	$150	$150	$150	$150	$120	$50

LML = Lowest Measured Level, meaning the minimum observed ozone concentration in the original epidemiological study

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Appendix A: Estimated Total NO_x Emissions Reductions Including All Model Years, by State

	2027	2028	2029	2030	2031	2032	2033	2034	2035	2036	2037	2038	2039	2040	2041	2042	2043	2044	2045	2046	2047	2048	2049	2050
U.S.	32,336	64,986	97,905	131,009	167,862	202,670	236,023	267,874	297,564	324,976	350,272	373,253	392,157	413,119	430,429	446,171	460,339	473,697	486,102	497,823	508,892	519,411	529,468	539,102
Alabama	684	1,374	2,070	2,769	3,549	4,286	4,992	5,666	6,295	6,875	7,410	7,896	8,337	8,740	9,106	9,439	9,739	10,021	10,284	10,532	10,766	10,988	11,201	11,405
Arizona	760	1,528	2,302	3,081	3,934	4,740	5,512	6,249	6,937	7,572	8,159	8,692	9,176	9,618	10,020	10,386	10,716	11,027	11,316	11,589	11,848	12,093	12,328	12,553
Arkansas	480	965	1,454	1,945	2,478	2,982	3,464	3,925	4,355	4,752	5,119	5,453	5,755	6,033	6,285	6,514	6,721	6,916	7,098	7,270	7,432	7,587	7,735	7,876
California	2,592	5,207	7,842	10,492	13,558	16,453	19,230	21,878	24,342	26,616	28,711	30,612	32,332	33,901	35,327	36,623	37,787	38,883	39,897	40,855	41,758	42,616	43,435	44,219
Colorado	544	1,094	1,649	2,206	2,825	3,409	3,970	4,504	5,003	5,463	5,888	6,274	6,624	6,944	7,235	7,500	7,738	7,962	8,171	8,368	8,554	8,731	8,900	9,062
Connecticut	204	411	618	827	1,076	1,311	1,537	1,752	1,952	2,137	2,306	2,460	2,599	2,726	2,841	2,945	3,039	3,127	3,209	3,285	3,358	3,426	3,492	3,554
Delaware	41	82	123	165	218	269	318	364	407	446	483	515	545	572	596	618	637	656	673	689	704	718	731	744
Florida	1,430	2,874	4,328	5,791	7,464	9,044	10,560	12,005	13,351	14,593	15,738	16,777	17,717	18,575	19,355	20,063	20,700	21,300	21,856	22,380	22,875	23,345	23,794	24,223
Georgia	1,352	2,717	4,094	5,478	6,998	8,433	9,808	11,121	12,347	13,478	14,523	15,473	16,334	17,122	17,839	18,491	19,078	19,632	20,147	20,634	21,094	21,531	21,949	22,350
Idaho	227	456	687	919	1,172	1,410	1,639	1,857	2,061	2,249	2,423	2,581	2,724	2,856	2,975	3,084	3,182	3,274	3,360	3,441	3,518	3,591	3,661	3,728
Illinois	564	1,132	1,704	2,278	2,984	3,651	4,292	4,901	5,467	5,988	6,467	6,901	7,292	7,648	7,971	8,264	8,526	8,773	9,000	9,214	9,416	9,606	9,789	9,963
Indiana	848	1,705	2,569	3,437	4,402	5,314	6,187	7,021	7,798	8,516	9,178	9,780	10,325	10,824	11,277	11,690	12,061	12,410	12,735	13,042	13,332	13,608	13,871	14,124
Iowa	486	977	1,472	1,970	2,509	3,017	3,504	3,970	4,404	4,806	5,177	5,514	5,820	6,101	6,355	6,587	6,797	6,994	7,178	7,351	7,516	7,672	7,821	7,964
Kansas	359	722	1,088	1,456	1,863	2,248	2,616	2,968	3,296	3,599	3,879	4,133	4,364	4,574	4,766	4,940	5,097	5,245	5,383	5,512	5,635	5,752	5,863	5,970
Kentucky	868	1,745	2,630	3,520	4,471	5,369	6,228	7,051	7,818	8,528	9,184	9,780	10,322	10,819	11,270	11,681	12,052	12,402	12,728	13,037	13,329	13,606	13,872	14,127
Louisiana	539	1,083	1,631	2,183	2,790	3,363	3,912	4,436	4,926	5,377	5,794	6,174	6,517	6,832	7,118	7,378	7,612	7,833	8,039	8,233	8,416	8,590	8,757	8,917
Maine	244	491	740	990	1,260	1,515	1,759	1,992	2,209	2,411	2,596	2,765	2,919	3,059	3,187	3,304	3,408	3,507	3,600	3,687	3,769	3,848	3,923	3,995
Maryland	498	1,000	1,506	2,016	2,599	3,149	3,677	4,180	4,649	5,082	5,481	5,843	6,171	6,470	6,742	6,989	7,211	7,420	7,614	7,797	7,970	8,134	8,290	8,440
Massachusetts	504	1,013	1,526	2,042	2,630	3,185	3,717	4,225	4,698	5,135	5,537	5,902	6,233	6,535	6,810	7,059	7,283	7,495	7,691	7,876	8,050	8,216	8,374	8,526
Michigan	1,153	2,318	3,492	4,673	5,980	7,214	8,396	9,526	10,579	11,551	12,449	13,264	14,004	14,680	15,294	15,853	16,357	16,831	17,272	17,689	18,082	18,457	18,814	19,157
Minnesota	563	1,131	1,703	2,279	2,932	3,549	4,141	4,706	5,231	5,717	6,164	6,570	6,938	7,274	7,580	7,857	8,107	8,342	8,560	8,766	8,961	9,145	9,322	9,491
Mississippi	420	845	1,273	1,703	2,183	2,636	3,070	3,485	3,871	4,228	4,557	4,856	5,127	5,375	5,600	5,805	5,989	6,163	6,325	6,477	6,621	6,758	6,889	7,014
Missouri	822	1,651	2,488	3,330	4,253	5,126	5,961	6,759	7,504	8,192	8,827	9,404	9,928	10,407	10,842	11,239	11,596	11,933	12,246	12,542	12,822	13,088	13,342	13,586
Montana	231	465	701	939	1,191	1,430	1,658	1,877	2,081	2,269	2,444	2,602	2,746	2,878	2,998	3,108	3,206	3,299	3,386	3,468	3,546	3,620	3,691	3,759
Nebraska	345	694	1,046	1,401	1,779	2,136	2,478	2,805	3,110	3,392	3,653	3,891	4,106	4,304	4,483	4,647	4,794	4,933	5,063	5,186	5,302	5,412	5,518	5,619
Nevada	270	542	816	1,092	1,400	1,690	1,968	2,234	2,481	2,710	2,920	3,112	3,286	3,444	3,588	3,720	3,838	3,949	4,052	4,150	4,242	4,330	4,413	4,494
New Hampshire	140	282	425	568	730	882	1,028	1,168	1,297	1,417	1,528	1,628	1,719	1,803	1,878	1,947	2,009	2,067	2,121	2,172	2,221	2,266	2,310	2,352
New Jersey	1,648	3,314	4,994	6,685	8,463	10,144	11,752	13,291	14,728	16,058	17,287	18,406	17,500	20,356	21,204	21,976	22,674	23,332	23,947	24,529	25,079	25,603	26,104	26,585
New Mexico	430	865	1,303	1,745	2,218	2,666	3,095	3,504	3,887	4,240	4,567	4,864	5,134	5,381	5,606	5,810	5,995	6,169	6,331	6,484	6,629	6,767	6,899	7,025

Exhibit B

	2027	2028	2029	2030	2031	2032	2033	2034	2035	2036	2037	2038	2039	2040	2041	2042	2043	2044	2045	2046	2047	2048	2049	2050
New York	1,068	2,145	3,231	4,322	5,586	6,779	7,924	9,016	10,031	10,968	11,832	12,615	13,324	13,970	14,558	15,091	15,571	16,022	16,440	16,835	17,207	17,560	17,897	18,220
North Carolina	925	1,858	2,799	3,745	4,829	5,853	6,835	7,771	8,643	9,448	10,190	10,863	11,472	12,028	12,534	12,993	13,406	13,795	14,155	14,495	14,816	15,121	15,412	15,690
North Dakota	160	322	486	650	827	994	1,154	1,307	1,449	1,581	1,703	1,814	1,914	2,007	2,090	2,167	2,236	2,300	2,361	2,418	2,472	2,524	2,573	2,620
Ohio	1,145	2,300	3,465	4,635	5,967	7,225	8,431	9,582	10,654	11,642	12,554	13,382	14,131	14,816	15,437	16,003	16,511	16,990	17,433	17,852	18,248	18,623	18,982	19,326
Oklahoma	555	1,116	1,681	2,250	2,878	3,472	4,040	4,583	5,090	5,557	5,989	6,381	6,737	7,062	7,358	7,627	7,869	8,098	8,310	8,510	8,700	8,880	9,052	9,217
Oregon	411	827	1,246	1,667	2,131	2,570	2,990	3,392	3,766	4,112	4,431	4,721	4,984	5,225	5,444	5,643	5,822	5,991	6,148	6,296	6,437	6,570	6,697	6,820
Pennsylvania	1,210	2,433	3,665	4,905	6,272	7,562	8,799	9,980	11,082	12,099	13,038	13,891	14,665	15,372	16,016	16,601	17,128	17,625	18,087	18,524	18,936	19,328	19,703	20,062
Rhode Island	72	144	217	291	386	475	561	643	719	789	853	910	962	1,010	1,052	1,091	1,126	1,158	1,188	1,216	1,243	1,268	1,291	1,314
South Carolina	699	1,405	2,118	2,834	3,609	4,341	5,042	5,712	6,338	6,916	7,449	7,935	8,375	8,779	9,146	9,479	9,780	10,064	10,329	10,578	10,815	11,039	11,254	11,460
South Dakota	188	377	569	761	965	1,158	1,342	1,519	1,683	1,836	1,977	2,105	2,221	2,328	2,425	2,514	2,593	2,669	2,739	2,805	2,868	2,928	2,985	3,040
Tennessee	899	1,808	2,724	3,645	4,654	5,607	6,519	7,392	8,205	8,957	9,651	10,281	10,853	11,377	11,853	12,286	12,676	13,043	13,386	13,709	14,015	14,305	14,583	14,849
Texas	2,419	4,860	7,322	9,798	12,573	15,193	17,705	20,103	22,338	24,401	26,304	28,033	29,599	31,031	32,332	33,516	34,580	35,584	36,516	37,396	38,226	39,016	39,771	40,493
Utah	268	538	810	1,084	1,392	1,684	1,963	2,229	2,478	2,707	2,918	3,110	3,284	3,443	3,587	3,719	3,837	3,948	4,051	4,149	4,241	4,328	4,412	4,492
Vermont	127	255	384	514	653	785	911	1,032	1,144	1,248	1,344	1,432	1,511	1,584	1,650	1,710	1,765	1,816	1,864	1,909	1,952	1,992	2,031	2,068
Virginia	1,005	2,020	3,044	4,073	5,207	6,279	7,305	8,286	9,201	10,045	10,825	11,533	12,175	12,763	13,297	13,783	14,221	14,634	15,017	15,380	15,722	16,048	16,359	16,658
Washington	700	1,407	2,120	2,837	3,627	4,373	5,088	5,771	6,407	6,995	7,538	8,031	8,479	8,888	9,260	9,598	9,903	10,190	10,458	10,710	10,949	11,175	11,392	11,600
West Virginia	275	552	832	1,113	1,417	1,705	1,980	2,243	2,489	2,716	2,925	3,116	3,289	3,448	3,592	3,723	3,841	3,952	4,056	4,155	4,247	4,336	4,420	4,501
Wisconsin	747	1,501	2,262	3,026	3,867	4,662	5,423	6,150	6,828	7,454	8,032	8,557	9,034	9,470	9,866	10,227	10,552	10,858	11,143	11,412	11,666	11,908	12,139	12,361
Wyoming	216	435	655	877	1,111	1,331	1,542	1,744	1,933	2,107	2,269	2,415	2,549	2,671	2,783	2,884	2,975	3,062	3,143	3,219	3,291	3,360	3,426	3,489

Appendix B: Benefit-per-Ton Estimates by State

	Zanobetti and Schwartz (2008); Respiratory Mortality (2019$/ton)			Krewski *et al.* (2009); All-Cause Mortality (2019$/ton)			Di *et al.* (2017); All-Cause Mortality (2019$/ton)		
	2030	2040	2050	2030	2040	2050	2030	2040	2050
U.S.	$569	$706	$795	$6,980	$7,856	$8,047	$8,310	$9,715	$10,129
Alabama	$208	$225	$218	$883	$946	$913	$1,013	$1,135	$1,114
Arizona	$701	$904	$1,084	$383	$489	$581	$443	$596	$722
Arkansas	$91	$100	$98	$2,063	$2,265	$2,252	$2,366	$2,704	$2,734
California	$3,643	$4,543	$5,246	$13,542	$16,372	$18,700	$15,714	$19,943	$23,412
Colorado	$511	$608	$661	$4,122	$4,886	$5,432	$4,811	$5,940	$6,772
Connecticut	$112	$129	$129	$8,069	$9,116	$9,068	$9,933	$11,603	$11,652
Delaware	$67	$73	$74	$20,708	$24,025	$25,156	$24,515	$29,594	$31,407
Florida	$799	$1,019	$1,204	$50	$63	$74	$60	$79	$94
Georgia	$358	$432	$471	$950	$1,144	$1,267	$1,093	$1,391	$1,571
Idaho	$32	$38	$41	$3,409	$4,131	$4,565	$4,105	$5,151	$5,807
Illinois	$102	$114	$114	$17,704	$19,670	$19,916	$21,065	$24,336	$25,141
Indiana	$143	$154	$146	$16,237	$17,468	$16,855	$19,244	$21,484	$21,036
Iowa	$29	$31	$28	$9,528	$9,880	$8,962	$11,643	$12,361	$11,317
Kansas	$28	$29	$27	$5,499	$5,840	$5,479	$6,614	$7,208	$6,838
Kentucky	$163	$173	$162	$6,987	$7,354	$6,903	$8,076	$8,828	$8,386
Louisiana	$168	$186	$186	$301	$326	$323	$343	$384	$390
Maine	$49	$57	$56	$508	$603	$619	$635	$778	$807
Maryland	$158	$196	$218	$11,743	$14,056	$15,473	$13,956	$17,434	$19,649
Massachusetts	$144	$164	$161	$4,947	$5,467	$5,380	$6,040	$6,894	$6,855
Michigan	$232	$257	$252	$19,125	$20,979	$20,742	$22,997	$26,266	$26,318
Minnesota	$82	$95	$93	$11,454	$13,360	$13,410	$14,129	$16,955	$17,224
Mississippi	$125	$137	$135	$990	$1,111	$1,149	$1,117	$1,314	$1,392
Missouri	$61	$65	$60	$5,744	$6,064	$5,640	$6,781	$7,384	$6,940
Montana	$25	$29	$30	$342	$410	$445	$419	$517	$571
Nebraska	$17	$18	$16	$4,766	$5,074	$4,769	$5,760	$6,280	$5,971
Nevada	$822	$1,125	$1,500	$1,164	$1,498	$1,900	$1,370	$1,855	$2,419
New Hampshire	$33	$40	$41	$1,882	$2,262	$2,326	$2,331	$2,902	$3,003

	Zanobetti and Schwartz (2008); Respiratory Mortality (2019$/ton)			Krewski *et al.* (2009); All-Cause Mortality (2019$/ton)			Di *et al.* (2017); All-Cause Mortality (2019$/ton)		
	2030	2040	2050	2030	2040	2050	2030	2040	2050
New Jersey	$225	$262	$266	$14,316	$16,189	$16,444	$17,215	$20,275	$20,924
New Mexico	$85	$106	$125	$231	$286	$338	$271	$351	$428
New York	$391	$441	$448	$10,710	$11,844	$12,058	$12,829	$14,709	$15,260
North Carolina	$322	$383	$412	$2,920	$3,441	$3,711	$3,437	$4,247	$4,654
North Dakota	$15	$18	$18	$1,175	$1,349	$1,368	$1,423	$1,668	$1,712
Ohio	$296	$321	$311	$18,322	$19,875	$19,429	$21,967	$24,723	$24,525
Oklahoma	$128	$133	$124	$3,683	$3,840	$3,614	$4,191	$4,509	$4,306
Oregon	$72	$82	$89	$1,562	$1,755	$1,852	$1,872	$2,171	$2,338
Pennsylvania	$282	$321	$321	$15,420	$17,427	$17,587	$18,958	$22,151	$22,644
Rhode Island	$40	$44	$44	$9,371	$10,479	$10,456	$11,412	$13,213	$13,336
South Carolina	$171	$211	$241	$1,377	$1,660	$1,865	$1,616	$2,046	$2,352
South Dakota	$16	$18	$17	$2,689	$2,902	$2,731	$3,268	$3,600	$3,422
Tennessee	$248	$275	$273	$2,599	$2,839	$2,805	$2,956	$3,365	$3,373
Texas	$946	$1,158	$1,302	$1,224	$1,484	$1,652	$1,407	$1,789	$2,034
Utah	$311	$385	$451	$8,326	$9,840	$10,850	$9,668	$11,855	$13,467
Vermont	$15	$18	$18	$1,770	$2,158	$2,327	$2,219	$2,782	$3,028
Virginia	$260	$321	$355	$2,737	$3,402	$3,875	$3,272	$4,247	$4,944
Washington	$116	$138	$150	$1,614	$1,923	$2,119	$1,941	$2,391	$2,688
West Virginia	$93	$96	$91	$4,023	$4,169	$3,967	$4,729	$5,081	$4,918
Wisconsin	$82	$93	$89	$13,567	$15,146	$14,674	$16,612	$19,123	$18,738
Wyoming	$14	$16	$17	$189	$220	$226	$223	$266	$280

Appendix C: Benefit-per-Truck Estimates by State, 7% Discount Rate

Figure 16: Map of $PM_{2.5}$-Only Benefits-per-Truck by State Using the Krewski *et al.* (2009) C-R Coefficient, 7% Discount Rate

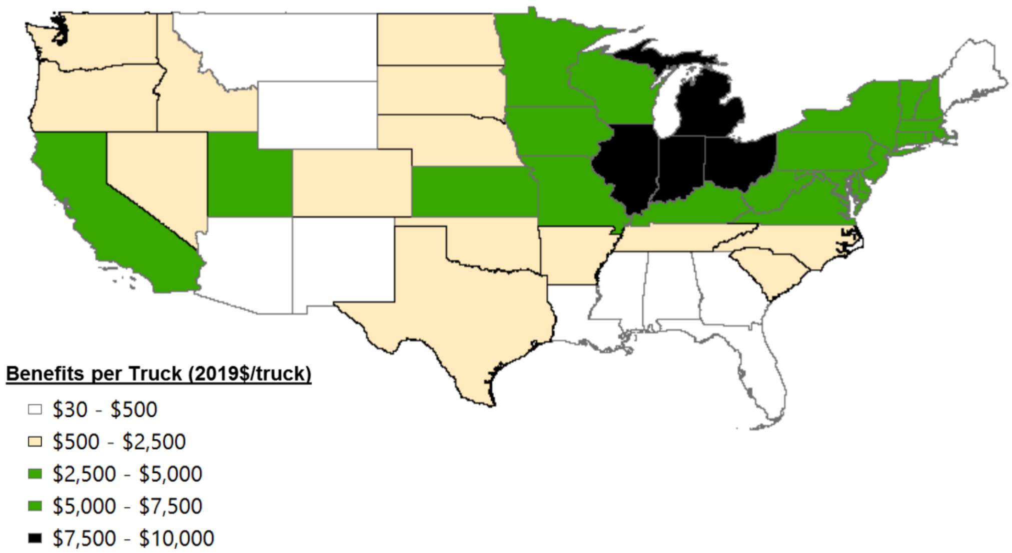


Figure 17: Cumulative Distribution of $PM_{2.5}$-Only Benefits-per-Truck by State Using the Krewski *et al.* (2009) C-R Coefficient, 7% Discount Rate

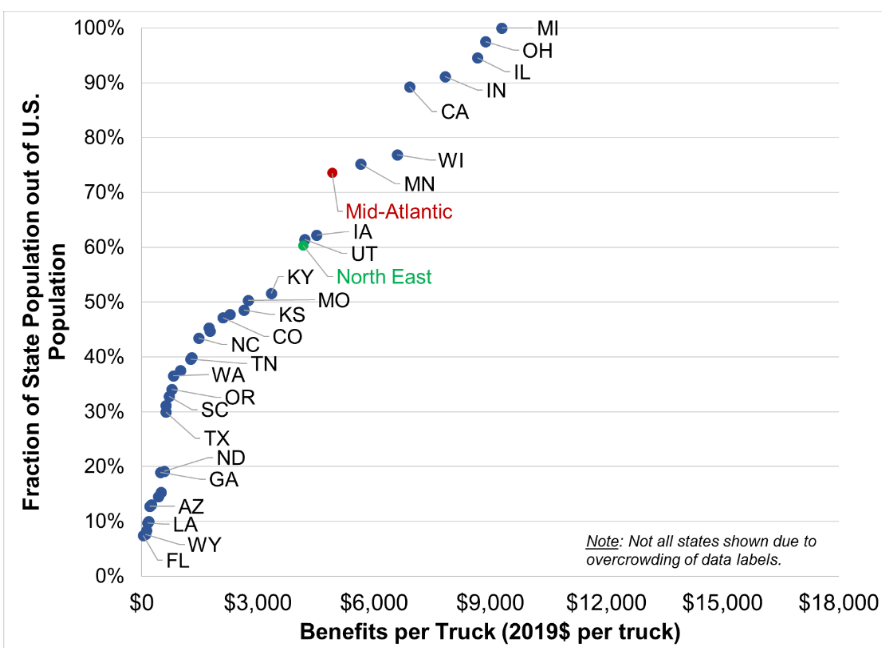


Figure 18: Map of $PM_{2.5}$-Only Benefits-per-Truck by State Using the Di *et al.* (2017) C-R Coefficient, 7% Discount Rate

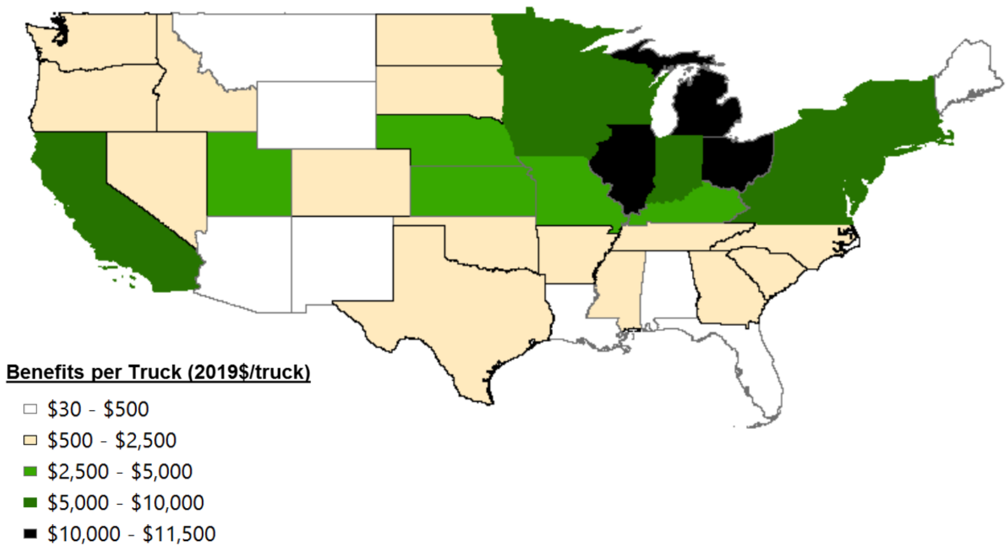


Figure 19: Cumulative Distribution of $PM_{2.5}$-Only Benefits-per-Truck by State Using the Di *et al.* (2017) C-R Coefficient, 7% Discount Rate

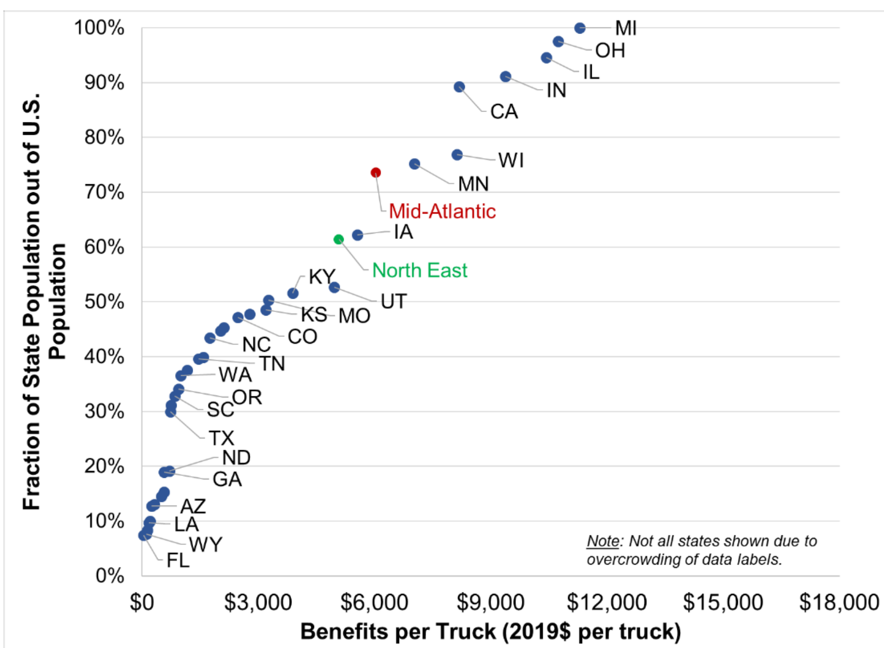


Figure 20: Map of Ozone-Only Benefits-per-Truck by State, 7% Discount Rate

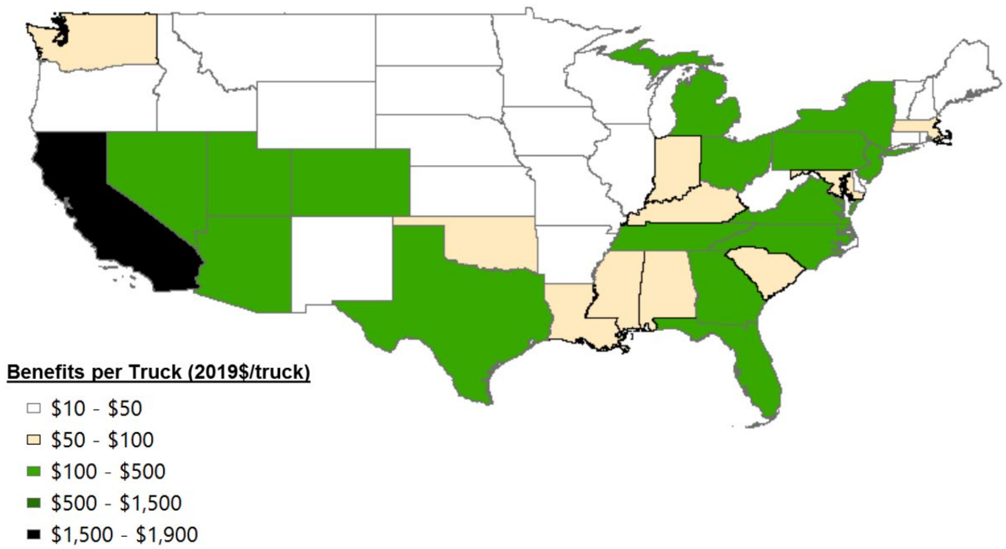
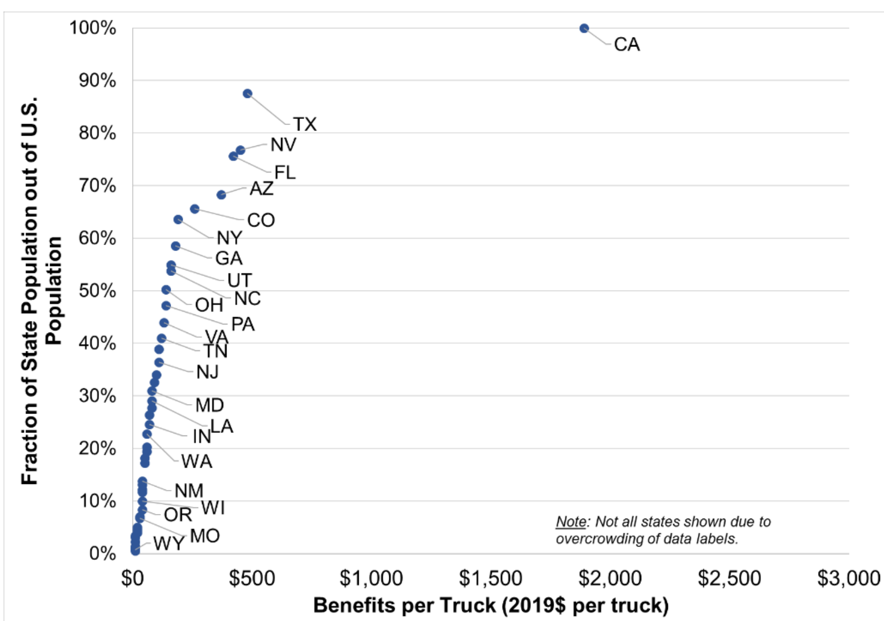


Figure 21: Cumulative Distribution of Ozone-Only Benefits-per-Truck by State, 7% Discount Rate



Appendix D: Estimated Average Ozone Response Factors by State

State	Ozone Response Factor (ppb/ton)
Alabama	0.000022
Arizona	0.000061
Arkansas	0.000014
California	0.000072
Colorado	0.000061
Connecticut	0.000019
Delaware	0.000017
Florida	0.000022
Georgia	0.000022
Idaho	0.000011
Illinois	0.000005
Indiana	0.000012
Iowa	0.000005
Kansas	0.000005
Kentucky	0.000017
Louisiana	0.000022
Maine	0.000016
Maryland	0.000019
Massachusetts	0.000015
Michigan	0.000014
Minnesota	0.000011
Mississippi	0.000022
Missouri	0.000005
Montana	0.000011
Nebraska	0.000005
Nevada	0.000135
New Hampshire	0.000012
New Jersey	0.000019
New Mexico	0.000021
New York	0.000015
North Carolina	0.000017
North Dakota	0.000011
Ohio	0.000014
Oklahoma	0.000018
Oregon	0.000011
Pennsylvania	0.000012
Rhode Island	0.000019
South Carolina	0.000017
South Dakota	0.000011
Tennessee	0.000019
Texas	0.000025
Utah	0.000098
Vermont	0.000010
Virginia	0.000020

State	Ozone Response Factor (ppb/ton)
Washington	0.000011
West Virginia	0.000019
Wisconsin	0.000009
Wyoming	0.000011

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Potential Air Quality Benefits of a 90%/50% Reduction in NO_x Emissions from New Heavy-Duty On-Highway Vehicles

– *Conceptual Summary of Methods and Key Results*

Prepared for the Truck and Engine Manufacturers Association

April 2020

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Contents

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List of Acronyms

ACE	Affordable Clean Energy
BCA	Benefit-Cost Analysis
BenMAP	Benefits Mapping and Analysis Program
CAMx	Comprehensive Air Quality Model with Extensions
C-R	Concentration-Response
EMA	Truck and Engine Manufacturer's Association
EPA	Environmental Protection Agency
FTP	Federal Test Procedure
GVWR	Gross Vehicle Weight Rating
HDOH	Heavy-Duty On-Highway
HHD	Heavy Heavy-Duty Vehicle; Class 8a and 8b Trucks (GVWR > 33,000 lbs)
HHDDV	Heavy Heavy-Duty Diesel Vehicle
LHD<=14k	Light Heavy-Duty Vehicle; Class 2b Trucks with 2 Axles and at least 6 Tires or Class 3 Trucks (8,500 lbs < GVWR <= 14,000 lbs)
LHD45	Light Heavy-Duty Vehicle; Class 4 and 5 Trucks (14,000 lbs < GVWR <= 19,500 lbs)
LHDDV	Light Heavy-Duty Diesel Vehicle
LML	Lowest Measured Level
MHD	Medium Heavy-Duty Vehicle; Class 6 and 7 Trucks (19,500 lbs < GVWR <= 33,000 lbs)
MHDDV	Medium Heavy-Duty Diesel Vehicle
MOVES2014	Motor Vehicle Emission Simulator 2014
NAAQS	National Ambient Air Quality Standards
NERA	NERA Economic Consulting
NOx	Nitrogen Oxides
OMB	Office of Management and Budget
$PM_{2.5}$	Fine Particulate Matter (that have a diameter of less than 2.5 micrometers)
RIA	Regulatory Impact Analysis

I. Introduction

The U.S. Environmental Protection Agency (EPA) announced a "Cleaner Trucks Initiative" in November 2018 to consider lowering the current federal nitrogen oxide (NO_x) standards for heavy-duty on-highway (HDOH) trucks under the provision of the Clean Air Act that authorizes such standards. An Advanced Notice of Proposed Rulemaking soliciting pre-proposal comments primarily on potential truck emissions control technologies was published in the *Federal Register* on January 21, 2020, and a Proposed Rule is expected to be released later in 2020.

Under the Clean Air Act, federal NO_x emissions standards for heavy-duty vehicles must be as stringent as technically feasible given "appropriate consideration of costs."[1] One approach for determining an appropriate cost level (and the one used by EPA in past rulemakings) is to conduct a benefit-cost analysis (BCA) of the tighter NO_x standard. Such BCAs are typically presented in the Regulatory Impact Analyses (RIAs) that EPA must prepare for every major rulemaking.[2]

To obtain insight into the range of potentially justifiable tighter HDOH NO_x standards, the Truck and Engine Manufacturers Association (EMA) engaged NERA to prepare estimates of the air quality benefits that EPA is likely to be able to attribute to a tighter NO_x standard, focusing specifically on the beneficial impacts attributable to a 90% reduction in the current NO_x FTP standard, which EMA estimated could lead to a 50% reduction in the in-use NO_x emissions from new HDOH trucks. This report provides a conceptual overview of NERA's approach and a summary of the main conclusions. More technical details of the data and calculations that NERA utilized are provided in a separate report.

In the case of an air quality regulation, such as that for a lower HDOH emissions standard, the main quantifiable benefits reported in the associated RIA are the societal value of potential improvements in health outcomes from reduced exposures of the U.S. population to the relevant ambient pollutants.[3] Typically, RIAs estimate the total benefits projected to occur in one or more specific future years, after several years of implementation and phase-in of the new emission standard. Those annual estimates are compared to estimates of the annualized incremental costs incurred in the same future years to assess the extent to which benefits are projected to exceed costs. Although there is no formal determination on this matter, one would reasonably expect that benefits must exceed costs (*i.e.*, the benefit-to-cost ratio must be greater than 1:1) in order to conclude that the regulation's costs have been appropriately considered (absent other offsetting or non-quantifiable impacts deemed to be a major concern).

The standard approach that EPA takes in RIAs uses several types of complex models and detailed data inputs, all of which are updated for each new regulatory analysis.[4] This is a highly complex process, and also difficult to emulate in advance of EPA's own analysis without having access to the specific models and data that will be used. One rarely even knows the specific future year(s) that EPA will select as the

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[1] Clean Air Act Section 202(a)(3)(A).

[2] RIAs are required under Executive Orders for every major proposed and final rulemaking of an executive branch agency, such as EPA. A major rulemaking is defined as a new regulation whose costs would exceed $1 million per year. Among other required contents, RIAs must provide estimates of the potential social benefits and costs of a regulation and their implications for the net benefits of the rule. BCAs can, of course, be prepared to evaluate an appropriate cost level outside of a formal RIA, but the upcoming truck emissions rulemaking can be expected to require a formal RIA.

[3] In RIAs, the term "benefit" refers to the monetized societal value that is assigned to a physical estimate of the health risk or environmental damage reduction from a regulation.

[4] The models involved just for the benefits portion of the analysis include emissions inventories and emissions projections models such as MOVES2014, 3-dimensional fate and transport models such as CAMx, and health risk analysis models such as BenMAP.

focus for its benefit and cost calculations. Therefore, a simpler and quicker approach is needed to develop approximate estimates of the maximum per-truck cost that EPA might expect to be able to justify with a full BCA, in order to provide preliminary guidance on which new emission control technologies, and their associated costs, are reasonable to account for in a proposed rule.

NERA has developed such an initial and more straightforward approach, which is described in high-level terms in this report. Our "scoping" approach has been designed around the fact that it will be quicker to categorize the array of potential control technologies in terms of their total cost *per truck* than to estimate what those costs will be when projected over the entire future HDOH fleet and annualized for some specific (yet to be known) future year. The scoping approach also takes into account that if annualized incremental costs in any future year will be less than the annual benefits, then the total lifecycle cost per truck will also have to be less than the present value of the benefit that will be produced (on average) by each truck that would be affected by the rule. Thus, NERA has developed a simplified approach that gauges the potential benefits *per truck* from the assumed tighter NO_x standard. Such per-truck benefits estimates can help identify the scope of the maximum per-truck compliance cost that will be likely to pass muster under a full BCA of the proposed tighter NO_x standard.

We emphasize that the estimates we summarize in the following sections of this report reflect an effort to anticipate what the Agency would estimate if it applied its own usual assumptions and analysis methodologies. In making our estimates of NO_x reduction benefits per truck, we have used analysis input assumptions that we believe are within the range of those that EPA would likely use. Of course, we do not know what may arise with updated EPA models, data, and input assumptions, but we have sought out the most recent studies and documents on air pollutants that EPA has released. Our estimates are nevertheless subject to revision as more up-to-date information is released. The specific assumptions that we have used for the present analyses are the subject of a separate technical report, while this report provides a more qualitative description of the approach and its most central results. Were we to undertake this type of benefits analysis without regard to what EPA is expected to do, it is likely that we would utilize different methods and assumptions.

II. Description of Methodology

The following are the specifics of the new anticipated federal HDOH low-NO_x standard that NERA analyzed:

- A 90% reduction in the Federal Test Procedure (FTP) standard from its current level of 0.2 g/hp-hr down to 0.02 g/hp-hr. For NERA's analysis, EMA provided the assumption that the 90% reduction in the FTP-standard would result in a 50% reduction in baseline in-use emissions for the categories of new HDOH trucks being analyzed.[5]
- Inclusion of all truck-types defined in EPA's emissions inventory model as heavy-duty-diesel and on-road. Specifically, those truck-types include long-haul and short-haul combination trucks, long-haul and short-haul single unit trucks, refuse trucks, school buses, transit buses, and intercity buses (a total of 8 types).
- Implementation of the new lower federal NO_x standard starting in 2027.

[5] This was based on guidance from EMA that the reduction in emissions associated with a 90% FTP standard reduction would be roughly equivalent to a 50% reduction in in-use emissions.

Given the above assumptions regarding the standard to be analyzed, we calculate the benefits per truck associated with a 50% reduction in those trucks' in-use NO_x emissions. The primary purpose of such a low-NO_x emission standard would be to achieve reductions in ambient ozone and fine particulate matter ($PM_{2.5}$) to help states attain or maintain attainment with the NAAQS standards for those two pollutants. Thus, we focus our benefits calculations on the value of projected health risk reductions from the projected reductions in ambient ozone and $PM_{2.5}$ exposures across the U.S. that would result from reduced HDOH truck NO_x emissions across the U.S. due to the implementation of a tighter HDOH NO_x standard.[6] Based on a long history of such benefits calculations (by EPA and many other entities), approximately 98% of estimated health benefits from reductions in ozone and $PM_{2.5}$ is due to reductions in mortality risks. Thus, we simplified our benefit-per-truck estimates by estimating only mortality risk benefits, having confidence that this simplification has no meaningful impact on our numerical conclusions.

In order to obtain per-truck benefit estimates, we first calculate the tons of NO_x emissions reductions from an average new truck that would be purchased in 2027 meeting the tighter NO_x standard, accounting for a potential life of up to 30 years. We do this calculation for each of the 8 truck types covered by the assumed standard. That computation is carried out for each year of a truck's operational life. We assess the average truck's continued operation in each future year based on truck survival rates over time.[7] The emissions reductions in each future year are then translated into a dollar estimate of each year's health benefits using a simple "reduced form" method in which the precursor emissions changes are multiplied by a "benefit per ton" value. EPA routinely uses such an approximation when it wishes to avoid a full, complex benefits analysis.[8]

The result of this methodology is a time line from 2027 through 2057 of annual benefits per truck in each year of the average 2027-vintage truck's operating life that varies across time (generally declining) as the truck ages. This stream of benefits is discounted to obtain the present value of benefits per truck for each of the 8 truck types. Those 8 values are then combined into a single sales-weighted average benefit-per-truck estimate. It is the latter value that can then be compared to the incremental compliance cost per truck to determine whether the costs of the regulation-driven low-NO_x technology is likely to pass a

[6] In this context, the emitted NO_x is called a "precursor" emission because it contributes to the formation of ambient concentrations of ozone and $PM_{2.5}$.

[7] NERA's analysis of the future emission reductions of vintage-2027 trucks extends through 2057, allowing at least some trucks in each category to last at least 30 years. However, those later-year reductions have minimal impact due to there being only a small fraction of trucks surviving that long (hence very few tons of reduction in the later years), and also because the benefits of any emissions reductions in the later years are heavily discounted. The survival rates in that dataset differ for each of the 8 truck-types, and so too in our analysis. Documentation of how we calculated the tons of reduction by year and the specific data sources is available in a separate, more technical report.

[8] A full benefit analysis requires that the specific projected precursor emissions changes be run through an air quality fate and transport model to project geographical changes of the relevant ambient pollutant concentrations. That map of pollutant concentration changes must then be run through a demographic health risk model, with the result being total benefits. The "reduced form" approach provides an approximation by conducting the full linked-model runs for a specific (but generic) number of tons of emissions reduction of a specific type of precursor, then dividing the estimated total benefits for that generic scenario by the tons of reduction. This yields an estimate of benefits stated in dollars-per-ton. This "benefit-per-ton" estimate is then multiplied by the tons of reduction of that precursor predicted for any of a variety of different policies to directly (but very approximately) produce a total benefits estimate without undertaking the complex steps of another full analysis. EPA has already produced and published a number of "benefits per ton" estimates. Although we considered those existing estimates, NERA followed the standard reduced form estimation process described above to derive its own benefits per ton estimates, enabling us to apply more up-to-date assumptions that we believe will be used in a full BCA, to enable us to derive more geographically disaggregated benefits per truck estimates, and to provide a range of estimates that vary in their qualitative confidence. When using the same underlying epidemiological risk relationship, NERA's per-ton benefits estimates are comparable to those published by EPA. The specific methods and resulting benefits per ton estimates are documented in a separate report.

robust benefit-cost test. Consistent with OMB and EPA guidance, we provide benefit-per-truck estimates that are calculated using discount rates of 3% and 7%.

III. Benefit-per-Truck Estimates Prior to Confidence-Weighting

The most important input that drives the benefit-per-ton estimates, and hence the benefit-per-truck estimates, is the assumption about the increase in mortality risk per unit change in ozone and $PM_{2.5}$ concentration. That is usually based on a statistically-derived association between mortality risk and observed pollutant concentrations or exposures, called a concentration-response (C-R) coefficient. The assumed C-R coefficient is usually obtained from one or more of many existing epidemiological studies and associated peer-reviewed papers. EPA tends to change this mortality risk assumption as new epidemiology papers are published and as each NAAQS review cycle is conducted. We reviewed statements in EPA's recent Policy Assessments for $PM_{2.5}$ and ozone (EPA, 2020 and 2019b) to attempt to anticipate which assumptions EPA may adopt in future RIAs. Without commenting on the appropriateness of any such studies, we decided it would be reasonable to provide a range of estimates for the $PM_{2.5}$ benefits per ton. The lower end of the range is based on a C-R coefficient for all-cause mortality risk from the Krewski *et al*. (2009) study, and the higher end of the range is based on a C-R coefficient estimate for all-cause mortality risk from the Di *et al*. (2017) study. For ozone, the recent ozone NAAQS review documents indicate that EPA is giving less causal credence to all-cause mortality risks than in the past, and they provide no quantitative risks based on epidemiological evidence. The ozone Policy Assessment document does, however, identify several epidemiological studies of respiratory health effects for its evidence-based evaluation of potential NAAQS levels, and we focused on those studies for anticipating what the Agency might use if it should include quantified ozone benefits in future RIAs. As a result, we have based our benefit-per-truck estimates for ozone on a risk estimate for respiratory mortality from the Zanobetti and Schwartz (2008) study. One should not, however, dismiss the possibility that the Agency will provide no quantitative estimate of ozone-related mortality benefits in the RIA for a tightened HDOH truck standard in 2020.

There are significant scientific uncertainties introduced when using such statistical associations from epidemiological studies to predict risks for different populations and under different air quality conditions. There are methods for identifying how the uncertainties may be reduced to derive benefits estimates having a higher degree of confidence. That is a complex issue that will be discussed in detail in the next section. However, Table 1 first presents our benefit-per-truck estimates *prior to any adjustment for confidence*. That is, the following raw per-truck benefits estimates assume that the epidemiological estimates of the increase in mortality risk per unit of ambient pollutant concentration are equally reliable no matter what the level of baseline pollutant exposures might be for the population being assessed in the risk analysis.

Table 1: National Ozone and $PM_{2.5}$-Related Benefit-per-Truck Estimates with No Adjustment for Confidence

	Ozone	**$PM_{2.5}$**
National Benefits per Truck (3% Discount Rate)	$390	$4,580 - $5,540
National Benefits per Truck (7% Discount Rate)	$290	$3,430 - $4,130

IV. Benefit-per-Truck Estimates with Qualitative Confidence-Weighting

As mentioned above, the mortality risk estimates for $PM_{2.5}$ and ozone are computed using statistically-derived estimates of associations between ambient pollutant levels in different locations or on different days and their respective mortality rates, often summarized in the form of a "C-R coefficient." The statistical methods of deriving those C-R coefficient estimates make extensive effort to control for a wide range of other drivers of mortality risk to avoid a spurious inference that a positive statistical association implies a causal relationship between the pollutant and elevated mortality risk. Nevertheless, even if there is a sufficiently "causal" relationship within the range of observed pollutant levels, any use of that unit risk estimate to predict changes in risks in different locations and under different levels of exposure necessarily involves extrapolation outside of the original range of data. Extrapolation always introduces uncertainties that are not included in any of the original study's statistical measures of confidence. The more extreme is the extrapolation that a risk analysis requires into exposure and population conditions not representative of the original study, the less qualitative confidence one would have in the derived risk estimate.

Such extrapolation can be a particular problem when using studies of air pollutant-health associations from even the relatively recent past to predict risk in a future year because of the rapid declines in pollutant concentrations that have taken place, and which are projected to continue in the future. For example, the average concentrations of $PM_{2.5}$ experienced by the individuals studied in Krewski *et al.* (2009) fell by 30% during the period from 1980 to 2000 over which their mortality risk levels were being observed. Furthermore, the EPA dataset we have used in this report to project average $PM_{2.5}$ levels in 2035 are another 50% lower (*before* any reductions due to a tightened HDOH NO_x standard) than the average exposures occurring at the *end* of the Krewski *et al.* study's period (*i.e.*, in 2000). As a result, a very large fraction of the health benefit estimate reported in Table 1 above requires use of an assumption that the risk association estimated over the historically much-higher range of pollutant exposures in the Krewski *et al.* study continues to exist when the relevant pollutant levels are far below the originally observed range. That important fact necessarily diminishes the confidence one can have in the estimates of Table 1.

It is possible to adjust the calculated risk estimates to exclude the portions that involve the most extreme amounts of extrapolation from the original study. As the amount of extrapolation in the benefits estimate is reduced, confidence in the resulting estimate is qualitatively improved. This creates a sliding scale of benefits estimates from least confident to most confident. In contrast, the estimates shown in Table 1 above make no exclusions of the calculated risk estimates at all, allowing extrapolation of the risk relationship even where projected baseline concentrations are lower than the lowest measured level (LML) of the original study and hence represent the least confident end of the full spectrum of benefits estimates.[9]

EPA introduced such a sliding confidence scale for its $PM_{2.5}$ co-benefits estimates in a recent RIA (EPA, 2019a), which employed a health risk estimate for all-cause mortality from the Krewski *et al.* (2009) epidemiology study. On that sliding scale, the "more confident" end of the spectrum of mortality risk estimates was calculated by excluding those portions of the underlying risk calculations that applied the original study's risk association to baseline $PM_{2.5}$ pollutant exposures below the 25th percentile of the originally-observed range of $PM_{2.5}$ exposures. The 25th percentile of a data set is generally viewed as the

[9] The Agency uses the acronym LML to denote the 0th percentile of the distribution of exposures in the original study.

point where sparseness of observations begins to undercut the ability to determine if an average slope detected over the entire set of originally-observed exposure levels remains at the lowest of those levels.[10]

Comparison of the exposure distributions in Figure 1 and Figure 2 (below) illustrates the degree of extrapolation involved in our benefits analysis.

- Figure 1 shows the range and population-weighted frequency of observed $PM_{2.5}$ concentrations in the Krewski *et al*. (2009) epidemiology study (using the concentrations estimated at the end of the follow up period, in 1999-2000). This shows that mean concentrations at the end of that epidemiology study were about 14 $\mu g/m^3$ and that 75 percent of those observations were higher than about 12 $\mu g/m^3$ (*i.e.*, higher than the dotted line indicating the 25^{th} percentile). Similarly, 95% of those observations were higher than about 9 $\mu g/m^3$ (*i.e.*, higher than the dotted line indicating the 5^{th} percentile).

- Figure 2 depicts the population-weighted frequency of $PM_{2.5}$ concentrations in California and Rest of U.S. (which comprises the conterminous U.S. other than California) that EPA projects will occur in 2035 (which is the period in which a majority of the anticipated HDOH low-NO_x benefits will be accruing). The vertical dotted lines indicate the 5^{th}, 10^{th} and 25^{th} percentiles of the original Krewski study's pollutant observations (*i.e.*, same as in Figure 1). For the Rest of U.S., one can see that the mean $PM_{2.5}$ concentration is about 7 $\mu g/m^3$, and almost none of the projected baseline exposures exceed the original study's 25^{th} percentile of $PM_{2.5}$ concentrations. Projected $PM_{2.5}$ levels in California are, as expected, significantly higher, but even so, less than 10% of the California population are exposed to $PM_{2.5}$ levels higher than the 25^{th} percentile of the original epidemiological study.

[10] It is notable that EPA's numerical implementation of this qualitative rating ends at the 25^{th} percentile, because EPA actually ascribes even greater confidence to estimates of risk nearer the mean of the observations in the original study. (See, for example, Figure 4-1 on p. 4-26 of the 2019 ACE Rule RIA (EPA, 2019a).

Figure 1: Range of Exposures During 1999-2000 Used in the Krewski *et al.* (2009) Epidemiology Study to Estimate the C-R Relationship Used for Benefits Calculations in this Analysis

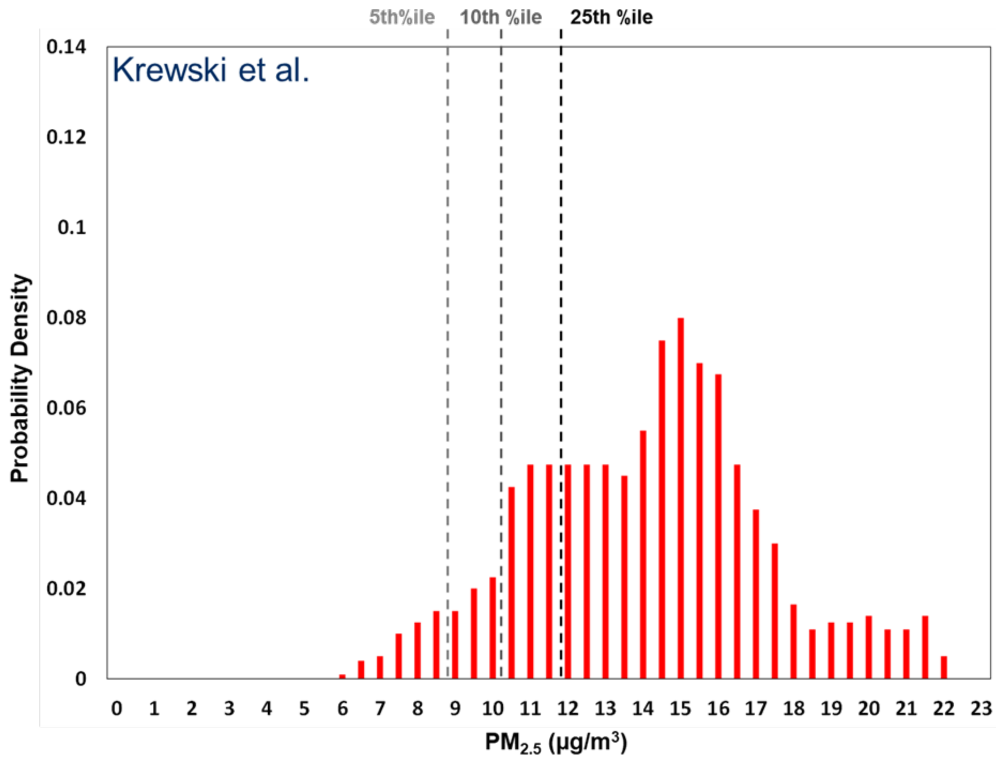
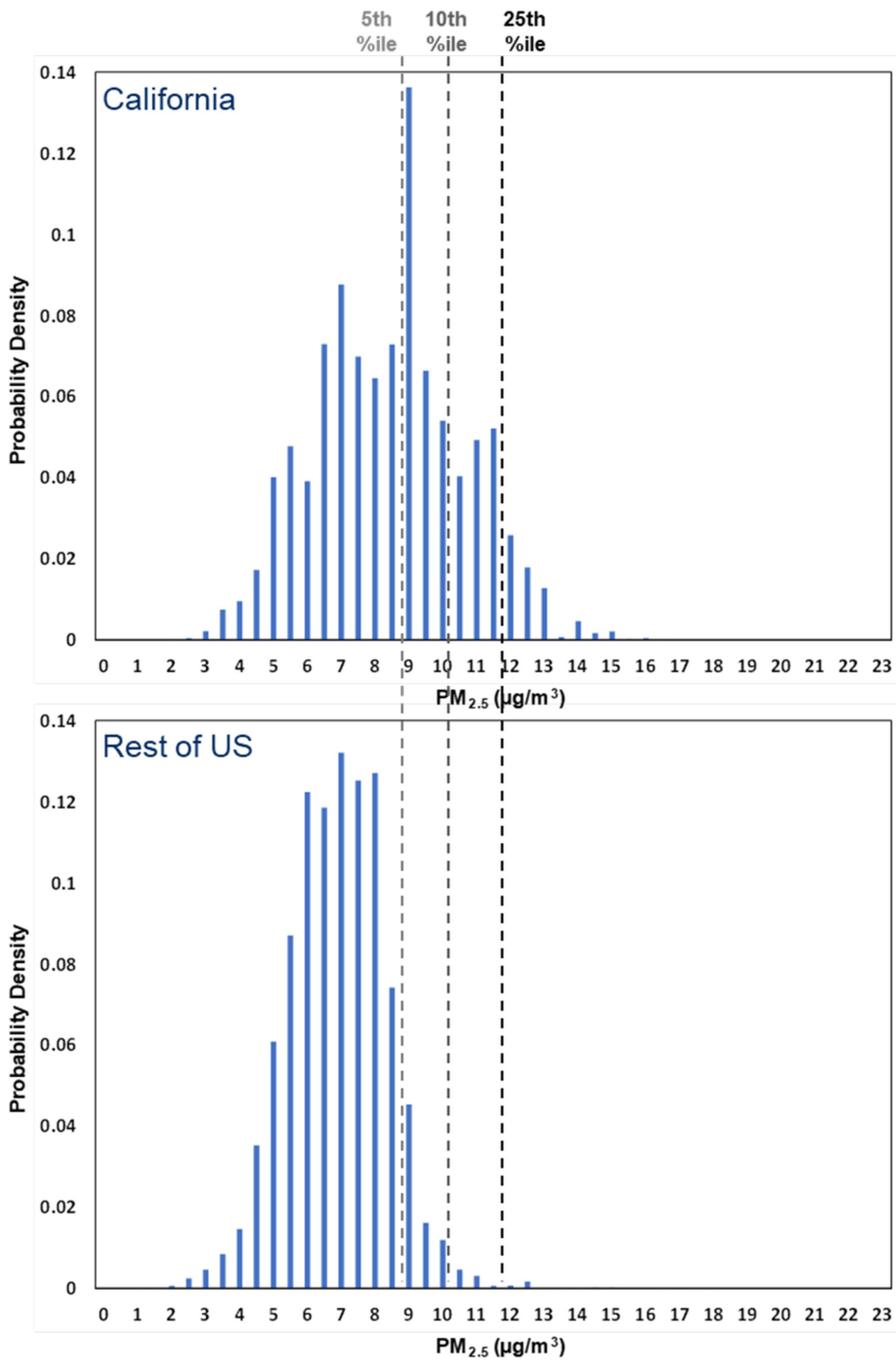


Figure 2: Range of Projected $PM_{2.5}$ Concentrations in California and Rest of U.S.



Thus, the reliability of predicted risk reductions in our benefits analysis is affected by a significant degree of extrapolation outside of the exposure range of the original epidemiology study that provided an indication (and quantification) of a risk relationship.[11] We next provide alternative estimates of our benefit-per-truck calculations that attempt to limit this extrapolation to varying degrees.

In developing our alternative confidence-adjusted estimates, we have used EPA's method (in EPA, 2019a) to assess how the benefit-per-truck estimates in Table 1 might be adjusted to gain confidence that they do not attribute health effects to exposure levels far outside the range that the underlying epidemiological study considered. In applying this method, we have compared our $PM_{2.5}$ and ozone exposure data (for the year 2035) to each respective original studies' distribution of exposures.[12]

Table 2 (below) shows how our $PM_{2.5}$-related benefit-per-truck estimates for $PM_{2.5}$ (in Table 1 above) are adjusted for confidence by this method. Table 2 presents a continuum of confidence-adjusted ozone benefit-per-truck estimates over a range of increasing limitations on the degree of extrapolation allowed in the risk calculations. The first column in each table contains the same estimates reported in Table 1 (*i.e.,* calculated without any limitations on extrapolation in the risk calculation) and the values in the columns to the right show estimates that have increasingly higher confidence (due to progressively reduced reliance on extrapolation), up to the point where only benefits in areas with exposures at or above the 25th percentile of the original epidemiological study are included. Clearly, requiring more confidence in the benefit-per-truck estimates causes the estimates to decline since we exclude benefits that are in areas with projected baseline concentrations that are below various percentile levels of the pollutant observations in the original study (up to the 25th percentile). For instance, the benefit-per-truck estimate of $4,580 for the lower bound in $PM_{2.5}$ exposures (using the 3% discount rate) declines to only $160 at the "more confident" higher end of the exposure spectrum (*i.e.*, the lower estimate in last column of Table 2). This is a dramatic reduction and suggests that the unadjusted risk estimates for current and future air quality based on the Krewski *et al.* (2009) study (the epidemiological basis for the lower $PM_{2.5}$-related benefit-per-truck estimates) are subject to an exceptional amount of potential error due to the necessary extrapolation outside of that study's range of observed exposures and study populations. The uncertainty due to extrapolation is much less pronounced when using the Di *et al.* (2017) study (the basis for the higher $PM_{2.5}$-related benefit-per-truck estimates), which used model-based estimates of ambient $PM_{2.5}$ to enable inclusion of individuals in lower-PM areas that were not monitored.[13]

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[11] The distribution of $PM_{2.5}$ observations depicted in Figure 1 are those that were used to estimate the specific C-R being used for benefits calculations in this analysis. However, Krewski *et al.* also estimated C-R coefficients using observed exposures from the earlier years of the 20-year cohort study. The distribution of concentrations observed at the start of that study sits about 50% to the right of the one in Figure 1, *and* it produces risk estimates about 33% lower. The correct C-R estimate to use is highly uncertain because it requires an assumption on which exposure window best explains the observed association – a scientific unknown that has not been answered by the available statistics. It is worth noting, however, that use of the earlier exposure window from the Krewski *et al*. study would reduce benefits estimates based on that study by about one-third and would result in even greater sensitivity to confidence adjustments than is presented in the next portion of this report.

[12] That is, while we use the distribution in Figure 1 to develop confidence-weighted adjustments for our lower estimates of $PM_{2.5}$ benefits-per-truck because they are based on a risk association reported in Krewski *et al*. (2009), we use information on the distribution of $PM_{2.5}$ exposures in the Di *et al*. (2017) study to develop confidence-weighted adjustments for our higher estimates. We use information on the distribution of city-specific average ozone concentrations in the Zanobetti and Schwartz (2008) study for adjusting our ozone benefits-per-truck estimates.

[13] The use of modeled rather than monitored $PM_{2.5}$ data raises its own risk estimation uncertainties in place of a reduction in the out-of-sample extrapolation error that we address here. We make no attempt to adjust for those other uncertainties in this analysis, as we are only attempting to emulate methods that the Agency has itself used in its prior RIAs. (We note that a large portion of the modeled exposure in Di *et al*. are actually lower than any of the exposures in the Agency's modeling of current U.S. $PM_{2.5}$ levels, which indicates a methodological inconsistency that merits future attention.)

Table 2: National $PM_{2.5}$ Benefit-Per-Truck Estimates (2019$/truck) Adjusted by Confidence Level Based on Health Effect Estimates from the Krewski *et al.* (2009) and Di *et al.* (2017) Epidemiology Studies, Applying 3% and 7% Discount Rates

Less confident **More confident**

	No Adjustment	LML and Above	1st Percentile and Above	5th Percentile and Above	10th Percentile and Above	25th Percentile and Above
3% Discount Rate	$4,580-$5,540	$4,150-$5,540	$3,440-$5,540	$870-$4,680	$360-$3,180	$160-$780
7% Discount Rate	$3,430-$4,130	$3,110-$4,130	$2,570-$4,130	$650-$3,490	$270-$2,370	$120-$580

LML = Lowest Measured Level, meaning the minimum observed $PM_{2.5}$ concentration in the original epidemiological study

Table 3: National Ozone Benefit-Per-Truck Estimates (2019$/truck) Adjusted by Confidence Level Based on a Health Effect Estimate from the Zanobetti and Schwartz (2008) Epidemiology Study, Applying 3% and 7% Discount Rates

Less confident **More confident**

	No Adjustment	LML and Above	1st Percentile and Above	5th Percentile and Above	10th Percentile and Above	25th Percentile and Above
3% Discount Rate	$390	$390	$390	$390	$330	$180
7% Discount Rate	$290	$290	$290	$290	$240	$130

LML = Lowest Measured Level, meaning the minimum observed ozone concentration in the original epidemiological study

There is no way to select a single "best" cut-off point for limiting extrapolation uncertainties. In its last $PM_{2.5}$ NAAQS decision (*i.e.*, the 2013 rulemaking), the Administrator discussed how insufficient confidence in the continued existence of health risk associations would arise somewhere between the 10th to 25th percentiles of a study's range of observations. She chose to set the standard near the lowest of the 25th percentiles of available studies. Based on that precedent, one could consider choosing to limit the benefit-per-truck estimates to those occurring in locations with exposures at or above the 25th percentile. In that case, our analysis indicates that the national average total benefits-per-truck *might be between $340 and $960* if using a 3% discount rate.[14] It would be somewhat lower if using a 7% discount rate. If one were instead to use the 10th percentile as the confidence cut-off, our analysis indicates that the national average total benefits-per-truck *might be between $690 and $3,510* if using a 3% discount rate, and somewhat lower still if using a 7% discount rate.[15]

The main conclusion is that, even accounting for much more recent $PM_{2.5}$ studies, a national average estimate of the combined $PM_{2.5}$ and ozone benefits-per-truck that includes adjustments for extrapolation uncertainties consistent with prior Administrator judgments would not likely exceed $4,000 per truck.

The above statement is based on a national average estimate of benefits, which is the typical way that EPA conducts its BCAs. Note, however, that Figure 2 shows significant differences in the projected $PM_{2.5}$ concentration distributions that exist between California and Rest of U.S. This suggests that there could be significantly different patterns in the confidence that this method would assign to the benefit-per-truck estimates for those two regions. It also suggests that even the raw (unadjusted) benefit-per-truck might be significantly higher for trucks operating in California than for those outside of California.

To understand this better, we have recomputed our benefits-per-truck for California and for the Rest of the U.S. separately. The results, including respective effects of confidence-adjustments, are provided in Table 4 (for $PM_{2.5}$) and Table 5 (for ozone). Those tables highlight the wide disparity in the benefit-per-truck estimates that exist for the two regions, with total per-truck benefits possibly as high as $11,680 in California even with a substantial confidence adjustment (*i.e.*, using the 10th percentile cut-off and a 3% discount rate), *while the equivalent per-truck benefits for the Rest of U.S. would likely not exceed $3,000.*[16]

[14] This range includes both ozone and $PM_{2.5}$ benefits and is the sum of the values in the last column of Tables 2 and 3.

[15] This is computed by summing the values in the penultimate columns of Table 2 and Table 3.

[16] These estimates sum the respective values in the penultimate columns of Table 4 and Table 5.

Table 4: Range of $PM_{2.5}$ Benefit-Per-Truck Estimates (2019$/truck) for California and Rest of U.S. Adjusted by Confidence Level Based on the Health Effect Estimates from the Krewski *et al*. (2009) and Di *et al*. (2017) Epidemiology Studies, Applying 3% and 7% Discount Rates

	No Adjustment	LML and Above	1st Percentile and Above	5th Percentile and Above	10th Percentile and Above	25th Percentile and Above
3% Discount Rate						
California	$9,390-$11,160	$9,050-$11,160	$8,530-$11,160	$6,300-$10,620	$3,760-$9,430	$1,600-$6,660
Rest of U.S.	$4,190-$5,080	$3,750-$5,080	$3,000-$5,080	$360-$4,180	$30-$2,620	$20-$210
National	$4,580-$5,540	$4,150-$5,540	$3,440-$5,540	$870-$4,680	$360-$3,180	$160-$780
7% Discount Rate						
California	$6,920-$8,180	$6,670-$8,180	$6,290-$8,180	$4,650-$7,780	$2,770-$6,910	$1,180-$4,880
Rest of U.S.	$3,140-$3,790	$2,810-$3,790	$2,250-$3,790	$270-$3,120	$20-$1,950	$10-$160
National	$3,430-$4,130	$3,110-$4,130	$2,570-$4,130	$650-$3,490	$270-$2,370	$120-$580

LML = Lowest Measured Level, meaning the minimum observed $PM_{2.5}$ concentration in the original epidemiological study

Table 5: Ozone Benefit-Per-Truck Estimates (2019$/truck) for California and Rest of U.S. Adjusted by Confidence Level Based on the Health Effect Estimates from the Zanobetti and Schwartz (2008) Epidemiology Study, Applying 3% and 7% Discount Rates

	No Adjustment	LML and Above	1st Percentile and Above	5th Percentile and Above	10th Percentile and Above	25th Percentile and Above
3% Discount Rate						
California	$2,570	$2,570	$2,570	$2,570	$2,250	$1,490
Rest of U.S.	$210	$210	$210	$210	$160	$70
National	$390	$390	$390	$390	$330	$180
7% Discount Rate						
California	$1,890	$1,890	$1,890	$1,890	$1,660	$1,100
Rest of U.S.	$150	$150	$150	$150	$120	$50
National	$290	$290	$290	$290	$240	$130

LML = Lowest Measured Level, meaning the minimum observed ozone concentration in the original epidemiological study

V. Conclusion

If a BCA is to be used to assess the level of cost that might be warranted to implement a tighter HDOH NO_x standard, it is reasonable, as an initial scoping exercise, to attempt to assess the maximum lifecycle cost per truck that might be justifiable, before a specific HDOH standard is proposed and a more complex, resource-intensive full BCA is prepared. Having such *ex ante* insights can help guide regulators towards regulatory proposals that will readily pass the more rigorous BCA test. To that end, NERA has developed rough estimates of the potential lifecycle per-truck benefits that one might expect to result from such a complete BCA, and has addressed issues of confidence that might be associated with such estimates. Our analysis has limitations but has been based on data and studies that are currently available, and has taken into consideration the current status of Agency discussions regarding the health risks driving $PM_{2.5}$ and ozone NAAQS decisions. In this report, we have explained our approach at a conceptual rather than technical level. The many assumptions that we have used, and the studies and data that we applied to set those assumptions, are documented in a separate technical report.

The goal of our analysis has been to develop approximate estimates of the per-truck lifecycle benefits associated with a 90% reduction in the FTP NO_x standard for HDOH trucks, and a corresponding 50% reduction in in-use NO_x emissions. We emphasize that the estimates we report here reflect an effort to anticipate what the Agency itself would estimate if it applied its own usual assumptions and analysis methodologies in a formal RIA, expected to be released later in 2020. We also note that our estimates have been based on data and modeling that the Agency has released in the past. Those will probably be replaced by updated information developed as part of the upcoming HDOH RIA. As there is no publicly available information on the nature of such updates, our present estimates are imprecise and subject to revision as such updated information becomes available. As noted above, were we to undertake this type of benefits analysis without regard to what we anticipate EPA is likely to do, it is likely that we would utilize different methods and assumptions.

We find that, *prior to any confidence weighting*, the Agency might determine that a 90% reduction in the FTP NO_x standard for HDOH (with a corresponding 50% reduction in-use NO_x emissions) would result in national average benefits-per-truck for 2027 model year trucks in the range of (roughly) \$5,000 to \$6,000 (for $PM_{2.5}$ and ozone combined). When confidence-adjusted for the multiple uncertainties associated with statistical extrapolations from the underlying epidemiological evidence of health risks, the Agency might project national average total per-truck benefits less than \$4,000. This suggests that a NO_x-control technology to achieve the estimated HDOH NO_x reductions would need to cost less than about \$4,000 per truck to pass a robust benefit-cost test.

In conducting this scoping analysis, we also noted that ozone benefits-per-ton were much higher for California than the rest of the U.S. We have thus also provided per-truck benefits estimates for California and separately for the Rest of the U.S.[17] In this disaggregated analysis, we estimate that EPA's future analyses might estimate per-truck benefits for trucks operating in California as high as \$13,730 at the least-confident level, and as high as about \$11,680 for a relatively moderate degree of increased confidence (at the 10th percentile exposure cut-off). At the same time, of course, the equivalent benefit-per-truck estimates for Rest of U.S. would be reduced to about \$5,300 (least confidence) and to about \$2,800 (greater confidence).

In all of the above numerical summaries, we rely on the 3% discount rate and the higher end of our $PM_{2.5}$ benefits ranges, which are the combination of assumptions that produces the highest benefits estimates.

[17] The latter estimate is for the average over the 47 other conterminous U.S. states.

Use of a 7% discount rate generally reduces the per-truck benefits by about 25%. Use of the lower $PM_{2.5}$ benefits study (the Krewski *et al*. study) has an even larger effect, though the amount of reduction varies with the confidence level and region of the estimate, as can be discerned from the detailed information provided in Tables 4 and 5. We also note that our analysis has assumed, based on input from EMA, that a 90% reduction in the FTP standard would reduce *in-use* HDOH NO_x emissions by 50%. NERA offers no opinion on what the correct in-use reduction percentage reduction should be, but it is straightforward to make adjustments. For example, if one expects *in-use* emissions to be reduced by the full 90% of the FTP standard's reduction, the benefit-per-truck estimates could increase by about 80%.

Finally, it should be noted that the benefits estimates we report are conservative or, stated differently, weighted to the high side. That conservative approach stems from the fact that in conducting our analyses we have assumed that: there is no exposure threshold to $PM_{2.5}$ or ozone below which mortality effects are no longer evident; it is still appropriate to include benefits associated with ozone-related mortality impacts; the slope of the relative risk function for mortality is linear all the way down to zero exposure; it is appropriate to account for and credit potential health effects benefits at exposure levels below the NAAQS for $PM_{2.5}$ and ozone; the statistical associations observed in the relevant epidemiological studies between exposure to air pollution and mortality effects are sufficient to infer causality, notwithstanding unresolved issues relating to manipulative or interventional causation; and it is appropriate to assess quantified benefits values at the 10th percentile of the exposure levels at issue in the underlying epidemiological studies, as opposed to utilizing a cut-point at the 25th percentile of exposures. Applying different assumptions regarding any of the foregoing points would lead to a reduction in the calculated benefits estimates.

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Exhibit C

On-Road Heavy-Duty Low-NOx Technology Cost Study

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National Renewable Energy Laboratory

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List of Acronyms

ASC	ammonia slip catalyst
CARB	California Air Resources Board
DEF	diesel exhaust fluid
DOC	diesel oxidation catalyst
DPF	diesel particulate filter
EGR	exhaust gas recirculation
EMFAC	EMission FACtor model
EPA	U.S. Environmental Protection Agency
FTP	Federal Test Procedure
FUL	full useful life
g/bhp-hr	grams per brake horsepower-hour
GHG	greenhouse gas
GVWR	gross vehicle weight rating
HD	heavy-duty
HDO	heavy-duty Otto-cycle
HHDD	heavy heavy-duty diesel
hp	horsepower
LHDD	light heavy-duty diesel
LLC	low-load certification
LO-SCR	light-off selective catalytic reduction
MECA	Manufacturers of Emission Controls Association
MHDD	medium heavy-duty diesel
MY	model year

NH_3	ammonia
NO_x	oxides of nitrogen
NREL	National Renewable Energy Laboratory
OBD	on-board diagnostics
OEM	original equipment manufacturer
OOS	out of state
PM	particulate matter
PNA	passive NO_x absorber
R&D	research and development
SCAB	South Coast Air Basin
SCR	selective catalytic reduction
SCRF	selective catalytic reduction on filter
SERA	Scenario Evaluation and Regionalization Analysis
SET-RMC	Supplemental Emission Test with Ramped Mode Cycles
SI	spark ignition
SwRI	Southwest Research Institute
TWC	three-way catalyst

Executive Summary

The National Renewable Energy Laboratory (NREL) conducted a cost analysis for emission control technologies under contract to the California Air Resources Board (CARB). CARB sought incremental cost analysis for emission control technologies for on-road heavy-duty (HD) engines used in vehicles greater than 14,000 pounds (lb) gross vehicle weight rating (GVWR) to achieve oxides of nitrogen (NO_x) emissions rates significantly lower than those required by current emissions standards (CARB 2017). This low-NO_x emission technology cost analysis comprised two main tasks:

- Task 1: An incremental cost analysis for engine and exhaust aftertreatment systems
- Task 2: An engine and exhaust aftertreatment life-cycle cost analysis incorporating incremental upfront costs and operating costs.

The incremental cost analysis included a review of current and under-development engine and exhaust aftertreatment technologies that could achieve 0.02 grams per brake horsepower-hour (g/bhp-hr) NO_x on certification test cycles, including a proposed updated certification test cycle that includes additional low-load operating conditions. Diesel, natural gas, and gasoline HD engine applications were studied. Three diesel technology package combinations of engine and exhaust aftertreatment options were selected based on research in progress at Southwest Research Institute (SwRI), also funded by CARB. The three diesel technology packages were intended to bracket potential cost ranges across two engine displacement levels: ~6–7 liters (L) and ~12–13 L. Representative technology packages for HD natural gas (12 L) and gasoline (6 L) engines were also defined, each with a single displacement level providing a tie point to similar diesel options.

Diesel engines were the primary consideration, as they comprise the majority of HD engines. In addition to studying three diesel technology packages across two engine displacement levels, incremental cost bracketing also included model year (MY) 2023 versus 2027 introduction, U.S. versus California-only implementation, and current full useful life (FUL) versus extended FUL and warranty. Direct and indirect incremental costs were broken down to as discrete a level as possible while maintaining data confidentiality. The calculation of incremental costs was limited by a small number of respondents.

The surveyed original equipment manufacturers (OEMs), Tier 1 suppliers, and trade organizations such as the Manufacturers of Emission Controls Association (MECA) responded with incremental cost, not validation that 0.02 g/bhp-hr emissions levels or specific technology packages are feasible. Engine OEM participation was crucial, as only they could provide estimates for indirect costs that represented a significant portion of the total cost. Incremental costs are largely driven by indirect costs associated with engineering research and development costs and warranty costs. The indirect costs are highly dependent on production volumes over which to amortize research and development costs. Indirect costs due to warranty are high, reflecting high uncertainty with new technology and the introduction timeframes. The incremental costs were not adjusted to reflect a retail markup due to the complexity with which pricing decisions are made.

The average incremental cost for the 6–7-L diesel engines for MY 2023 with current FUL ranged from $3,685 to $5,344, but the absolute low and high bounds were between ~$2,000 and over

$9,000. Extending FUL and warranty moved the average incremental costs to a range of $15,370 to $16,245, with tighter low and high bounds (constrained in part by the limited number of responses). The average incremental cost for the 12–13-L diesel engines for MY 2023 with current FUL ranged from $5,340 to $6,063, but the absolute low and high bounds were between ~$3,000 and over $10,000. Extending FUL and warranty moved the average incremental costs to a range of $28,868 to $47,042, with much wider low and high bounds (driven in part by the limited number of responses). The natural gas 12-L engine application was unable to be studied in detail, but OEM feedback indicated the anticipated incremental cost for natural gas engines and aftertreatment technology is within 10% of the low-cost diesel technology package incremental cost for equivalent displacement, possibly due to requiring a moving average window method to assess emission compliance. The gasoline engine 6-L application was also unable to be studied in detail due to lack of OEM feedback, but comparatively low incremental costs were estimated.

A life-cycle cost analysis was completed to understand the full costs to the owner of the vehicles with a 0.02 g/bhp-hr NO_x technology package outside of the direct upfront vehicle cost increase. The life-cycle cost analysis sought to incorporate costs associated with the following elements: initial incremental purchase cost, fuel consumption changes (changes in fuel economy), diesel exhaust fluid (DEF) consumption changes, and the maximum FUL of the aftertreatment package (major overhaul intervals). Thus, the life-cycle costs depend on the vehicle type (mileage), region, fuel, engine displacement, maximum useful life, fuel economy change, DEF consumption change, and discount rate.

Three scenarios were defined to evaluate the bounds of the life-cycle costs across all parameters evaluated. For the three scenarios evaluated (Low-Cost, Mid-Cost, High-Cost), the life-cycle costs were evaluated for each EMission FACtor (EMFAC) model vehicle type (CARB 2018b), aggregated to a representative average and calculated across the vehicle fleet for the MY 2027 vehicles. The analysis showed that EMFAC vehicles can have significantly different life-cycle costs and that the spread depends on the scenario evaluated: approximately a $4,000 spread across vehicle types in the Low-Cost scenario, while the High-Cost scenario had nearly a $40,000 difference. This large spread was found to be due to the number of aftertreatment package replacements needed throughout the vehicle lifetime. The aggregated, representative average life-cycle costs for the Mid-Cost scenario were estimated to be $12,700 for the 6-L diesel engine, $13,200 for the 12-L diesel engine, $4,800 for the 12-L natural gas engine, and $800 for the 6-L gasoline engine. The total life-cycle costs to California vehicle owners for the MY 2027 vehicles were estimated to range between $92 million and $1.2 billion, depending on the scenario (Low-Cost or High-Cost) realized.

The sensitivity analysis indicated that the manufacturing volume may be the most important parameter impacting the life-cycle cost; however, limited data were received from the external stakeholders surveyed. The next most important parameter was the assumption of extended FUL and extended warranty, as the increase in aftertreatment lifetime may not exceed the vehicle's travel requirement, which results in larger replacement costs over the vehicle's life. However, one may expect that the higher upfront purchase incurred by the vehicle owner should effectively be offset by the repair savings over the lifetime of the vehicle. Next, the aftertreatment cost bound (low/high error bars on the incremental cost data), fuel economy improvement, and discount rate were found to have a moderate impact on the life-cycle cost. Lastly, the region and DEF consumption change were found to have minimal influence on the life-cycle cost.

The results of this cost analysis reflect the specific technology and aftertreatment FUL assumptions on which the study was based. In particular, the incremental cost of moving from a 0.2g/bhp-hr to 0.02 g/bhp-hr standard is expected to be non-linear due to diminishing returns on technology performance. Extrapolating the results beyond this specific study and outside of these specific assumptions is not recommended and should only be done with careful attention to the scope and limits of this study.

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Abstract

The National Renewable Energy Laboratory (NREL) conducted a cost analysis for emission control technologies under contract to the California Air Resources Board (CARB). CARB sought incremental cost analysis for emission control technologies for on-road heavy-duty (HD) engines used in vehicles greater than 14,000 pounds (lb) gross vehicle weight rating (GVWR) to achieve oxides of nitrogen (NO_x) emissions rates significantly lower than those required by current emissions standards. Specifically, incremental costs (without any retail price markup) were estimated for representative diesel, natural gas, and gasoline engine and emission aftertreatment systems that were selected to represent potential technology packages that could achieve 0.02 grams per brake horsepower-hour (g/bhp-hr) NO_x on certification test cycles, including a proposed updated certification test cycle that includes additional low-load operating conditions. NREL surveyed stakeholders including industry association groups, Tier 1 suppliers, and engine original equipment manufacturers (OEMs) to estimate incremental direct and indirect costs. Incremental costs were considered for current engine full useful life (FUL) definitions, as well as with proposed increased FUL and warranty periods. The incremental costs were subsequently incorporated in life-cycle cost analyses examining the incremental engine and aftertreatment costs along with life-cycle costs over the various engine FUL scenarios. Life-cycle costs analysis included the incremental upfront cost, fuel consumption changes (changes in fuel economy), diesel exhaust fluid (DEF) consumption changes, and the maximum FUL of the aftertreatment package (major overhaul intervals).

Project Background and Objective

Current emission standards for heavy-duty diesel engines, established by the United States Environmental Protection Agency (EPA) for 2010, specify a limit of 0.20 grams per brake horsepower-hour (g/bhp-hr) NO_x. This standard represents a 90% reduction from the previous benchmark of 2.0 g/bhp-hr and applies to both heavy-duty diesel engines and heavy-duty Otto-cycle engines used in vehicles greater than 14,000-lb GVWR.

Diesel-engine manufacturers utilize a variety of technologies in order to meet these standards, primarily among them being selective catalytic reduction (SCR). Natural-gas engine manufacturers use SCR for lean-burn engines and three-way catalysts (TWCs) for stoichiometric engines. Both of these methods reduce NO_x emissions by removing them from the engine-out exhaust prior to exiting the tailpipe. These manufacturers have used lessons learned from other applications such as stationary-source and light-duty vehicles to meet current NO_x emission requirements, and as these technologies mature there are opportunities to reduce emissions even further.

The California Air Resources Board (CARB), together with the Southwest Research Institute (SwRI), is currently funding several research programs to investigate the feasibility of achieving NO_x emissions less than the 2010 limit of 0.20 g/bhp-hr. The first ("Stage 1") project is a $1.6 million research contract between CARB and SwRI to evaluate improved engine emission control calibration, enhanced aftertreatment technologies and configurations, improved aftertreatment thermal management, urea dosing strategies, and engine management practices for two heavy-duty engines: one natural-gas engine with a TWC and one diesel engine with a diesel particulate filter (DPF) and SCR. The target emission rate for this project, which was finalized in December 2016, is 0.02 g/bhp-hr NO_x.

CARB is also contracting a $1.05 million "Stage 2" project with SwRI to further optimize the diesel engine aftertreatment system for low engine-load duty cycles typical of city driving. Stage 2 objectives are to develop a supplemental low-load certification test cycle that will, along with the Federal Test Procedure (FTP), ensure NO_x control under nearly all driving conditions and evaluate metrics for in-use testing under low-load operations. The "Stage 3" project, currently in the planning stage, will complement the Stage 1 and Stage 2 efforts with testing on an additional engine that is representative of likely future engine configurations.

Alongside current emission standards, CARB and EPA both require that heavy-duty engines meet these standards throughout their entire useful life. The useful life period is defined according to a vehicle's GVWR, and for heavy-duty engines ranges from 110,000–435,000 miles. The useful life period for Otto-cycle and light heavy-duty diesel engines (14,001–19,500-lb GVWR) is 110,000 miles/10 years; for medium heavy-duty diesel engines (19,501–33,000-lb GVWR) 185,000 miles/10 years; and for heavy heavy-duty diesel engines (greater than 33,000-lb GVWR) 435,000 miles/10 years, or 22,000 hours.

Well-maintained on-road diesel engines can operate significantly beyond their currently defined useful life periods (e.g., many heavy-duty diesel engines currently operate upwards of 800,000 miles to over a million miles), and CARB is taking this reality into consideration as it evaluates the consequences of lowering its NO_x emission targets. Engine durability becomes a critical factor with longer useful life definitions, particularly in preventing "upstream" engine component failures that can damage "downstream" emission control system components and cause excess emissions of criteria pollutants such as particulate matter (PM) and NO_x. Therefore, manufacturers will need to improve the durability of their engines and emission control systems by developing higher-quality parts and assembly methods and replacement of components and/or subsystems.

CARB is expected to propose new standards to be implemented by 2024, which will set even lower NO_x emission standards and add new certification test cycles to ensure emission control at low-load operations. Adding this new test cycle to the certification requirement is expected to drive further improvements to aftertreatment hardware and engine control and calibration.

With these new emission standards of approximately 0.02 g/bhp-hr NO_x in mind, it is important to examine the direct and indirect costs of implementing new technologies, both the incremental costs to original equipment manufacturers and the costs of using the technology packages throughout the engines' useful life. These costs can be divided by category, including the specific technologies for achieving the NO_x standard, the costs to increase durability (extended useful life), and the costs of the on-board diagnostics (OBD) hardware and calibration works impacted by the changes. This cost analysis will use specific emission control and engine technologies identified by SwRI in Stages 1 and 2, along with testing that is representative of likely future engine configurations.

Project Summary

This project was defined by two tasks—Task 1: Engine Incremental Cost Analysis and Task 2: Engine Life-Cycle Costs. For Task 1, NREL reviewed current technologies and technology packages that are being examined as part of the SwRI projects, Stages 2 and 3, as provided by CARB. NREL identified and reviewed likely emission control and engine technologies to meet 0.02 g/bhp-hr NO_x requirements with CARB staff based on Stage 2 and 3 efforts from SwRI testing of potential future engine configurations. These technologies were then defined as the potential technologies and the starting point of developing a low-NO_x technology incremental cost analysis from 2018 baseline costs.

NREL then evaluated these potential technologies and technology packages for engine plus aftertreatment incremental cost analysis via a series of surveys sent to Tier 1 suppliers, trade organizations, and engine OEMs. The surveys defined the potential technologies broken into engine components, emission control components, subsystems, and indirect costs. The combination of incremental costs (over the 2018 baseline) associated with developing and integrating the specified lower NO_x emission control technologies into the engines, the costs of increasing the durability of these engines and their emission control systems, and the costs of directly impacted OBD hardware and calibration works of these specified technology packages were then examined to understand the total incremental cost implications to Tier 1 suppliers and engine OEMs of the potential technologies.

The evaluation of costs was dependent on cooperation from Tier 1 suppliers, trade organizations and engine OEMs, as well as the availability of direct and indirect cost information for engine and emission control technologies. NREL utilized existing relationships with industry partners in order to perform a thorough cost assessment but could not guarantee full cooperation or sharing of confidential cost information from Tier 1 suppliers, trade organizations, and engine OEMs.

After accounting for the initial incremental cost implications to Tier 1 suppliers (both collectively through the Manufacturers of Emission Controls Association [MECA] and individually) and engine OEMs, NREL conducted a life-cycle cost analysis as Task 2 to examine the costs of using the specified technology packages during the engines' certification full useful life (FUL). NREL utilized a range of FUL values for each heavy-duty vehicle category, Classes 4 through 8. The current FUL mileage—for heavy-duty engines of 110,000 miles up to 435,000 miles, depending on a vehicle's GVWR; 110,000 miles/10 years for heavy-duty Otto-cycle (HDO) and light heavy-duty diesel (LHDD) engines (14,001–19,500-lb GVWR); 185,000 miles/10 years for medium heavy-duty diesel (MHDD) engines (19,501–33,000-lb GVWR); and 435,000 miles/10 years or 22,000 hours for heavy heavy-duty diesel (HHDD) engines (greater than 33,000-lb GVWR)—was defined as the low-end value of the range for each specific vehicle class. For the high-end value of the range, NREL utilized input from CARB for proposed extended FUL targets as the upper-bound levels for each specific vehicle class: 250,000 miles/15 years for HDO engines (14,001–19,500-lb GVWR), 550,000 miles/15 years for LHDD engines (14,001–19,500-lb GVWR) and MHDD engines (14,001–19,500-lb GVWR), and 1,000,000 miles/15 years for HHDD engines (greater than 33,000-lb GVWR). Additionally, per CARB's guidance, the high-end value with extended FUL also includes the provision that warranty periods will increase to 80% of the extended FUL, both in mileage and time, except for heavy-duty Otto-cycle, which was specified as 220,000 miles/12 years. The current FUL defining the lower bound and the extended FUL defining the upper bound are summarized in Table 1.

Table 1. Current and Proposed Extended Full Useful Life and Warranty for Engine Life-Cycle Cost Analysis

	LHDD	MHDD	HHDD	Natural Gas – Otto	Heavy-Duty – Otto
GVWR (lb)	14,001–19,500	19,501–33,000	>33,000	>33,000	14,000
Current full useful life	110,000 miles/10 years	185,000 miles/10 years	435,000 miles/10 years, 22,000 hours	435,000 miles/10 years, 22,000 hours	110,000 miles/15 years
Proposed extended full useful life	550,000 miles/15 years	550,000 miles/15 years	1,000,000 miles/15 years	1,000,000 miles/15 years	250,000 miles/15 years
Proposed warranty period with extended full useful life	440,000 miles/12 years	440,000 miles/12 years	800,000 miles/12 years	800,000 miles/12 years	220,000 miles/12 years

After accounting for the initial incremental costs of the technologies, as determined in Task 1, the life-cycle cost assessment of Task 2 then took into account the aftertreatment technologies' effects on fuel consumption, DEF consumption, major overhaul intervals (full useful life estimates), manufacturing volume, and financial discount rates. The life-cycle cost modeled for each vehicle is specific to the EMission FACtor (EMFAC) model's vehicle definition of vehicle miles traveled, which depends on the specific region, vocation, model year, fuel type, and age.

For the life-cycle cost analysis in Task 2, the aftertreatment full useful life mileage was used to set the equipment overhaul schedule. For all scenarios in the life-cycle cost analysis, the incremental cost associated with the aftertreatment package was assumed to be incurred after the truck mileage exceeded the stated maximum FUL. This assumption is expected to be conservative, as not all aftertreatment packages will fail immediately after they exceed their stated maximum FUL and statistical analysis of failure rates combined with data on aftertreatment technology operating and maintenance costs were not available. To understand the impact of this assumption on the life-cycle cost, a sensitivity analysis was completed assuming the aftertreatment package would not need to be replaced over the vehicle's lifetime, as that provides the lower bound on the life-cycle cost.

1. Task 1: Engine Incremental Cost Analysis

1.1 Representative Engine Platform Approach

The engine and aftertreatment incremental cost analysis began with a review of 54 model year (MY) 2018 medium- and heavy-duty engine family CARB certification summaries, covering Class 4–8 vehicle applications. The review provided background on the fuels used, range of engine displacements for each service class (i.e., LHDD, MHDD, HHDD, HDO), current technologies utilized, and certification levels versus Federal Test Procedure (FTP) and heavy-duty Supplemental Emissions Test with Ramped Mode Cycles (SET-RMC) standards for NO_x. Because the majority of Class 4–8 engines are diesel fueled, incremental costs for diesel engines was the primary focus of the study. Natural gas and gasoline were also studied, but liquified petroleum gas/propane was not. A limited number of engine platforms were initially selected to represent the Class 4–8 vehicle population, based on engine displacement. This down-selection was necessary to come up with a reasonable number of representative engine platforms to use for the incremental cost analysis that could subsequently be used in the Task 2 life-cycle cost analysis over large vehicle populations, while keeping manageable the burden of calculating incremental cost for surveys conducted with Tier 1 suppliers, trade organizations, and engine OEMs. The initial engine platforms included: 6-L LHDD, 9-L MHDD, 12-L HHDD, 15-L HHDD, 12-L natural gas, and 6-L HDO (gasoline). Initial reviews with industry provided feedback that this number of engine platforms was still too large, and the diesel engine platforms could be consolidated and referenced to approximate horsepower levels. As a result, the diesel engine platforms were reduced to ~6–7 L with ~300 horsepower (hp) and ~12–13 L with ~475 hp. This reduction would still provide incremental costs with appropriate discrete levels. The in-between calculation for a 9-L engine was agreed to not be worth the additional burden for industry survey responses. The elimination of the 15-L engine was agreed to be covered by increased power density from ~12–13-L engines with future trends.

Current technologies were reviewed to benchmark the baseline for the 0.02 g/bhp-hr NO_x incremental cost. The industry surveys were designed to collect direct and indirect cost information for engine and aftertreatment subsystems from a 2018 baseline, with a 0.20 g/bhp-hr standard, as well as multiple technology packages assumed to meet a potential future 0.02 g/bhp-hr NO_x standard under a proposed new low-load certification (LLC), in addition to FTP and SET-RMC. The incremental costs would form the basis of Task 1. While the surveys were designed to allow industry respondents to start with their own 2018 baseline and did not explicitly define a common set of identical technologies, the CARB certification review showed most diesel engines in the 6–7-L and 12–13-L ranges were common in having direct diesel injection, cooled exhaust gas recirculation (EGR), turbocharging, a diesel oxidation catalyst (DOC), a diesel particulate filter (DPF), and selective catalytic reduction (SCR) using DEF. The technology packages supporting 0.02 g/bhp-hr NO_x selected for incremental cost study are described in more detail below.

A single natural-gas engine platform was selected at 12 L to align with the ~12–13-L diesel platform. The CARB certification review showed a number of natural-gas engines (in various displacements, meeting MHDD and HHDD requirements) sharing the same technologies: stoichiometric Otto-cycle operation, spark ignition (SI), throttle body fuel injection, turbocharging, cooled EGR, and a three-way catalyst (TWC).

A single gasoline-fueled HDO platform was selected at 6 L to align with the ~6–7-L diesel platform. The CARB certification review showed HDO gasoline is approaching 0.02 g/bhp-hr NO_x on the current certification cycles using stoichiometric, SI, naturally aspirated, EGR technologies with a TWC technology package.

Utilizing the results and recommendations from Stage 2 and 3 efforts from SwRI testing of potential future diesel-engine configurations, NREL identified three diesel technology packages to evaluate the total incremental cost implications for an MY 2023 release nationwide. These identified diesel technology packages were intended to represent potential low-, average-, and high-cost options to meet a 0.02 g/bhp-hr NO_x standard and were meant to provide a broader assessment of potential incremental costs than a single option. As previously referenced, no natural-gas technology package was surveyed for incremental costs related to 0.02 g/bhp-hr NO_x, and the HDO gasoline technology package only included TWC and calibration upgrades. The resulting engine platforms defined for the incremental cost study are summarized in Table 2.

Table 2. Engine Platform Analysis for Incremental Cost Analysis

	LHDD	HHDD	Natural Gas – HHDD standard	Gasoline – HDO
Engines	~6–7 L ~300 hp	~12–13 L ~475 hp	12 L	6 L
Current full useful life	110,000 miles/10 years	435,000 miles/10 years, 22,000 hours	435,000 miles/10 years, 22,000 hours	110,000 miles/10 years
Low-Cost Tech.	$$$	$$$	Not applicable	Not applicable
Avg.-Cost Tech.	$$$	$$$	Not applicable	$$$
High-Cost Tech.	$$$	$$$	Not applicable	Not applicable

NREL then directly surveyed heavy-duty engine OEMs, Tier 1 suppliers, emission control technology manufacturers, and industry trade organizations to obtain the most accurate and current cost information for the identified likely technology packages to meet 0.02 g/bhp-hr NO_x requirements and the cost implications for using these specific technologies. The cost survey included a definition of the potential technologies as engine components, emission control components, subsystems and strategies, and indirect costs broken into categories of research and development (R&D) costs, certification costs, and warranty costs. The combination of costs associated with developing and integrating the specified lower NO_x emission control technologies into the engines, the costs of increasing the durability of these engines and their emission control systems, and the costs of impacted OBD hardware and calibration of these specified technology package were then examined to understand the total incremental cost implications to Tier 1 suppliers and engine OEMs of the potential technologies in two different surveys. Any incremental costs associated with future OBD requirements unrelated to meeting 0.02 g/bhp-hr NO_x were excluded from this study. Similarly, incremental costs related to future greenhouse gas (GHG) or fuel efficiency requirements and not specifically to meeting 0.02 g/bhp-hr NO_x were also excluded.

The first survey assumed that the 0.02 g/bhp-hr NO_x regulation beginning MY 2023 included current FTP and SET-RMC steady-state test cycles, as well as a proposed new LLC for medium- and heavy-duty engine system certification. While not finalized and currently the topic of ongoing research, the new LLC engine cycle was assumed to last approximately 90 minutes, including a combination of motoring, sustained low load, and high-power transients. This first survey considered FUL hours/miles to remain the same as the current regulation. The survey was designed to allow industry respondents to start with their own 2018 baseline and did not explicitly define a common set of identical technologies. As a reference point, NREL provided internally generated estimates (from research, literature review, and engineering judgement) for the 2018 current technology costs (Posada, Chambliss, and Blumberg 2016; Posada Sanchez, Bandivadekar, and German 2012; Ou et al. 2019). Direct costs for both a 2018 baseline and 0.02 g/bhp-hr technology packages were surveyed on discrete engine and aftertreatment subsystem levels, along with indirect costs. The level of discrete subsystems was kept as small as possible to provide insight for where the costs accumulate while also being kept large enough to prevent identification of proprietary or confidential cost information from an individual respondent. Furthermore, only incremental costs are reported in this report and preliminary reviews with CARB to prevent identifying proprietary or confidential 2018 baseline costs. The survey requested future costs be calculated in 2018 dollars. The first survey asked for production volumes to be identified and to provide guidance on cost impacts for 0.02 g/bhp-hr incremental costs if regulation were to include all of the United States or California only.

The second survey was a follow-up survey sent to those Tier 1 suppliers, trade organization, and engine OEMs that responded to the first survey. The technology packages remained the same as the first survey, but instead assumed 0.02 g/bhp-hr NO_x regulation beginning MY 2027 and again included current FTP and SET-RMC steady-state test cycles, as well as a new LLC. This second survey also considered extended useful life hours/miles as proposed by CARB in Table 1. The second survey asked for costing information to consider 0.02 g/bhp-hr regulation if only California were included, representing lower production volumes than a scenario where all of the U.S. were included.

NREL then aggregated all of the data from the cost survey responses and the initial estimates derived by NREL from research, literature review, and engineering judgement. The incremental costs were not adjusted to reflect a retail markup due to the complexity with which pricing decisions are made. In responding to NREL's surveys, trade organizations, Tier 1 suppliers, and OEMs did provide feedback that they did not agree or conclude that these technologies would be feasible for meeting the 0.02 g/bhp-hr NO_x requirements by MY 2023. Their valuable input was strictly a costing exercise and not a technology feasibility assessment. The diesel incremental cost information resulted in a range of costs due to the format of the provided data from the responses received. This range consisted of a low, average, and high estimate for engine technology costs, aftertreatment technology costs, OBD-related direct costs, and indirect costs. The survey results for the diesel engine and aftertreatment technology packages were then defined as three total incremental costs of low, average, and high estimates based on the identified potential technology packages to achieve 0.02 g/bhp-hr NO_x requirements.

Fewer responses were received for the natural gas (HHDD standard) engine platform, preventing NREL from sufficiently aggregating incremental cost information to protect proprietary information. Therefore, NREL reported the total integrated incremental cost as an order of magnitude in comparison to the diesel engine with similar displacement results; the subsystem-level engine, aftertreatment, and OBD system direct costs as well as the indirect costs were not broken out or reported.

Similarly, few responses were received for the gasoline HDO engine platform. Some aggregation was possible for direct costs, but only NREL estimates were available for indirect costs. As a result, only total integrated incremental costs are reported.

1.2 Identifying Potential Diesel Technologies to Achieve 0.02 g/bhp-hr NO_x

CARB is currently funding several research programs with SwRI to investigate the feasibility of achieving 0.02 g/bhp-hr NO_x emissions with a diesel engine and is in the Stage 3 process of testing specific emission control and diesel engine technologies. Based on SwRI's research and results from Stages 1 and 2 (Sharp et al., "Thermal Management," 2017; Sharp et al., "Comparison of Advanced," 2017; Sharp et al., "NO_x Management," 2017), NREL identified different engine and emission control technologies that showed potential capabilities of achieving 0.02 g/bhp-hr NO_x emissions during current FTP and SET-RMC steady-state test cycles, as well as a proposed new LLC cycle by MY 2023. These diesel engine and emission control technologies were grouped into three different diesel technology packages to represent a range of potential low-, average-, and high-costing diesel technology package solutions.

The potential low-cost diesel technology package consisted of an EPA 2017 certification-compliant engine with a variable-geometry turbo charger, no turbo compounding, and a combined engine thermal management strategy of EGR cooler bypass, charge air cooler bypass, and a turbine bypass. In addition to the engine system, the emission control technologies included two points of DEF dosing and DEF mixers, one light-off SCR (LO-SCR), one DOC, one DPF, two SCRs, and one ammonia slip catalyst (ASC). The aftertreatment system also contained a NO_x sensor upstream of the first DEF dosing system and mixer, a temperature sensor upstream of the LO-SCR, a second temperature sensor downstream of the LO-SCR, a second NO_x sensor downstream LO-SCR and upstream of the DOC, a third temperature sensor downstream of the LO-SCR and upstream of the DOC, a fourth temperature sensor downstream of the DOC and upstream of the DPF, a fifth temperature sensor downstream of the DPF and upstream of the first second DEF dosing system and mixer, an ammonia (NH_3) sensor downstream the first SCR and upstream the second SCR, a sixth temperature sensor downstream of the ASC, and a third NO_x sensor downstream of the ASC. An example of the aftertreatment technology system with sensors is illustrated in Figure 1.

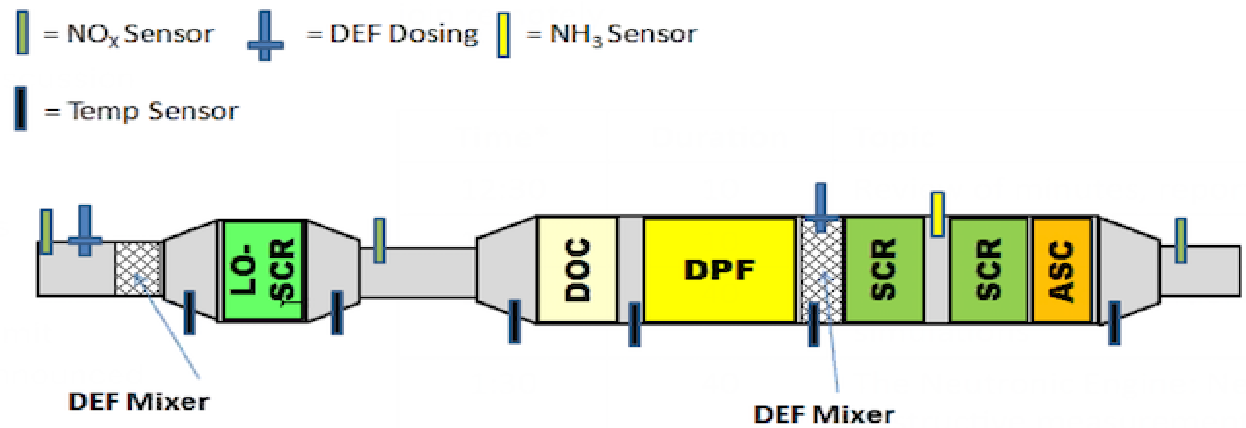

Figure 1. Schematic of proposed low- and average-cost diesel aftertreatment technology
Figure from SwRI

The potential average-cost diesel technology package consisted of an EPA 2017 certification-compliant engine with a variable-geometry turbo charger, no turbo compounding, and an engine thermal management strategy and technology for cylinder deactivation. In addition to the engine system, the emission control technologies again included the same aftertreatment system as the low-cost diesel technology package with two points of DEF dosing and DEF mixers, one LO-SCR, one DOC, one DPF, two SCRs, and one ASC, as shown in Figure 1. The aftertreatment system also contained a NO_x sensor upstream of the first DEF dosing system and mixer, a temperature sensor upstream of the LO-SCR, a second temperature sensor downstream of the LO-SCR, a second NO_x sensor downstream LO-SCR and upstream of the DOC, a third temperature sensor downstream of the LO-SCR and upstream of the DOC, a fourth temperature sensor downstream of the DOC and upstream of the DPF, a fifth temperature sensor downstream of the DPF and upstream of the first second DEF dosing system and mixer, an NH_3 sensor downstream of the first SCR and upstream of the second SCR, a sixth temperature sensor downstream of the ASC, and a third NO_x sensor downstream of the ASC.

The proposed high-cost diesel technology package consisted of an EPA 2017 certification-compliant engine with a variable-geometry turbo charger, no turbo compounding, and a combined engine thermal management strategy of EGR cooler bypass, charge air cooler bypass, and a turbine bypass. In addition to the engine system, the emission control technologies included a passive NO_x absorber (PNA), one DOC, one DEF doser and DEF mixer, one selective catalytic reduction on filter (SCRF), one SCR, and one ASC. The aftertreatment system also contained a NO_x sensor upstream of the PNA, a second NO_x sensor downstream of the PNA, an NH_3 sensor downstream of the SCRF and upstream of the SCR, and a third NO_x sensor downstream of the ASC. An example of the aftertreatment technology is illustrated in Figure 2.

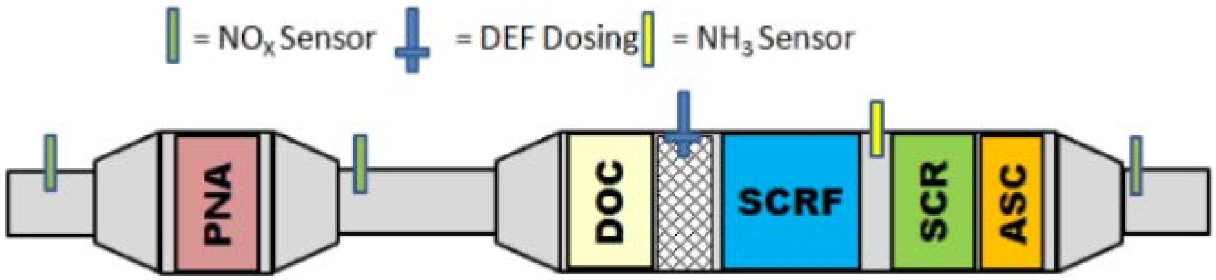

Figure 2. Schematic of proposed high-cost diesel aftertreatment technology

Figure from SwRI

Note that the proposed technology packages that were initially designed to represent low-, average-, and high-cost combinations. It was assumed that the PNA, as a very new technology, would drive incremental costs to be higher than other packages. Likewise, cylinder deactivation was assumed to have a higher incremental cost than cooler bypasses for charge air, EGR, and turbine given the same aftertreatment package. However, once incremental cost information became available, the relative incremental costs did not necessarily turn out in that order. Nevertheless, to maintain consistency in the study, the proposed technology packages continued to be referred by their initial naming convention.

1.3 Identifying Potential Gasoline and Natural Gas Technologies to Achieve 0.02 g/bhp-hr NO_x

The single natural-gas 12-L engine platform was selected to align with the ~12–13-L diesel platform. The CARB certification review showed a number of natural-gas engines (in various displacements, meeting MHDD and HHDD requirements) sharing the same technologies: stoichiometric Otto-cycle operation, SI, throttle body fuel injection, turbocharging, cooled EGR, and a TWC. Notably, most of the natural-gas engines already meet CARB's optional low-NO_x standard at 0.02 g/bhp-hr under the current certification cycles. Because the proposed LLC certification was assessed to be less challenging for a stoichiometric SI engine than a diesel engine, it was assumed that the current 2018 "baseline" technology package would already meet the new 0.02 g/bhp-hr NO_x requirement. Incremental cost for 0.02 g/bhp-hr NO_x was therefore not calculated, but cost increases related to extending FUL were considered. As noted later in this report, industry feedback identified this assumption as incorrect.

The single gasoline-fueled HDO platform was selected at 6 L to align with the ~6–7-L diesel platform. The CARB certification review showed HDO gasoline is approaching 0.02 g/bhp-hr NO_x on the current certification cycles, and similar technology (stoichiometric, SI, naturally aspirated, EGR technologies with a TWC) with liquified petroleum gas fuel has recently been certified at 0.05 g/bhp-hr and 0.02 g/bhp-hr under CARB's optional low-NO_x standards. The base engine was assumed to need no significant upgrades for the 0.02 g/bhp-hr standard with proposed LLC certification cost study, but TWC direct cost upgrades and indirect costs for engineering, certification, and warranty were surveyed, as well as extended FUL impacts. Vehicle packaging impacts were noted to also potentially be required to enable close coupling of the TWCs.

1.4 NREL Survey of Potential Technologies to Achieve 0.02 g/bhp-hr NO_x

NREL created a cost survey with a baseline price of an MY 2018 system representing an EPA 2018 certification-compliant engine and aftertreatment system in 2018 dollars and asked trade organizations, Tier 1 suppliers, and engine OEMs to provide incremental cost estimates in comparison to the above-defined technologies with the potential to achieve 0.02 g/bhp-hr NO_x requirements. The cost survey was reviewed with CARB and EPA staff and approved by CARB before submitting for requested responses. The survey consisted of two technology packages for diesel engine and aftertreatment systems, one technology package for natural-gas engines and aftertreatment, and one technology package for gasoline engines and aftertreatment systems. To simplify the survey for stakeholder input and avoid asking for input on three separate combinations of engine and aftertreatment technology packages, the two unique diesel engine technology packages (charge air, EGR, and turbine cooler bypass vs. cylinder deactivation) were surveyed with the two unique aftertreatment technology packages (Figure 1 and Figure 2). From these incremental cost inputs, NREL could construct the proposed low-, average-, and high-cost combined engine and aftertreatment technology packages.

The first survey assumed that the 0.02 g/bhp-hr NO_x regulation beginning MY 2023 included current FTP and SET-RMC steady-state test cycles, as well as a new LLC cycle. While not finalized and currently the topic of ongoing research, the LLC was assumed as a new engine certification cycle lasting approximately 90 minutes and included a combination of motoring, sustained low load, and high-power transients. This first survey also considered FUL hours/miles to remain the same as the current regulation. NREL also prefaced the likely follow-up survey seeking additional guidance on how increasing FUL hour/mile requirements may further affect the provided costs.

The second survey was a follow-up survey sent to the same Tier 1 suppliers, trade organizations, and engine OEMs that responded to the first survey. The technology packages remained the same and instead assumed 0.02 g/bhp-hr NO_x regulation beginning MY 2027 and again included current FTP and SET-RMC steady-state test cycles, as well as a proposed new LLC cycle. Again, while not finalized and currently the topic of ongoing research, the LLC was assumed as a new engine certification cycle lasting approximately 90 minutes and included a combination of motoring, sustained low load, and high-power transients. This second survey considered extended FUL hours/miles as proposed by CARB's Stage 2 definitions defined in Table 1. Additionally, per CARB's guidance, the extended FUL also included the assumption that warranty periods will increase to 80% of the extended FUL, both in mileage and time, except for heavy-duty Otto cycle, which was specified as 220,000 miles/12 years.

1.4.1 Definition of Baseline Costs of Current Technologies With 2018 EPA Certification

As a starting point for the incremental cost definition of potential technologies to meet 0.02 g/bhp-hr NO_x requirements, NREL estimated the direct manufacturing costs and indirect costs for an EPA 2018-certified engine and aftertreatment system production costs of current technology to meet 0.20 g/bhp-hr NO_x in 2018 dollars for the U.S. market based on literature reviews and engineering judgement (Posada, Chambliss, and Blumberg, 2016; Posada Sanchez, Bandivadekar, and German 2012; Ou 2019). These estimates were defined for two diesel platforms, 6–7 L and 12–13 L, based on the majority of current market offerings. NREL then estimated the incremental cost of MY 2023 technologies to meet a 0.02 g/bhp-hr NO_x requirement based on literature review, engineering judgement, and feedback from SwRI to provide a baseline estimate of the incremental costs for the two potential diesel technology packages for each of the two engine platforms. The NREL estimates for EPA 2018-certified (0.20 g/bhp-hr NO_x) engine and aftertreatment direct and indirect costs, as well as NREL estimates for incremental direct and indirect costs for MY 2023 0.02 g/bhp-hr NO_x were generated as starting points for stakeholders to consider in the survey. NREL requested survey responses to utilize the baseline estimates, if accurate, or to correct NREL's incremental cost estimates as necessary. Only incremental costs are revealed in this report.

The baseline technology packages for the diesel engine and aftertreatment technology consisted of an EPA 2018-certified engine, a DOC, a DPF, a DEF dosing system and mixer (with a single doser), am SCR with ASC, one NO_x sensor, three NH_3 sensors, and four temperature sensors. These components were the same for the two platforms of 6–7 L and 12–13 L. The baseline costs and resulting incremental costs were scaled accordingly. The baseline technology package for the gasoline HDO engine platform consisted of stoichiometric, SI, naturally aspirated, EGR technologies with a TWC. The baseline technology package for the natural-gas system consisted of stoichiometric Otto-cycle operation, SI, throttle body fuel injection, turbocharging, cooled EGR, and a TWC.

1.4.2 NREL Initial Incremental Cost Estimates

NREL’s initial estimated incremental costs of the potential diesel technology package likely to be the lowest incremental cost to meet 0.02 g/bhp-hr NO_x for the 6–7-L platform are depicted in Table 3. This technology package consisted of an EPA 2017 certification-compliant engine with a variable-geometry turbo charger, no turbo compounding, and a combined engine thermal management strategy of EGR cooler bypass, charge air cooler bypass, and a turbine bypass. In addition to the engine system, the emission control technologies included two points of DEF dosing and DEF mixers, one LO-SCR, one DOC, one DPF, two SCRs, and one ASC. In the following tables, note that negative incremental costs mean the cost for that component/subsystem reduce from the 2018 baseline.

Table 3. NREL Estimates of Potential Low-Cost Diesel Technology Package 6–7 L

Cost Component	Incremental Cost Estimate
EGR Cooler Bypass	$330
Charge Air Cooler Bypass	$200
Turbine Bypass	$220
Total Engine Technology Incremental Cost	**$750**
LO-SCR	$530
DOC	($15)
DPF	($45)
SCR+ASC and DEF Dosing System	$751
OBD Sensors and Controllers (NO_x, NH_3, and Temp Sensors)	($66)
Total Aftertreatment Technology Incremental Cost	**$1,155**
R&D Engineering Incremental Cost	$100
Certification Incremental Costs	$0
Warranty Incremental Costs	$0
Total Indirect Incremental Costs to Manufacturer	**$100**
Total Incremental Cost Comparison	**$2,005**

NREL's initial estimated incremental costs of the potential diesel technology package, likely to be the lowest incremental cost to meet 0.02 g/bhp-hr NO_x for the 12–13-L platform, are depicted in Table 4.

Table 4. NREL Estimates of Potential Low-Cost Diesel Technology Package 12–13 L

Cost Component	Incremental Cost Estimate
EGR Cooler Bypass	$330
Charge Air Cooler Bypass	$200
Turbine Bypass	$220
Total Engine Technology Incremental Cost	**$750**
LO-SCR	$750
DOC	$504
DPF	($98)
SCR+ASC and DEF Dosing System	$1,277
OBD Sensors and Controllers (NO_x, NH_3, and Temp Sensors)	($66)
Total Aftertreatment Technology Incremental Cost	**$2,367**
R&D Engineering Incremental Cost	$100
Certification Incremental Costs	$0
Warranty Incremental Costs	$0
Total Indirect Incremental Costs to Manufacturer	**$100**
Total Incremental Cost Comparison	**$3,217**

NREL's initial estimated incremental costs of the potential diesel technology package, likely to be an average of incremental cost to meet 0.02 g/bhp-hr NO_x for the 6–7-L platform, are depicted in Table 5. The potential average-cost diesel technology package consisted of an EPA 2017 certification-compliant engine with a variable-geometry turbo charger, no turbo compounding, and an engine thermal management strategy and technology for cylinder deactivation. In addition to the engine system, the emission control technologies again included the same aftertreatment system as the low-cost diesel technology package with two points of DEF dosing and DEF mixers, one LO-SCR, one DOC, one DPF, two SCRs, and one ASC.

Table 5. NREL Estimate of Potential Average-Cost Diesel Technology Package 6–7 L

Cost Component	Incremental Cost Estimate
Cylinder Deactivation	$1,050
Total Engine Technology Incremental Cost	**$1,050**
LO-SCR	$530
DOC	($15)
DPF	($45)
SCR+ASC and DEF Dosing System	$751
OBD Sensors and Controllers (NO_x, NH_3, and Temp Sensors)	($66)
Total Aftertreatment Technology Incremental Cost	**$1,155**
R&D Engineering Incremental Cost	$100
Certification Incremental Costs	$0
Warranty Incremental Costs	$0
Total Indirect Incremental Costs to Manufacturer	**$100**
Total Incremental Cost Comparison	**$2,305**

NREL's initial estimated incremental costs of the potential diesel technology package, likely to be the average incremental cost to meet 0.02 g/bhp-hr NO_x for the 12–13-L platform, are depicted in Table 6.

Table 6. NREL Estimates of Potential Average-Cost Diesel Technology Package 12–13 L

Cost Component	Incremental Cost Estimate
Cylinder Deactivation	$1,050
Total Engine Technology Incremental Cost	**$1,050**
LO-SCR	$750
DOC	$504
DPF	$98
SCR+ASC and DEF Dosing System	$1,277
OBD Sensors and Controllers (NO_x, NH_3, and Temp Sensors)	($66)
Total Aftertreatment Technology Incremental Cost	**$2,563**
R&D Engineering Incremental Cost	$100
Certification Incremental Costs	$0
Warranty Incremental Costs	$0
Total Indirect Incremental Costs to Manufacturer	**$100**
Total Incremental Cost Comparison	**$3,713**

NREL's initial estimated incremental costs of the potential diesel technology package, likely to be the highest incremental cost to meet 0.02 g/bhp-hr NO_x for the 6–7-L platform, are depicted in Table 7. The potential high-cost diesel technology package consisted of an EPA 2017 certification-compliant engine with a variable-geometry turbo charger, no turbo compounding, and a combined engine thermal management strategy of EGR cooler bypass, charge air cooler bypass, and a turbine bypass. In addition to the engine system, the emission control technologies included a PNA, one DOC, one DEF doser and DEF mixer, one SCRF, one SCR, and one ASC.

Table 7. NREL Estimates of Potential High-Cost Diesel Technology Package 6–7 L

Cost Component	Incremental Cost Estimate
EGR Cooler Bypass	$330
Charge Air Cooler Bypass	$200
Turbine Bypass	$220
Total Engine Technology Incremental Cost	**$750**
PNA	$730
DOC	($15)
DPF (2018 baseline system only)	($759)
SCRF	$714
SCR+ASC and DEF Dosing System	$74
OBD Sensors and Controllers (NO_x, NH_3, and Temp Sensors)	$314
Total Aftertreatment Technology Incremental Cost	**$1,058**
R&D Engineering Incremental Cost	$0
Certification Incremental Costs	$0
Warranty Incremental Costs	$0
Total Indirect Incremental Costs to Manufacturer	**$0**
Total Incremental Cost Comparison	**$1,808**

NREL's initial estimated incremental costs of the potential diesel technology package, likely to be the highest incremental cost to meet 0.02 g/bhp-hr NO_x for the 12–13-L platform, are depicted in Table 8.

Table 8. NREL Estimates of Potential High-Cost Diesel Technology Package 12–13 L

Cost Component	Incremental Cost Estimate
EGR Cooler Bypass	$330
Charge Air Cooler Bypass	$200
Turbine Bypass	$220
Total Engine Technology Incremental Cost	**$750**
PNA	$1,256
DOC	$4
DPF (2018 baseline system only)	($1,398)
SCRF	$1,300
SCR+ASC and DEF Dosing System	$227
OBD Sensors and Controllers (NO_x, NH_3, and Temp Sensors)	$314
Total Aftertreatment Technology Incremental Cost	**$1,703**
R&D Engineering Incremental Cost	$0
Certification Incremental Costs	$0
Warranty Incremental Costs	$0
Total Indirect Incremental Costs to Manufacturer	**$0**
Total Incremental Cost Comparison	**$2,453**

1.4.3 First Survey Responses for Incremental Costs of Potential Diesel Technologies

NREL received a total of five survey responses from a mix of advanced engine technology and emission control technology trade organizations, Tier 1 suppliers, and engine OEMs. As referenced in the Acknowledgements, MECA responded to the survey in a single, aggregated response (to protect confidential cost information). NREL does not know how many MECA member companies are included in that aggregated response.

As a reminder, the first survey specified:

- 0.02 g/bhp-hr NO_x on FTP, RMC-SET, in addition to the new proposed LLC
- MY 2023 introduction
- Current FUL
- Current warranty offered by the OEMs (whatever that may be)
- Production volumes for all of the United States, with guidance for changes for California-only adoption.

NREL received feedback for U.S. volumes, with very little information regarding impacts for California-only adoption. As NREL was unable to aggregate California-only adoption incremental costs, only incremental costs for U.S. volumes are reported.

After receiving the responses to the first survey request, NREL aggregated the incremental cost data into a range of low, average, and high responses for the potential low-cost diesel technology package, as summarized below for 6–7 L in Table 9 and 12–13 L in Table 10. Note that these low, average, and high incremental cost responses are not to be confused with the proposed low-, average-, and high-cost technology packages. Also, note that the low, average, and high responses for each component/subsystem (row) were calculated so that the total low, average, and high incremental cost may not directly reflect any single survey response.

Table 9. Survey Responses for Potential Low-Cost Diesel Technology Package 6–7 L

6–7 L	Low	Avg.	High
EGR Cooler Bypass	$170	$243	$330
Charge Air Cooler Bypass	$128	$167	$200
Turbine Bypass	$170	$207	$230
Total Engine Technology Incremental Cost	**$468**	**$617**	**$760**
LO-SCR	$401	$944	$2,200
DOC	($15)	$10	$30
DPF	($45)	($17)	$0
SCR+ASC and DEF Dosing System	$300	$621	$823
OBD Sensors and Controllers (NO_x, NH_3, and Temp Sensors)	$141	$333	$800
Other	$50	$175	$300
Total Aftertreatment Technology Incremental Cost	**$832**	**$2,066**	**$4,153**
R&D Engineering Incremental Cost	$70	$85	$100
Certification Incremental Costs	$0	$25	$50
Warranty Incremental Costs	$750	$1,875	$3,000
Total Indirect Incremental Costs to Manufacturer	**$820**	**$1,985**	**$3,150**
Total Incremental Cost Comparison	**$2,120**	**$4,668**	**$8,063**

Table 10. Survey Responses for Potential Low-Cost Diesel Technology Package 12–13 L

12–13 L	Low	Avg.	High
EGR Cooler Bypass	$170	$302	$408
Charge Air Cooler Bypass	$128	$185	$240
Turbine Bypass	$170	$215	$240
Total Engine Technology Incremental Cost	**$468**	**$702**	**$888**
LO-SCR	$574	$1,120	$2,450
DOC	$0	$89	$250
DPF	($98)	($44)	$0
SCR+ASC and DEF Dosing System	$500	$784	$1,100
OBD Sensors and Controllers (NO_x, NH_3, and Temp Sensors)	$158	$330	$600
Other	$50	$150	$300
Total Aftertreatment Technology Incremental Cost	**$1,184**	**$2,429**	**$4,700**
R&D Engineering Incremental Cost	$110	$354	$503
Certification Incremental Costs	$0	$21	$50
Warranty Incremental Costs	$1,500	$1,833	$2,500
Total Indirect Incremental Costs to Manufacturer	**$1,610**	**$2,208**	**$3,053**
Total Incremental Cost Comparison	**$3,262**	**$5,339**	**$8,641**

After receiving the responses to the first survey request, NREL aggregated the incremental cost data into a range of low, average, and high estimates for the potential average-cost diesel technology package, as summarized for 6–7 L in Table 11 and 12–13 L in Table 12.

Table 11. Survey Responses for Potential Average-Cost Diesel Technology Package 6–7 L

6–7 L	Low	Avg.	High
Cylinder Deactivation	$480	$790	$1,140
Other	$150	$505	$860
Total Engine Technology Incremental Cost	**$630**	**$1,295**	**$2,000**
LO-SCR	$401	$944	$2,200
DOC	($15)	$10	$30
DPF	($45)	($17)	$0
SCR+ASC and DEF Dosing System	$300	$621	$823
OBD Sensors and Controllers (NO_x, NH_3, and Temp Sensors)	$141	$333	$800
Other	$50	$175	$300
Total Aftertreatment Technology Incremental Cost	**$832**	**$2,064**	**$4,153**
R&D Engineering Incremental Cost	$70	$85	$100
Certification Incremental Costs	$0	$25	$50
Warranty Incremental Costs	$750	$1,875	$3,000
Total Indirect Incremental Costs to Manufacturer	**$820**	**$1,985**	**$3,150**
Total Incremental Cost Comparison	**$2,282**	**$5,344**	**$9,303**

Table 12. Survey Responses for Potential Average-Cost Diesel Technology Package 12–13 L

12–13 L	Low	Avg.	High
Cylinder Deactivation	$561	$952	$1,550
Other	$150	$625	$1,100
Total Engine Technology Cost	**$711**	**$1,577**	**$2,650**
LO-SCR	$574	$1,120	$2,450
DOC	$0	$89	$250
DPF	($98)	($44)	$0
SCR+ASC and DEF Dosing System	$500	$784	$1,100
OBD Sensors and Controllers (NO_x, NH_3, and Temp Sensors)	$158	$330	$600
Other	$50	$150	$300
Total Aftertreatment Technology Incremental Cost	**$1,184**	**$2,429**	**$4,700**
R&D Engineering Incremental Cost	$110	$354	$503
Certification Incremental Costs	$0	$21	$50
Warranty Incremental Costs	$1,500	$1,833	$2,500
Total Indirect Incremental Costs to Manufacturer	**$1,610**	**$2,209**	**$3,053**
Total Incremental Cost Comparison	**$3,505**	**$6,214**	**$10,403**

After receiving the responses to the first survey request, NREL aggregated the incremental cost data into a range of low, average, and high estimates for the potential high-cost diesel technology package, as summarized for 6–7 L in Table 13 and 12–13 L in Table 14.

Table 13. Survey Responses for Potential High-Cost Diesel Technology Package 6–7 L

6–7 L	Low	Avg.	High
EGR Cooler Bypass	$170	$243	$330
Charge Air Cooler Bypass	$128	$167	$200
Turbine Bypass	$170	$207	$230
Total Engine Technology Incremental Cost	**$468**	**$617**	**$760**
PNA	$701	$883	$1,000
DOC	($15)	($12)	($9)
DPF (2018 baseline system only)	($759)	($549)	($377)
SCRF	$500	$559	$677
SCR+ASC and DEF Dosing System	$584	$722	$793
OBD Sensors and Controllers (NO_x, NH_3, and Temp Sensors)	$141	$214	$313
Other	$50	$50	$50
Total Aftertreatment Technology Incremental Cost	**$1,202**	**$1,868**	**$2,447**
R&D Engineering Incremental Cost	$400	$400	$400
Certification Incremental Costs	$50	$50	$50
Warranty Incremental Costs	$750	$750	$750
Total Indirect Incremental Costs to Manufacturer	**$1,200**	**$1,200**	**$1,200**
Total Incremental Cost Comparison	**$2,870**	**$3,685**	**$4,407**

Table 14. Survey Responses for Potential High-Cost Diesel Technology Package 12–13 L

12–13 L	Low	Avg.	High
EGR Cooler Bypass	$170	$302	$408
Charge Air Cooler Bypass	$128	$185	$240
Turbine Bypass	$170	$215	$240
Total Engine Technology Incremental Cost	**$468**	**$702**	**$888**
PNA	$1,147	$2,270	$3,880
DOC	$0	$11	$22
DPF (2018 baseline system only)	($881)	($673)	($560)
SCRF	$800	$930	$1,162
SCR+ASC and DEF Dosing System	($209)	$387	$723
OBD Sensors and Controllers (NO_x, NH_3, and Temp Sensors)	$158	$254	$330
Other	$50	$75	$100
Total Aftertreatment Technology Incremental Cost	**$1,065**	**$3,253**	**$5,657**
R&D Engineering Incremental Cost	$350	$427	$503
Certification Incremental Costs	$13	$32	$50
Warranty Incremental Costs	$1,500	$1,650	$1,800
Total Indirect Incremental Costs to Manufacturer	**$1,863**	**$2,108**	**$2,353**
Total Incremental Cost Comparison	**$3,396**	**$6,063**	**$8,898**

1.4.4 Incremental Costs of Potential Technologies with Extended FUL and Warranty, and California-Only Volumes

After receiving the responses to the first survey request, NREL aggregated the incremental cost data into a range of low, average, and high estimates, as summarized previously. NREL then followed up with an additional survey to identify incremental costs from the MY 2018 baseline, but also to add extended FUL and warranty per Table 1. Lower production volumes representing California only (instead of all of the United States) were also incorporated. The survey assumed implementation for MY 2027 (instead of MY 2023, as in the first survey), as additional time would be necessary to engineer for extended FUL and warranty. Table 15 through Table 20 summarize these additional survey responses.

Table 15. Survey Responses for Potential Low-Cost Diesel Technology Package 6–7 L with Extended FUL, Extended Warranty, and California-Only Volumes

6–7 L	Low	Avg.	High
EGR Cooler Bypass	$289	$390	$490
Charge Air Cooler Bypass	$191	$225	$259
Turbine Bypass	$255	$296	$345
Total Engine Technology Incremental Cost	**$735**	**$911**	**$1,094**
LO-SCR	$513	$1135	$2,200
DOC	$0	$99	$171
DPF	$0	$95	$164
SCR+ASC and DEF Dosing System	$300	$1161	$1829
OBD Sensors and Controllers (NO_x, NH_3, and Temp Sensors)	$738	$845	$997
Other	$300	$300	$300
Total Aftertreatment Technology Incremental Cost	**$1,851**	**$3,635**	**$5,661**
R&D Engineering Incremental Cost	$70	$70	$70
Certification Incremental Costs	$0	$0	$0
Warranty Incremental Costs	$10,800	$10,800	$10,800
Total Indirect Incremental Costs to Manufacturer	**$10,870**	**$10,870**	**$10,870**
Total Incremental Cost Comparison	**$13,456**	**$15,416**	**$17,625**

Table 16. Survey Responses for Potential Low-Cost Diesel Technology Package 12–13 L with Extended FUL, Extended Warranty, and CA Volumes

12–13 L	Low	Avg.	High
EGR Cooler Bypass	$289	$390	$490
Charge Air Cooler Bypass	$191	$246	$288
Turbine Bypass	$255	$296	$345
Total Engine Technology Incremental Cost	**$735**	**$932**	**$1,123**
LO-SCR	$736	$1,330	$2,450
DOC	$0	$144	$330
DPF	$0	$83	$191
SCR+ASC and DEF Dosing System	$500	$1,240	$1,892
OBD Sensors and Controllers (NO_x, NH_3, and Temp Sensors)	$476	$765	$997
Other	$300	$950	$1,600
Total Aftertreatment Technology Incremental Cost	**$2,012**	**$4,512**	**$7,460**
R&D Engineering Incremental Cost	$110	$357	$603
Certification Incremental Costs	$0	$7	$13
Warranty Incremental Costs	$7,840	$23,061	$38,282
Total Indirect Incremental Costs to Manufacturer	**$7,950**	**$23,424**	**$38,898**
Total Incremental Cost Comparison	**$10,697**	**$28,868**	**$47,481**

Table 17. Survey Responses for Potential Average-Cost Diesel Technology Package 6–7 L with Extended FUL, Extended Warranty, and California-Only Volumes

6–7 L	Low	Avg.	High
Cylinder Deactivation	$638	$880	$1,140
Other	$860	$860	$860
Total Engine Technology Incremental Cost	**$1,498**	**$1,740**	**$2,000**
LO-SCR	$513	$1,135	$2,200
DOC	$0	$99	$171
DPF	$0	$95	$164
SCR+ASC and DEF Dosing System	$300	$1,161	$1,829
OBD Sensors and Controllers (NO_x, NH_3, and Temp Sensors)	$738	$845	$997
Other	$300	$300	$300
Total Aftertreatment Technology Incremental Cost	**$1,851**	**$3,635**	**$5,661**
R&D Engineering Incremental Cost	$70	$70	$70
Certification Incremental Costs	$0	$0	$0
Warranty Incremental Costs	$10,800	$10,800	$10,800
Total Indirect Incremental Costs to Manufacturer	**$10,870**	**$10,870**	**$10,870**
Total Incremental Cost Comparison	**$14,219**	**$16,245**	**$18,531**

Table 18. Survey Responses for Potential Average-Cost Diesel Technology Package 12–13 L with Extended FUL, Extended Warranty, and California-Only Volumes

12–13 L	Low	Avg.	High
Cylinder Deactivation	$724	$1,176	$1,860
Other	$1,100	$1,100	$1,100
Total Engine Technology Cost	**$1,824**	**$2,276**	**$2,960**
LO-SCR	$736	$1,330	$2,450
DOC	$0	$144	$330
DPF	$0	$83	$191
SCR+ASC and DEF Dosing System	$500	$1,240	$1,892
OBD Sensors and Controllers (NO_x, NH_3, and Temp Sensors)	$476	$765	$997
Other	$300	$950	$1,600
Total Aftertreatment Technology Incremental Cost	**$2,012**	**$4,512**	**$7,460**
R&D Engineering Incremental Cost	$110	$357	$603
Certification Incremental Costs	$0	$7	$13
Warranty Incremental Costs	$7,840	$23,061	$38,282
Total Indirect Incremental Costs to Manufacturer	**$7,950**	**$23,424**	**$38,898**
Total Incremental Cost Comparison	**$11,786**	**$30,212**	**$49,318**

Table 19. Survey Responses for Potential High-Cost Diesel Technology Package 6–7 L with Extended FUL, Extended Warranty, and California-Only Volumes

6–7 L	Low	Avg.	High
EGR Cooler Bypass	$289	$340	$391
Charge Air Cooler Bypass	$191	$225	$259
Turbine Bypass	$255	$296	$345
Total Engine Technology Incremental Cost	**$735**	**$865**	**$995**
PNA	$924	$1,097	$1,250
DOC	$101	$119	$136
DPF (2018 baseline system only)	($511)	($444)	($377)
SCRF	$679	$799	$919
SCR+ASC and DEF Dosing System	$1,374	$1,616	$1,858
OBD Sensors and Controllers (NO_x, NH_3, and Temp Sensors)	$738	$868	$997
Other	$0	$0	$0
Total Aftertreatment Technology Incremental Cost	**$3,305**	**$4,044**	**$4,783**
R&D Engineering Incremental Cost	$xx	$xx	$xx
Certification Incremental Costs	$xx	$xx	$xx
Warranty Incremental Costs	$xx	$xx	$xx
Total Indirect Incremental Costs to Manufacturer	**$xx**	**$xx**	**$xx**
Total Incremental Cost Comparison	**$xx**	**$xx**	**$xx**

Note for Table 19 that insufficient responses were received for this technology package with respect to indirect costs to allow sufficient aggregation. Therefore, indirect and total incremental costs were not calculated.

Table 20. Survey Responses for Potential High-Cost Diesel Technology Package 12–13 L with Extended FUL, Extended Warranty, and California-Only Volumes

12–13 L	Low	Avg.	High
EGR Cooler Bypass	$289	$390	$490
Charge Air Cooler Bypass	$191	$246	$288
Turbine Bypass	$255	$296	$345
Total Engine Technology Incremental Cost	**$735**	**$932**	**$1,123**
PNA	$1,592	$2,801	$4,656
DOC	$0	$153	$263
DPF (2018 baseline system only)	($881)	($698)	($560)
SCRF	$960	$1,220	$1,553
SCR+ASC and DEF Dosing System	($209)	$1,077	$1,977
OBD Sensors and Controllers (NO_x, NH_3, and Temp Sensors)	$426	$720	$997
Other	$1,600	$1,600	$1,600
Total Aftertreatment Technology Incremental Cost	**$3,488**	**$6,873**	**$10,486**
R&D Engineering Incremental Cost	$603	$603	$603
Certification Incremental Costs	$13	$13	$13
Warranty Incremental Costs	$38,621	$38,621	$38,621
Total Indirect Incremental Costs to Manufacturer	**$39,237**	**$39,237**	**$39,273**
Total Incremental Cost Comparison	**$43,460**	**$47,042**	**$50,846**

It should be noted that the total indirect incremental cost estimates by manufacturers, and the total incremental costs in Table 15 to Table 20, are dominated by the warranty incremental costs. In some cases, the high estimate of incremental warranty costs is over $38,000. As discussed in Section 1.4.5, the warranty incremental costs were based on a very small sample size, and may be biased high due to the OEMs' uncertainty regarding covering warranty for unfamiliar technology needed to meet a 0.02 g/bhp-hr NOx standard at the same time with much longer FULs than current FULs.

1.4.5 Incremental Cost Survey Response Observations

The following general observations can be made regarding the incremental costs reported in Table 3 through Table 20.

- The initial NREL estimates for total incremental costs were fairly close to the lower end of survey responses for the first survey (MY 2023, U.S. volume, current FUL).
- Indirect costs are a significant portion of the total cost.

- Total costs are not necessarily tied to engine displacement/power but are heavily dependent on indirect costs. Production volumes of various engine displacements have more of an impact than engine "size" on indirect cost, and therefore total incremental cost.
- High engineering, certification, and warranty costs spread over relatively small volumes are the drivers of indirect costs. Survey respondents did not share amortization strategies or exact volumes, so those effects are unknown.
- Only OEMs responded with indirect costs, as Tier 1 and MECA responses included only direct costs. Due to the limited number of OEM responses, the indirect costs may have a high level of variation and may not necessarily represent indirect costs for all OEMs.
- The second survey (MY 2027, California-only volume, extended FUL and warranty) was intended to present "worst case" in many parameters, and the survey results reflect that.
- The second survey results report very high incremental indirect costs, especially for warranty. The OEMs did not break that warranty down into how much was attributed to extended FUL versus the extension of the warranty period. Feedback from OEMs indicated high levels of uncertainty in projected warranty costs for this scenario.
- The second survey results assumed CA-only volumes, but OEMs were free to interpret that assumption on their own. OEMs did not report how these CA-only volumes differed from U.S. volumes in the first survey. They did not explicitly state different assumptions regarding market share or changes in CA-only volume due to potential increased pre-purchases ahead of new emissions regulations or potential reduced purchases due to new emissions regulations.
- Some apparent anomalies in the survey responses may be attributed to the limited number of responses. As noted above, not all respondents reported incremental cost estimates for all proposed technology combinations. The aggregated data reported is the best NREL has available that still protects individual confidential costing information.

1.4.6 Incremental Costs for Natural Gas and Gasoline Technology Packages

As previously referenced, few responses were received for the natural gas (HHDD standard) engine platform, preventing NREL from sufficiently aggregating incremental cost information to protect proprietary information. The study assumption that natural-gas engine technology meeting CARB's current optional low-NO_x certification at 0.02 g/bhp-hr would require no significant upgrades to meet a proposed 0.02 g/bhp-hr standard with a new LLC was flawed, based on industry feedback. The feedback focused on changes needed to meet the new LLC cycle and the potential that a moving average window method for emission compliance may be necessary. Based on NREL's analysis and research from literature review, trade organization feedback, and OEM feedback, the anticipated incremental cost of both indirect and direct incremental costs for natural-gas engines and aftertreatment technology to meet an MY 2023 target of 0.02 g/bhp-hr utilizing the moving average window method to assess emission compliance is within 10% of the low-cost diesel technology package for equivalent displacement. A round number estimate total of $3,000 incremental cost was subsequently used for the Task 2: Engine Life-Cycle Costs study.

Similarly, few responses were received for the gasoline HDO engine platform. Some aggregation was possible for direct costs, but only NREL estimates were available for indirect costs. As a result, only total integrated (including direct and indirect) incremental costs ranging from $353 to $468 for MY 2023 were calculated with current FUL.

1.5 Low-, Average-, and High-Cost Estimates

Because NREL received a range of values in response to both surveys, the diesel incremental cost analysis results in nine different points of costs, with low-, average-, and high-cost responses to each of the potential low-, average-, and high-cost diesel technology packages.

1.5.1 Low-, Average-, and High-Cost Estimates for MY 2023 with Current FUL and Warranty

These different points of cost defining the range of data received in response to the first survey for MY 2023 and current full useful life as defined in Table 1 are depicted by error bars within the summary graphs in Figure 3 and Figure 4. The incremental cost variance within any one package is larger than the differences between the engine and aftertreatment packages. In addition, the range of costs seem to have a greater impact on the larger displacement platforms, resulting in a large variance within the individual technology packages.

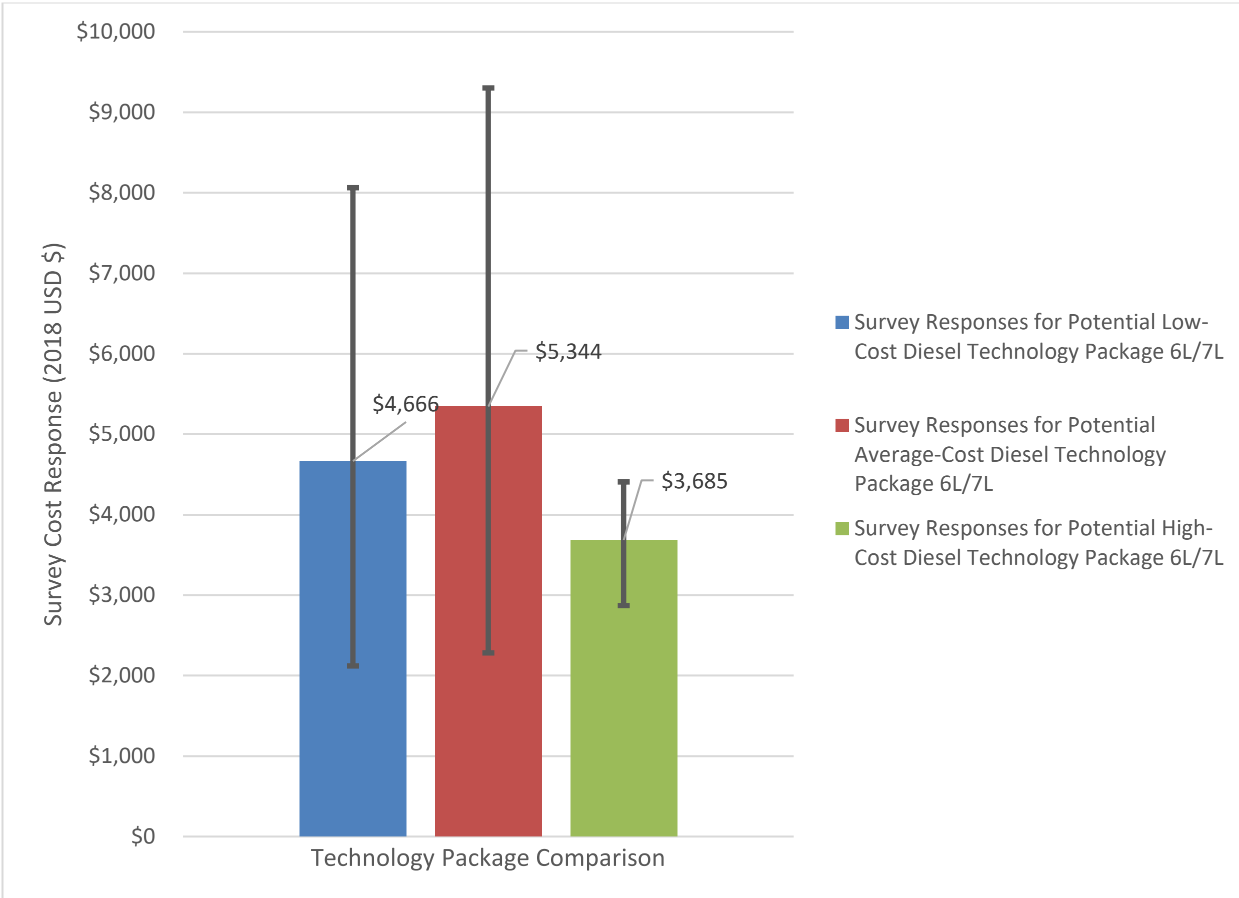

Figure 3. Summary of 6–7-L potential technology packages for MY 2023 with current FUL

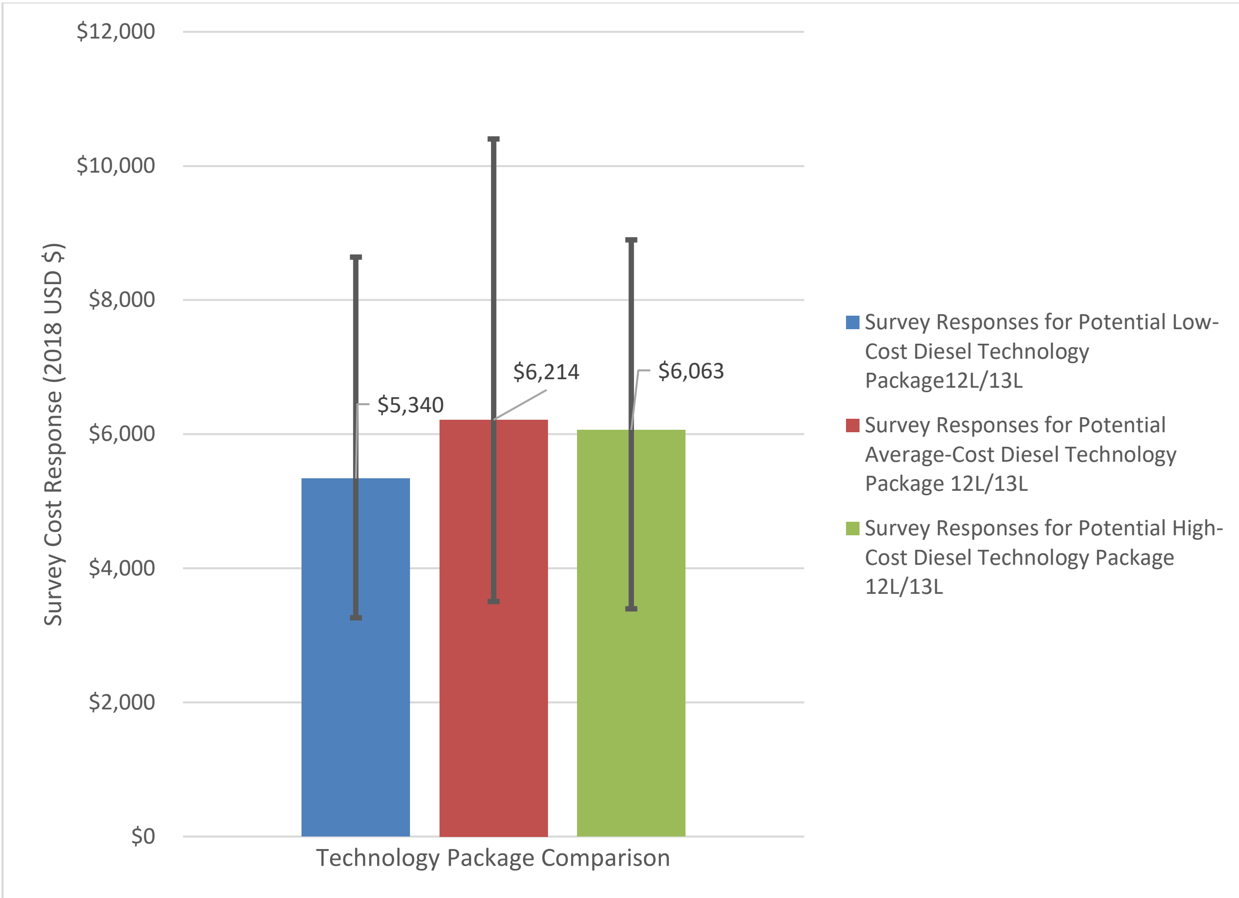

Figure 4. Summary of 12–13-L potential technology packages for MY 2023 with current FUL

1.5.2 Low-, Average-, and High-Cost Estimates for MY 2027 with Extended Warranty and Extended Useful Life

The range of incremental costs received in response to the second survey for MY 2027 with extended useful life and warranty as defined in Table 1 are depicted by error bars within the summary graphs in Figure 5 and Figure 6. NREL did not receive enough responses for the third technology package of the potential high-cost diesel technology to aggregate and therefore did not include the estimates received in order to protect the source of the data.

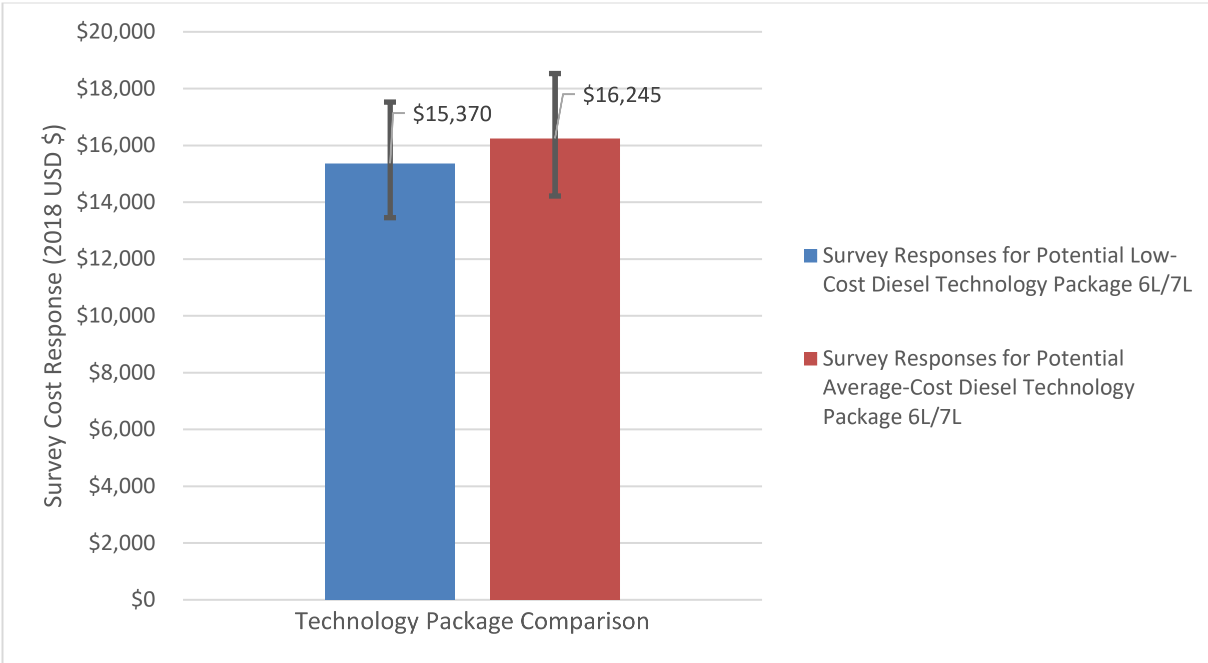

Figure 5. Summary of 6–7-L potential technology packages for MY 2027 with extended FUL and warranty

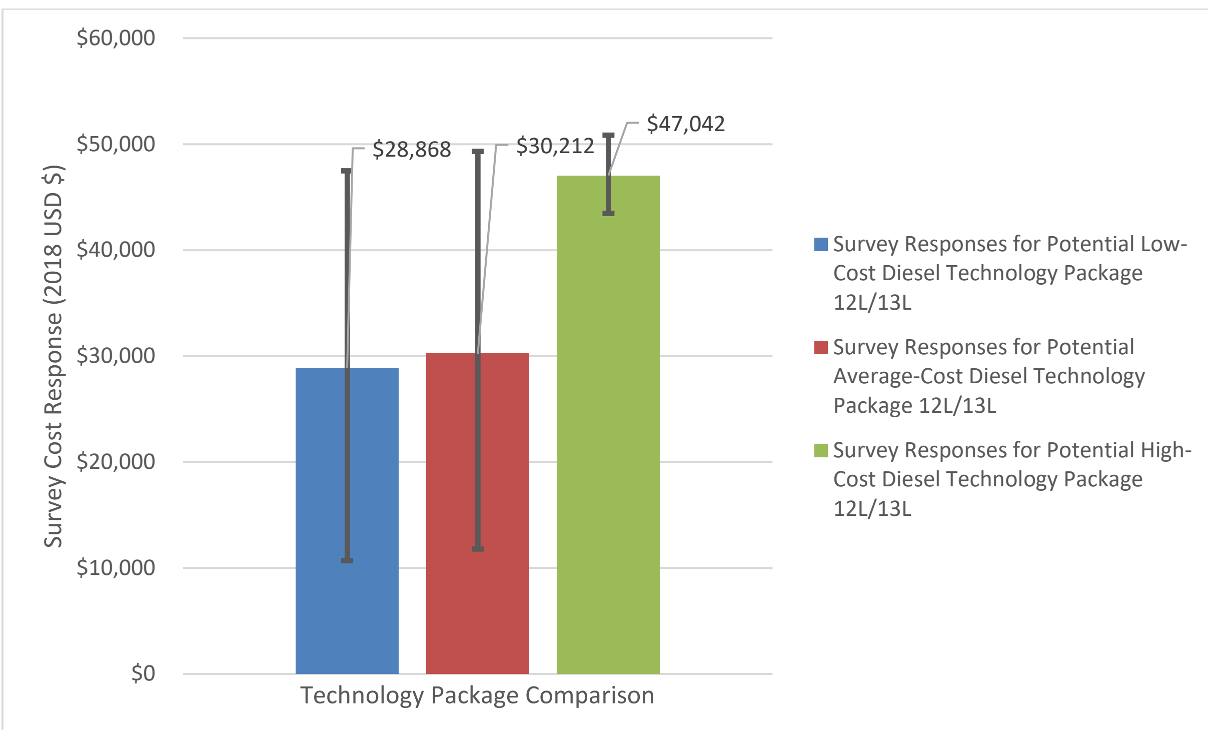

Figure 6. Summary of 12–13-L potential technology packages for MY 2027 with extended FUL and warranty

1.6 Summary of Incremental Cost Analysis

NREL received a total of five survey responses from a mix of advanced engine technology and emission control technology trade organizations, Tier 1 suppliers, and engine OEMs. Data were aggregated with the incremental cost estimates NREL derived from literature review and engineering judgments. The survey responses included incremental cost estimates in a range of values, creating variance for each potential low-, average-, and high-cost technology package. The wide variance in the SCR+ASC and DEF dosing system costs drive most of the variance within the total aftertreatment costs. The cost variance is also much greater in larger displacements due to the high costs of the aftertreatment components and the variance within each of those. Indirect costs are a significant portion of the combined hardware costs of the engine and aftertreatment. Lastly, the incremental costs were not adjusted to reflect a retail markup due to the complexity with which pricing decisions are made.

2 Task 2: Engine Life-Cycle Costs

This section details a life-cycle cost analysis completed to understand the true costs to the owner of a vehicle with a 0.02 g/bhp-hr NO_x aftertreatment package outside of the direct upfront vehicle cost increase. The life-cycle cost analysis sought to incorporate costs associated with the following elements:

- Initial purchase cost
- Fuel consumption changes (changes in fuel economy)
- DEF consumption
- Maximum useful life of the aftertreatment package (major overhaul intervals)
- Other operating and maintenance costs.

To complete the life-cycle cost analysis, two main tasks were completed: assessing the maximum useful life for the aftertreatment packages and computing the life-cycle costs. Section 2.1 reviews the maximum useful life analysis in detail, Section 2.2 reviews the life-cycle cost approach, Section 2.3 outlines the scenarios evaluated in this study, and Section 2.4 summarizes the results of the life-cycle cost analysis.

2.1 Maximum Full Useful Life Analysis

The maximum useful life for the aftertreatment system determines the mileage at which costs to the owner may be incurred if the system begins to fail. For all scenarios in the life-cycle cost analysis, the incremental cost associated with the aftertreatment package was assumed to be incurred after the truck mileage exceeded the stated maximum useful life. This assumption is expected to be conservative as not all aftertreatment packages will fail immediately after they exceed their stated maximum useful life. Statistical analysis of failure rates combined with data on aftertreatment technology operating and maintenance costs could give a more accurate depiction of life-cycle costs. However, such data are not currently available.

The extended maximum useful life option was evaluated by considering the tradeoff between increased upfront costs due to improved durability needed for the extended maximum useful life[1] and the decrease in owner-related replacement costs at the end of the maximum useful life.

The maximum useful life depends on both the displacement of the vehicle and the fuel type. The extended maximum useful life values were defined based on the CARB proposal in January 2019 and previously shown in Table 1.

2.2 Approach

This analysis leverages the high-fidelity vehicle stock model within NREL's Scenario Evaluation and Regionalization Analysis (SERA) model. The SERA stock model tracks vehicle miles traveled, fuel consumption, and ownership costs throughout each vehicle's lifetime and is resolved temporally and spatially with high fidelity. The SERA model was complemented by

[1] It is important to note that the data received from the cost survey (Section 1.3) combined both an extended useful life and an extended warranty. Thus, the cost data used for the extended useful life scenarios couples both the extended useful life and extended warranty information together.

additional data sets to effectively map the vehicles to the aftertreatment packages evaluated in this study.

The following sections provide a brief overview of the SERA stock model, the data sources used in this study, model validation, scenario design, and the life-cycle cost results.

2.2.1 Scenario Evaluation and Regionalization Analysis (SERA) Model

The SERA model's stock module capability provides a flexible framework for tracking vehicles over their life. The SERA's stock model has been used for a variety of U.S. Department of Energy and California Energy Commission projects and, in particular, is described in detail in Bush et al. (2019). The general data flow for the SERA stock model is shown in Figure 7, which shows how data for regional sales (total vehicles sold), market shares (disaggregation of vehicle sales by vehicle type), vehicle survival (salvage rate data), annual travel (vehicle-miles traveled), fuel consumption data (fuel economy and fuel types), and emission rate data are combined to track vehicle population, travel, and resulting energy consumption and emissions.

For this analysis, the SERA model was expanded to track vehicle life-cycle costs over the vehicle's lifetime. The model was updated to account for vehicle costs that could be incurred when purchasing a vehicle or driving the vehicle, as the model already has those data within it.

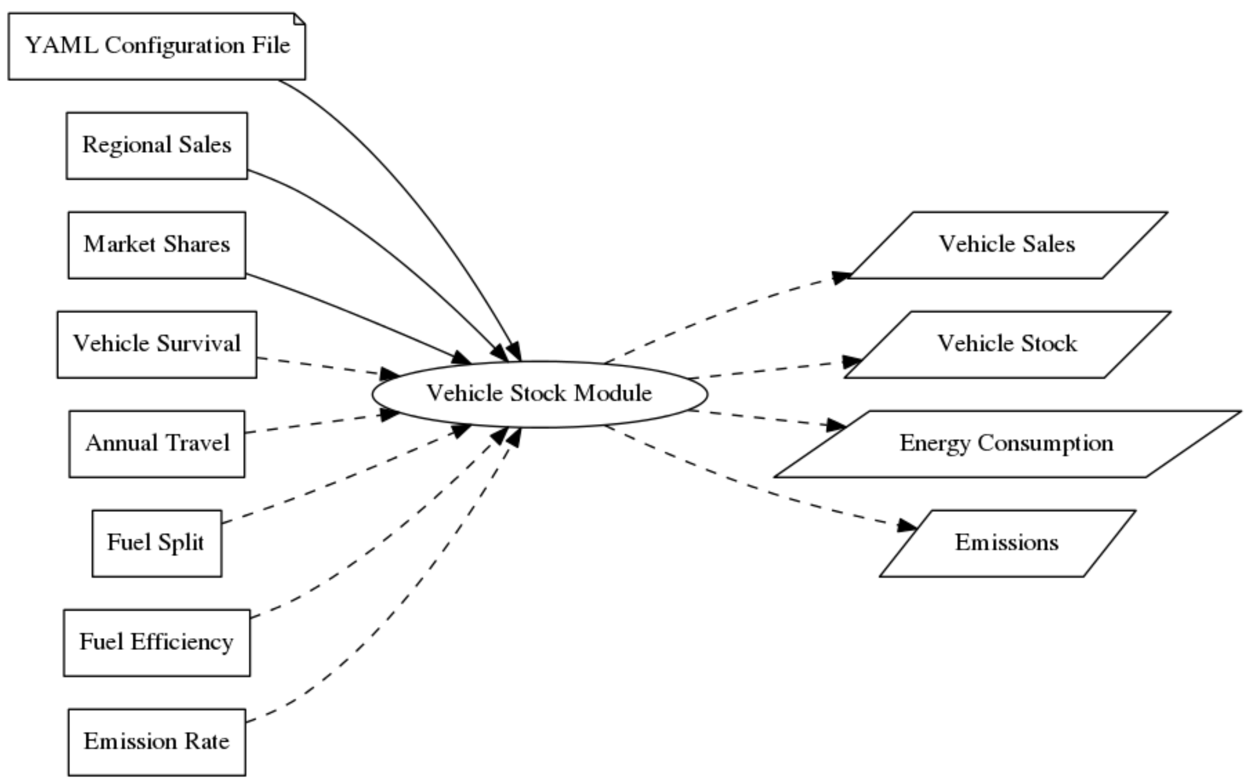

Figure 7. The general SERA stock model data flow

2.2.2 Data Sources

The SERA model provides the analytic framework for a detailed stock model but is complemented by additional data sets to complete the life-cycle analysis required in this study. The data sources used in this analysis are summarized in Table 21.

Table 21. Data Sources Used in Life-Cycle Cost Analysis

Data Source	Description	How it was used
EMFAC/CA Vision 2.1	The EMFAC emissions model is used by CARB to assess emissions from on-road vehicles (cars, trucks, and buses). The CA Vision 2.1 model (2017) is a scenario-planning model and provides the detailed stock data required for the SERA model. It should be noted that the CA Vision model is based on the EMFAC 2014 results.	The CA Vision 2.1 model data was used as the base stock model to create within SERA (e.g., vehicle sales, survival, vehicle miles traveled, and fuel economy were matched between SERA and the CA Vision 2.1 model). Thus, the SERA stock model vehicles, population, total mileage, and fuel consumption match the EMFAC and CA Vision 2.1 models.
IHS Markit (Polk) Department of Motor Vehicles Registration Data	The IHS Markit (formerly known as Polk) Department of Motor Vehicles registration database (2013) provides data across the United States on the quantity and types of trucks registered in each zip code.	The IHS Markit data were used to disaggregate EMFAC vehicles by their engine displacement to compute fleet-wide costs. For example, the T6 Instate Small truck comprises GVWR classes 4–7, which correspond to multiple engine displacements. The IHS Markit data were used to determine the fraction of T6 Instate Small trucks within each engine displacement class.
Task 1 Cost Data	The Task 1 survey cost data includes the incremental cost for three different aftertreatment packages, two engine displacements, three different fuel types, different maximum useful life estimates, different manufacturing volumes, and different model years.	The Task 1 data were incorporated into the SERA model as upfront costs to the vehicle owner mapped to the appropriate vehicle (model year, engine displacement, fuel type). The incremental upfront cost was also assumed to be incurred after the maximum useful life of the aftertreatment package was surpassed in most scenarios.
California Energy Commission Fuel Prices	California Energy Commission's forecast of fuel prices (2017)	Scenario analysis was used to evaluate a 1.25% improvement in fuel economy. The marginal improvement in fuel economy results in fuel cost savings during the vehicle's life. Preliminary data from SwRI indicates an improvement of 0%–4%, depending on the engine cycle, with 1.25% as a good central estimate per SwRI feedback. No reductions in fuel economy were evaluated as the vehicles must still meet the existing GHG standards regulated by CARB.
Diesel Exhaust Fluid Price	A constant $6/gal DEF cost was assumed based on NREL's Co-Optima analysis	Scenario analysis as completed to determine the life-cycle cost of increased DEF consumption.

As seen in Table 21, there are several data sources that combine within the SERA model to evaluate the life-cycle cost of the low-NO_x fuel standard. Visually, these data sources are combined as seen in Figure 8.

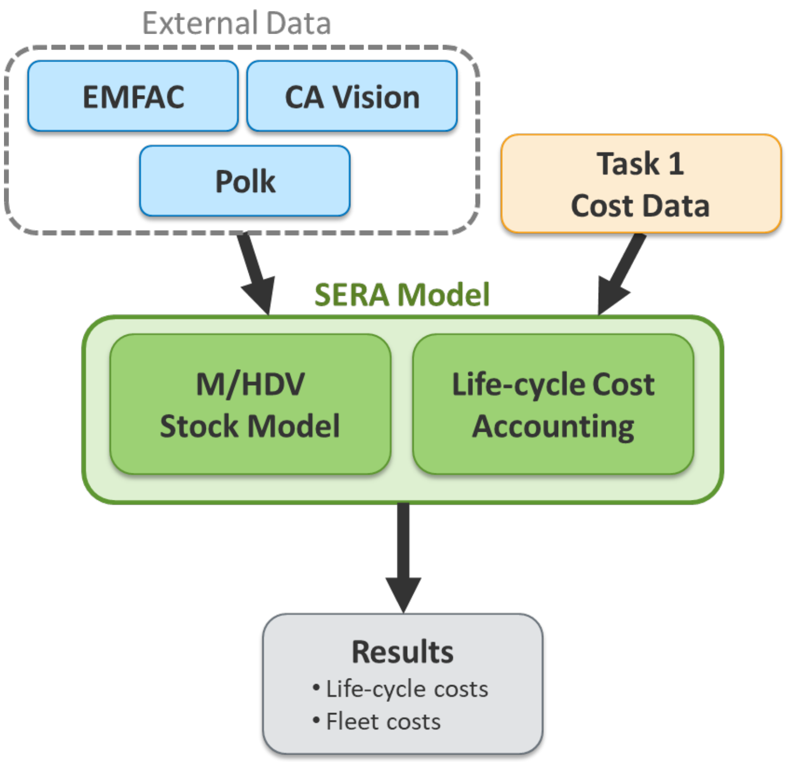

Figure 8. Data flow and analysis using the SERA model for life-cycle cost analysis

Due to the EMFAC and CA Vision 2.1 model spatial and temporal fidelity, each vehicle is defined by a specific region, vocation, model year, fuel type, and age. These vehicles are then further disaggregated by engine displacement using the IHS Markit (formerly Polk) Department of Motor Vehicles registration data. Thus, the life-cycle costs for each vehicle are a function of all of these parameters, and there is a distribution of life-cycle costs across the California fleet due to different vehicle types and travel profiles. For example, the life-cycle costs for a Class 8 long haul tractor will be very different than a Class 6 parcel delivery truck due to the different aftertreatment package costs (which vary by displacement), in addition to the different marginal fuel cost reductions, because they have very different travel requirements profiles and fuel economies.

The distribution in life-cycle costs will be analyzed across the California fleet vehicle types, engine technologies, displacements, and regions using multiple analytic methods, including scenario analysis and sensitivity analysis.

2.2.3 SERA Model Validation

The SERA model was validated against the CA Vision 2.1 model to ensure the starting point for the life-cycle cost analysis was accurate. Figure 9 summarizes the results of the model validation, which show very close agreement between the SERA model and the CA Vision model for predicting stock through 2050. Additionally, validating the model by region, Figure 9 shows there is a less than 1.2% error in predicting the California vehicle population through 2050 for each region.

This model validation indicates that the SERA model matches the CA Vision 2.1 model closely through 2050. For this analysis, the life-cycle cost analysis is focused on model years 2023 and 2027, so this validation signifies that those vehicle sales and survival (lifetimes) will be accurately accounted for in the life-cycle analysis. Additionally, the vehicle travel and fuel consumption data influence the life-cycle costs for each vehicle, and this validation indicates that those costs will be accurately accounted for.

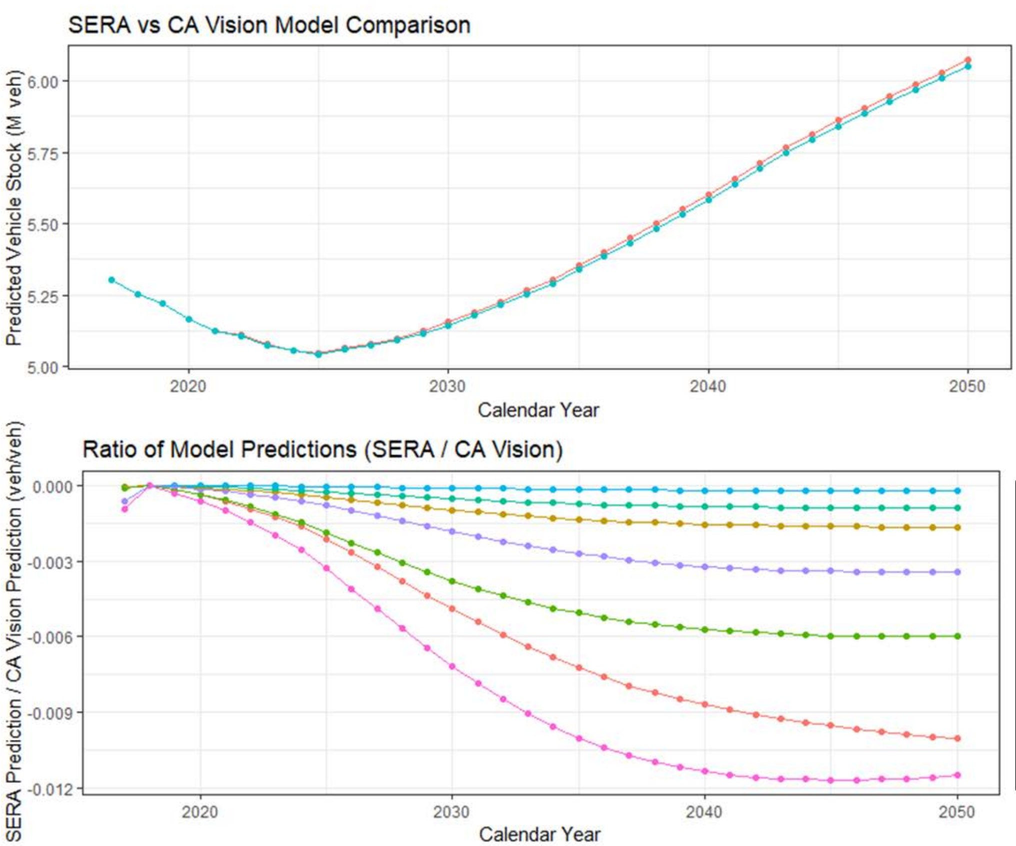

Figure 9. SERA model validation against the CA Vision 2.1 model

2.2.4 Manufacturing Volume Analysis

Manufacturing volume influences the upfront cost of aftertreatment systems, as large manufacturing volumes allow the firm to spread capital and fixed operating costs over more units sold, reducing the per-unit cost. As discussed in the Task 1 section of this report, most data collected from OEMs are for a national manufacturing volume. One OEM provided cost estimates for the 12–13-L diesel engine for a California-only manufacturing volume basis. These data were included in the sensitivity analysis to show its potential importance but not in the scenario analysis given the limited data set.

2.3 Parameters Investigated

The realized life-cycle cost to the vehicle owner depends on a variety of parameters that need to be evaluated. Some of the key parameters assessed in this study include:

- Aftertreatment design cost basis (Task 1)

- Extended maximum useful life
- Manufacturing volume
- Engine displacement
- Vehicle type, region, model year
- Fuel economy impact
- DEF consumption impact.

These parameters and their analysis bounds are summarized in Table 22. Each parameter was varied independently of others to understand the life-cycle cost sensitivity to that parameter.

Table 22. Life-Cycle Cost Parameters Investigated in this Study

Parameter	Description
Adoption Rate	1) 100% compliance by 2023 (Current useful life, only) 2) 100% by 2027 (Extended full useful life, only)
Max Useful Life	1) (Min) Current useful life 2) (Max) Extended useful life 3–5) 25%/50%/75% of min/max spread
Cost Basis	1–3) Low/Avg/High cost basis from Task 1
Other	Will be needed to investigate life-cycle costs differences due to: 1) Varying aftertreatment packages (displacement) 2) Vehicle types (EMFAC definition) 3) Region (Seven CA Vision 2.1 Model Regions) 4) Model year (2023, 2027) 5) Fuel economy impacts (e.g., no change, 1.25% improvement) 6) DEF consumption changes (e.g., 0%, 2.5%, 5% change) 7) Discount rates (3%, 7%) 8) Manufacturing volume (U.S. vs. California-only)

Due to the large number of parameters, each with its own uncertainty around it, the results look at a scenario analysis (varying multiple parameters at one time) and a sensitivity analysis (varying one parameter at a time).

Adoption rate was originally intended to be a parameter of investigation. However, data were only available for current useful life with 100% compliance by 2023 and extended useful life with 100% compliance by 2027. No data were available to determine learning curves or how costs might change depending on the adoption deadline. For this reason, it was assumed that the current full useful life costs for 2023 adoption would hold for 2027 adoption as well. This allows side-by-side comparison of current and extended full useful life life-cycle costs.

2.3.1 Scenario Analysis

Due to the large number of parameters that could influence the life-cycle cost of each vehicle, a scenario analysis approach was taken. Three scenarios were defined to understand the bounds on the life-cycle costs: low-cost scenario, mid-cost scenario, and high-cost scenario. These scenarios were defined to bound the life-cycle cost as well as provide a scenario evaluating a mid-cost life-cycle analysis; however, they do not represent the most likely scenarios that could be realized.

The three scenarios are defined in Table 23 and outline the parameter assumptions used for each scenario. The scenarios were defined to look at the bounds of the life-cycle cost analysis, while the sensitivity analysis was completed to understand the critical parameters driving the life-cycle cost of the aftertreatment system. Because California manufacturing volume data were available from only one OEM for only one engine displacement, all scenarios consider U.S. manufacturing volumes.

Additionally, the upfront cost (Task 1 data) was based only on the average-cost technology package and used the low/average/high error bar bounds. This technology package was selected because the error bar bounds of the average-cost technology package effectively span the full spectrum of potential costs (as seen in Section 1.4). Additionally, the low-cost technology package and high-cost technology package may not actually represent the lowest-cost or highest-cost packages, as found from the survey data in Task 1.

Table 23. Scenario Definitions for Bounding Analysis

Parameter	Low-Cost Scenario	Mid-Cost Scenario	High-Cost Scenario
Upfront Cost	Low	Mid	High
Manufacturing Scale	U.S.	U.S.	U.S.
Useful Life	Current Full Useful Life	Current Full Useful Life	Extended Full Useful Life
Fuel Economy Change	1.25% improvement	No change	No change
DEF Consumption Impact	No change	2.5% increase	5% increase
Discount Rate	7%	7%	3%

In addition to the above parameters, the life-cycle cost also depends on the model year of the vehicle (compliance rate), the engine displacement, the fuel type (diesel, gasoline, natural gas), the vehicle's vocation (defined by EMFAC, which affects the vehicle miles traveled over its lifetime), as well as the region the vehicle is operating in (vehicle miles traveled varies slightly by region within the EMFAC model). Thus, to explore the life-cycle costs across this parameter space, three primary metrics were evaluated for each scenario:

1. Life-cycle costs for each vehicle/displacement/fuel/vocation/region combination
2. A vehicle sales weighted-average life-cycle cost across all vehicle/displacement/fuel/vocation/region combinations
3. A life-cycle cost across the full California fleet.

First, the life-cycle cost was calculated for each vehicle, engine displacement, fuel technology, EMFAC vocation, and region within each of low-cost, mid-cost, and high-cost scenarios. This provides vehicle-specific data and can be used to demonstrate the potential life-cycle costs that could be realized for each vehicle owner.

Second, a sales-weighted average life-cycle cost was determined based on the CA Vision 2.1 predicted sales for the model year 2027. This average metric weights the regions and vocations more heavily if there are more vehicles sold in that aftertreatment definition. For example, assume there are only two vehicles in California and each has a different life-cycle cost and are sold in different proportions, as seen in Table 24.

Table 24. Example Vehicle Sales Weighted Average

Vehicle/Vocation	Example Life-Cycle Cost	Example Sales (vehicles)
T7 Tractor	$1,000	100
T7 Single	$2,000	50

One estimate of representative life-cycle costs for vehicles in California may be a simple average of the two life-cycle costs ($1,500). However, a more accurate and representative life-cycle cost would be a vehicle sales weighted average that accounts for the relative proportion of vehicles within each vocation ($1,333).[2] This approach was used to estimate a single life-cycle cost across all vehicles in California, which would represent an approximate cost for all vehicle owners in the state.

To complete the sales-weighted average, the EMFAC vehicles must be disaggregated into specific vocation, fuel, and engine displacement categories. IHS Markit (formerly Polk) Department of Motor Vehicles registration data were used to disaggregate the EMFAC vehicles into the appropriate vocation, fuel, and engine displacement categories. A summary of the breakdown can be found in Appendix B, while the full data file is provided as an attachment to the report.

In addition to the vehicle-specific life-cycle costs discussed previously, the life-cycle costs of all vehicles sold across California in 2027 were assessed for each scenario. This metric accounts for the relative proportion of vehicle types sold in California and the total cost California fleet owners would be expected to bear for each scenario. This calculation also accounts for the fact that not all vehicles survive the full expected lifetime (e.g., some Class 8 tractors will last only three years while others will last seven). These survival data are important, as vehicles may be retired before they travel more than the aftertreatment package's maximum useful life and thus would not incur those future replacement costs.

2.3.2 Sensitivity Analysis

To better understand the relative importance of each parameter affecting the life-cycle cost of the aftertreatment package, a sensitivity analysis was completed. A sensitivity analysis varies one single parameter and then shows the impact of that parameter on the life-cycle cost of the vehicle. For this analysis, the mid-cost scenario was used as the starting point for the sensitivity analysis, and the variation in each parameter either increases or decreases the life-cycle cost. By varying each parameter independently, one can determine which parameters are the key cost drivers for the life-cycle cost.

[2] Calculated as: $1,000 * (100/(100 + 50)) + $2,000 * (50/(100 + 50)) = $1,333/vehicle

2.4 Results

The results are presented in three sections: a case study to demonstrate life-cycle cost methodologies, scenario analysis results, and a sensitivity analysis.

The case study section illustrates the calculation methodologies that are described above and ultimately used in both the scenario and sensitivity analyses. The case study looks at the calculation methods and assumptions through the lens of two specific vehicles of interest to CARB: the T7 Tractor (heavy heavy-duty tractor truck) and the T6 OOS small (medium heavy-duty out-of-state truck with GVWR ≤ 26,000 lb) (CARB 2018b). The case-study graphics aim to systematically depict some of the key calculation assumptions, limitations, and findings in an easier-to-understand format than when aggregated across all the California vehicles, vocations, displacements, regions, and scenario descriptions. Additional, single-vehicle results for EMFAC vehicles of specific interest to CARB can be found in Appendix A.

The Scenario Analysis and Sensitivity Analysis sections then summarize the core findings of the study, as discussed in Section 2.3.

2.4.1 Case Study: T7 Tractor and T6 OOS Small Vehicle Life-Cycle Costs

The life-cycle cost analysis methodologies are most easily understood through a specific example. Figure 10 shows the present value annual costs[3] for a T7 Tractor (Class 8 line-haul) equipped with a 12–13-L diesel engine for two aftertreatment scenarios: (1) current FUL and (2) extended FUL. Life-cycle costs include the incremental replacement costs after full useful life is achieved (vehicle costs) and potential fuel economy improvements associated with the aftertreatment technology discounted back to present value (fuel costs). For the T7 Tractor 12–13-L engine, the current full useful life is 435,000 miles. If designed for this lifespan, the aftertreatment technology would require two replacements. Extending the aftertreatment's full useful life to 1,000,000 miles significantly increases the upfront cost of the aftertreatment technology but eliminates the need for replacements through 2050, as seen in Figure 10.

[3] The present value annual costs for future years are determined using the discount rate (7% for Figure 10). All values are reported in 2018 dollars, consistent with the Task 1 data, and the first year for discounting is assumed to be in 2027. Using this convention, the incremental vehicle costs (i.e., those due directly to the aftertreatment package) incurred in year 2027 exactly match the Task 1 incremental cost data, while future years are lower due to discounting.

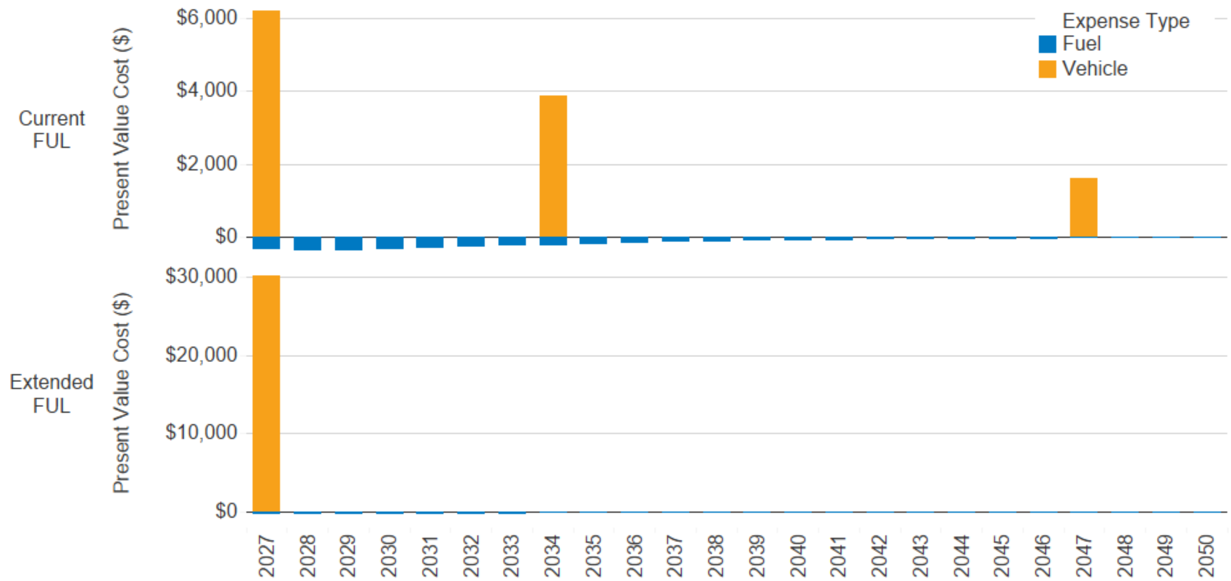

Figure 10. Annual present value cost for a T7 Tractor 12-L diesel engine designed for current full useful life (435,000 miles; top) and extended full useful life (1,000,000 miles; bottom) for MY 2027 in the South Coast Air Basin with a 2.5% increase in DEF consumption, a discount rate of 7%, and national manufacturing volumes

Figure 11 shows annual costs for a T6 OOS small truck with a 6–7-L diesel engine. For the current full useful life design scenario of 110,000 miles, the aftertreatment technology must be replaced three times through 2050. Designing the aftertreatment technology for an extended full useful life of 550,000 miles results in no aftertreatment replacements through 2050.

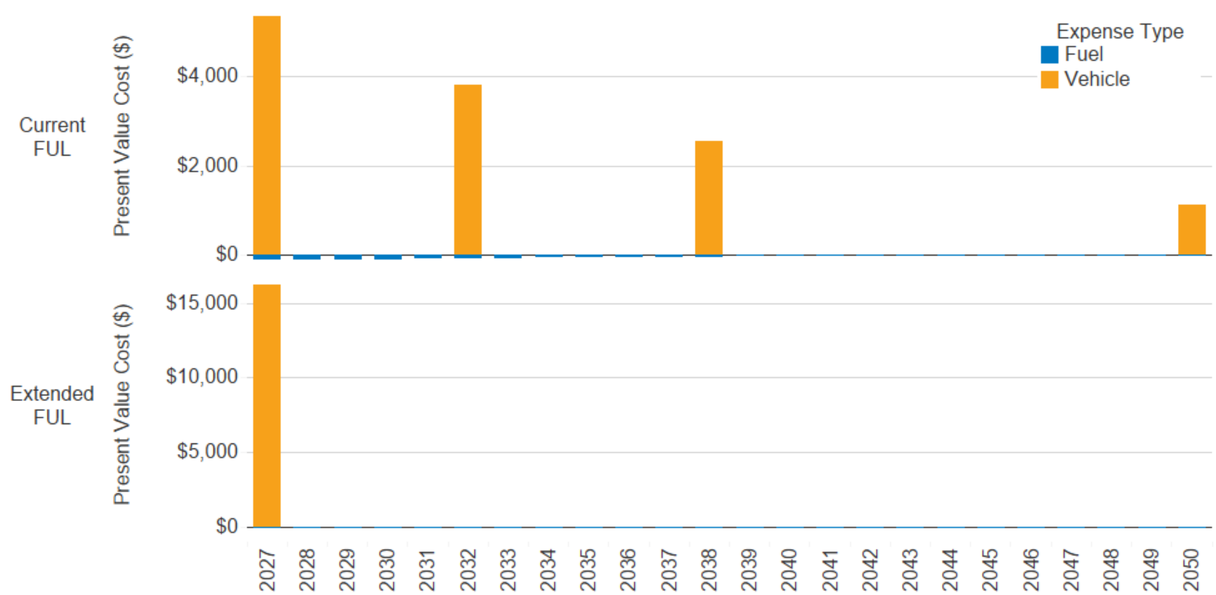

Figure 11. Annual present value cost for a T6 OOS small 6–7-L diesel engine designed for current full useful life (110,000 miles; top) and extended full useful life (550,000 miles; bottom) for MY 2027 in the South Coast Air Basin with a 2.5% increase in DEF consumption, a discount rate of 7%, and national manufacturing volumes

The previous two plots assume that replacement costs are incurred to the owner immediately upon termination of full useful life. In practice, full useful life might be extended by routine maintenance.[4] As a result, Figure 10 and Figure 11 likely represent the upper bound on actual life-cycle costs. Statistical analysis of failure rates combined with data on aftertreatment technology operating and maintenance costs could give a more accurate depiction of life-cycle costs. However, such data were not available for these potential future systems.

To explore the full useful life replacement assumption, the life-cycle costs of a vehicle can be compared assuming either no replacements are completed after vehicle mileage exceeds the aftertreatment's maximum useful life or that replacements are completed. The lower bound on life-cycle costs is set by the condition in which no replacements or maintenance are performed on the aftertreatment package regardless of vehicle mileage. This is unlikely for the current full useful life design but could be realistic for an extended full useful life scenario in which the full useful life of the aftertreatment technology is met near the end of life of the entire truck.

Figure 12 shows total present value cost for the T7 Tractor and T6 OOS small diesel engines as a function of the aftertreatment package's maximum useful life. The orange markers represent the upper-cost bound that assumes the aftertreatment package will be replaced after the vehicle mileage exceeds the maximum useful life. The blue markers reflect the lower-cost bound of no aftertreatment package replacements over the vehicle lifetime. This analysis assumes linear increments in aftertreatment cost as the designed full useful life increases from current to extended. The actual total present value cost lies somewhere between these two bounds, which are typically less than ~$5,000–$7,000 but depend on the vehicle being evaluated. As the aftertreatment package maximum useful life increases, the spread between the two conditions (orange and blue markers) typically decreases as the number of replacements decreases to zero over the lifetime of the vehicle.

Interestingly, for the T7 Tractor, designing for 75% of extended FUL is slightly more expensive than designing for 100% of extended FUL, as the one replacement that would be necessary in 2047 costs more than the incremental step in upfront cost associated with a 25% longer FUL. However, it is unlikely that the truck owner will replace the entire aftertreatment system that close to the end of life, indicating that the true cost is likely lower than the value estimated here.

[4] It should be noted that rather than incurring the replacement cost at the end of the full useful life, one could amortize those costs throughout each year of the vehicle's operation. This would effectively add incremental routine maintenance for each year and the cost would be mathematically equivalent to the end-of-full-useful-life assumption calculated here. The true incremental lifetime repair cost depends on the expected failure rates for these new aftertreatment packages which were not obtained within this study.

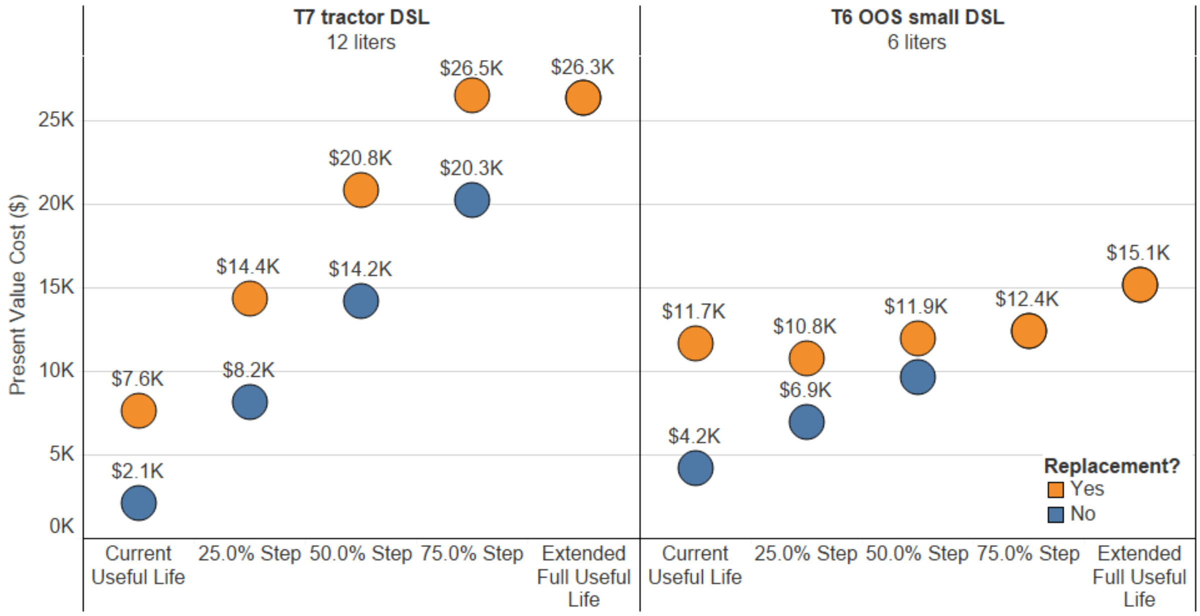

Figure 12. Total present value cost for the T7 Tractor and T6 OOS small vehicles with diesel engine aftertreatment technology as a function of incremental steps between current FUL and extended FUL for two scenarios: replacements at end of FUL (orange) and no replacements (blue)

Because aftertreatment package repair costs are either paid by the vehicle owner or the vehicle manufacturer through the warranty (if applicable), one may expect the higher upfront cost incurred to the vehicle owner for an aftertreatment package with extended full useful life and extended warranty to be offset by the aftertreatment repair cost savings over the life of the vehicle. CARB staff made this assumption when estimating costs for CARB's 2018 Step 1 warranty rulemaking, and CARB's Initial Statement of Reasons (staff report) for this rulemaking (CARB 2018a) assumes that the cost of the warranty packages is equivalent to the lifetime repair savings that the vehicle owner would realize.

The incremental upfront purchase cost that one could estimate based on the survey responses for extended FUL and warranty, and CA-only volumes, as described in Section 1.4.4, would be significantly higher than the repair cost savings that vehicle owners would realize. However, as described more fully in Section 1.4.5, the total incremental costs are dominated by the warranty incremental costs which were based on an extremely small sample size, which may be biased high because of the OEMs' uncertainty regarding covering warranty for unfamiliar technology and much longer useful lives than today's useful lives. These warranty costs may be interpreted to represent "worst case" due to these uncertainties.

While NREL does not know the method used by each OEM to determine their incremental warranty cost estimates and it is beyond the scope of this study to evaluate them in detail, a few additional potential reasons for the vehicle owner upfront costs (driven by the high warranty costs) being higher than the lifetime marginal repair savings could include:

- **Failure uncertainty** – Because the OEMs will not perfectly estimate the probability of failure for their aftertreatment packages, they may charge more than needed initially to ensure they have enough capital to cover any future liabilities. This would be an amount in excess of what the vehicle owners would actually incur but would be expected to decrease over time as the failure rates on new technologies become known with more certainty.

- **Cost of capital** – The OEMs have higher costs of capital than individual vehicle owners. Thus, their cost to reserve funding to cover future warranty liabilities would be more than what a vehicle owner would realize in lifetime repair costs on average.
- **Soft costs** – The OEMs may have embedded additional "soft" costs into the cost estimate for the extended full useful life and extended warranty to account for costs associated with warranty administration (tracking warranty data, contacting vehicle owners, processing payments), legal liability (increased legal staffing in the event of fraud), and potentially others.
- **Customer relationships** – Some manufacturers may reduce the price of the aftertreatment package with extended warranty for some customers with long-standing relationships or high volumes of purchases. These discounts may need to be offset with the "typical" aftertreatment cost, which may be reflected in the values reported from NREL's survey

The previous plots assumed medium-cost aftertreatment technologies, U.S. manufacturing volumes, up to a 1.25% improvement in fuel economy, a 2.5% increase in DEF consumption, and vehicle sales/operation in the South Coast Air Basin region. The next series of plots illustrates some sensitivity of present value cost to some of these assumptions.

Figure 13 shows present value cost of the T7 Tractor and T6 OOS small diesel trucks for the three aftertreatment cost scenarios presented in Task 1 for current full useful life. This graphic suggests that for a T7 Tractor with a 12–13-L diesel engine with current FUL, the present value cost could be ~42% lower or ~65% higher than the average, depending on which aftertreatment technology cost is realized. For the T6 OOS small truck with a 6–7-L diesel engine, the cost could potentially be 57% lower or 74% higher.

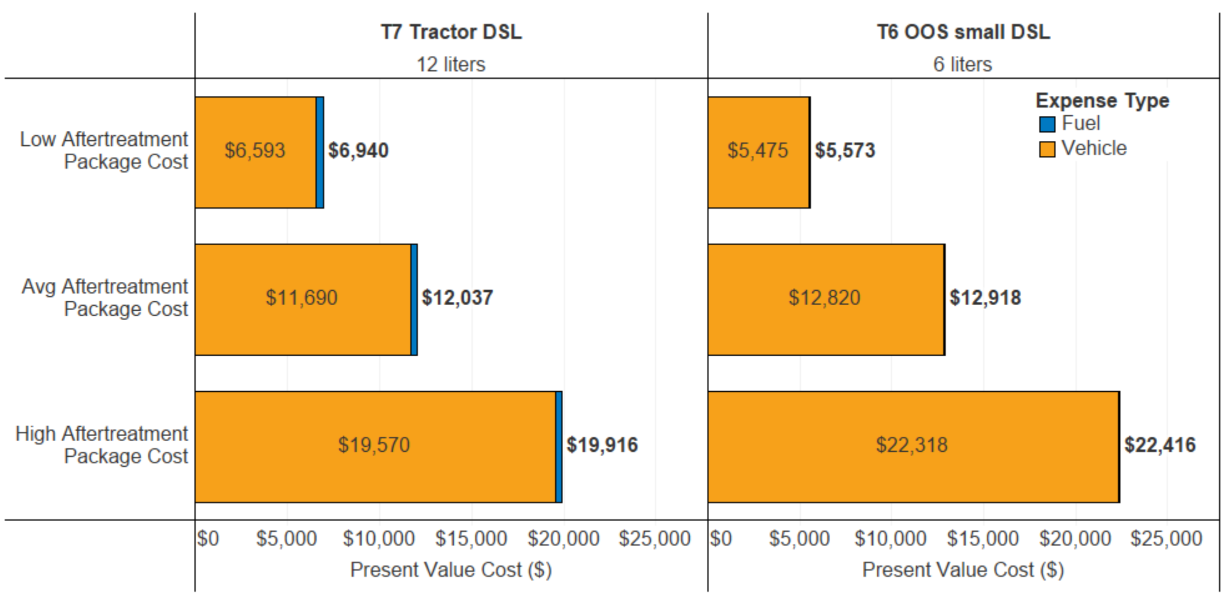

Figure 13. Present value cost for different Class 6 and Class 8 diesel engine aftertreatment technologies with current full useful life

Figure 14 shows present value cost for different aftertreatment technologies with extended full useful life. For this condition, the T6 OOS small truck with a 6–7-L diesel engine could have a life-cycle cost 12% lower or higher. For the T7 Tractor with a 12–13-L diesel engine, the range in present value cost spans 60% lower or 63% higher, about the average aftertreatment cost technology present value.

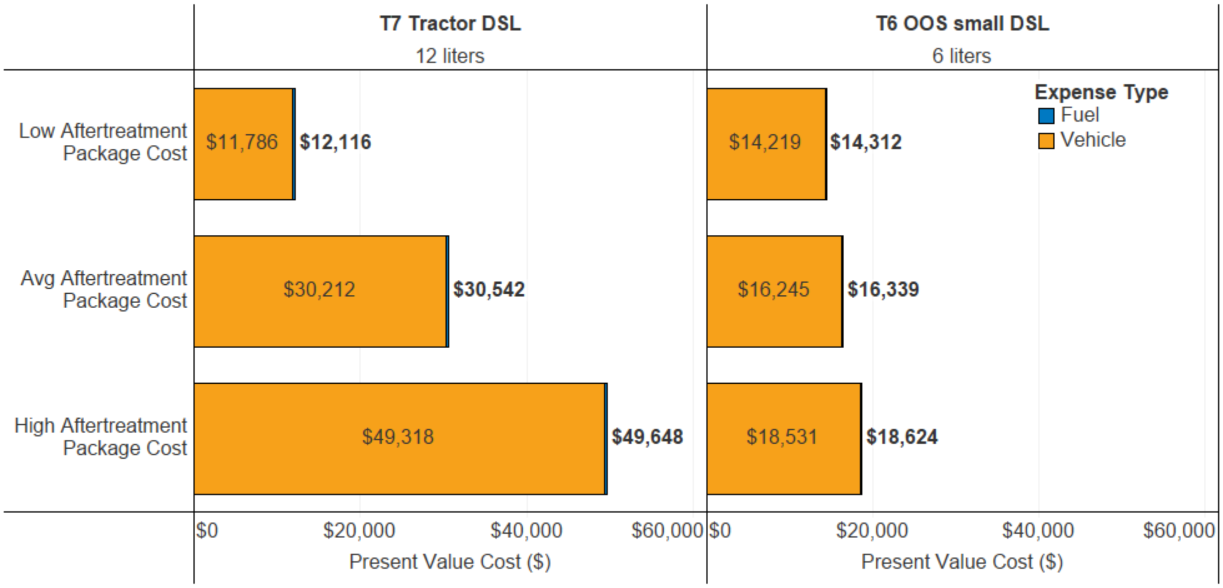

Figure 14. Present value cost for different Class 6 and Class 8 diesel engine aftertreatment technologies with extended full useful life

Figure 15 shows the present value cost for the T7 Tractor with a 12–13-L diesel engine aftertreatment technology manufactured at California and national volumes for current full useful life. No OEM data were available for California manufacturing volumes for extended full useful life. However, this figure suggests that reducing manufacturing volumes to California scales could increase the present value cost by a factor of approximately four to five.

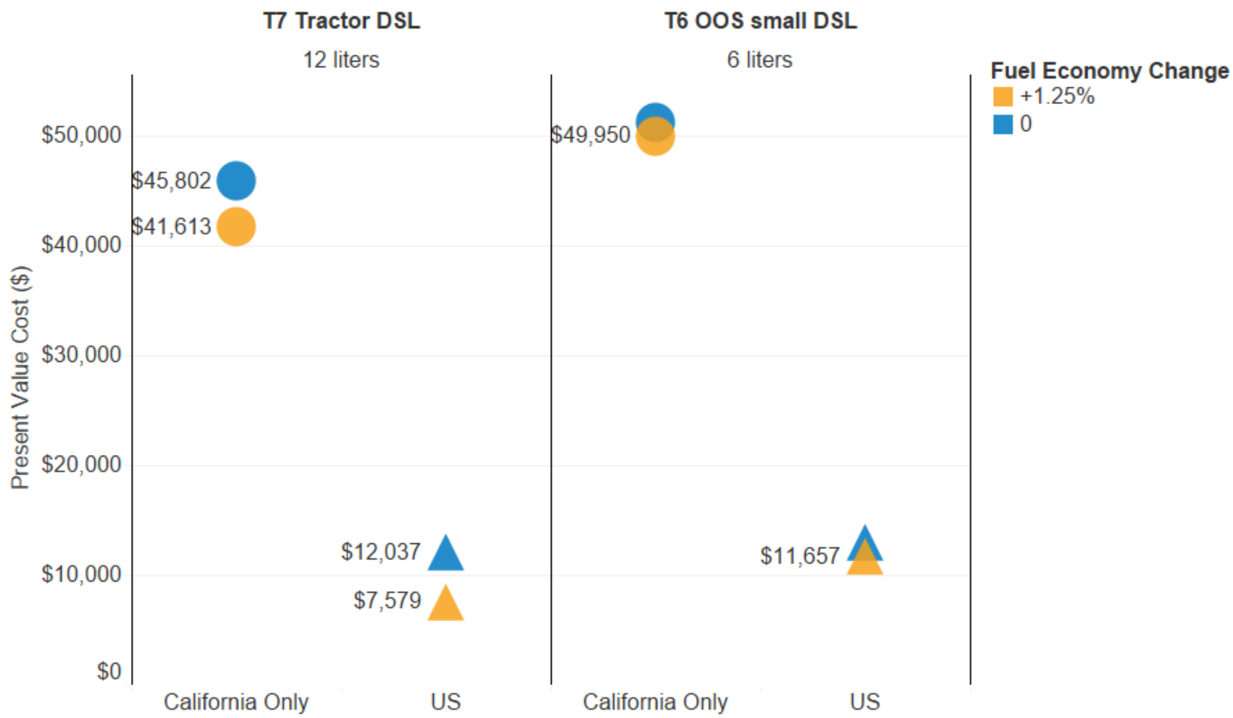

Figure 15. Present value cost for the T7 Tractor and T6 OOS small trucks with diesel engines designed for current full useful life at both California and national manufacturing volumes

Figure 16 and Figure 17 show present value cost for the T7 Tractor and T6 OOS small trucks with diesel engine aftertreatment technologies as a function of the CA Vision model-defined region for current and extended full useful life, respectively. In both cases, regional life-cycle differences are very small—generally less than ~$100. While vehicle miles traveled is dependent on the region the truck operates in, these differences are small across regions. This leads to the conclusion that regional differences in life-cycle costs are not an important factor in the life-cycle cost assessment.

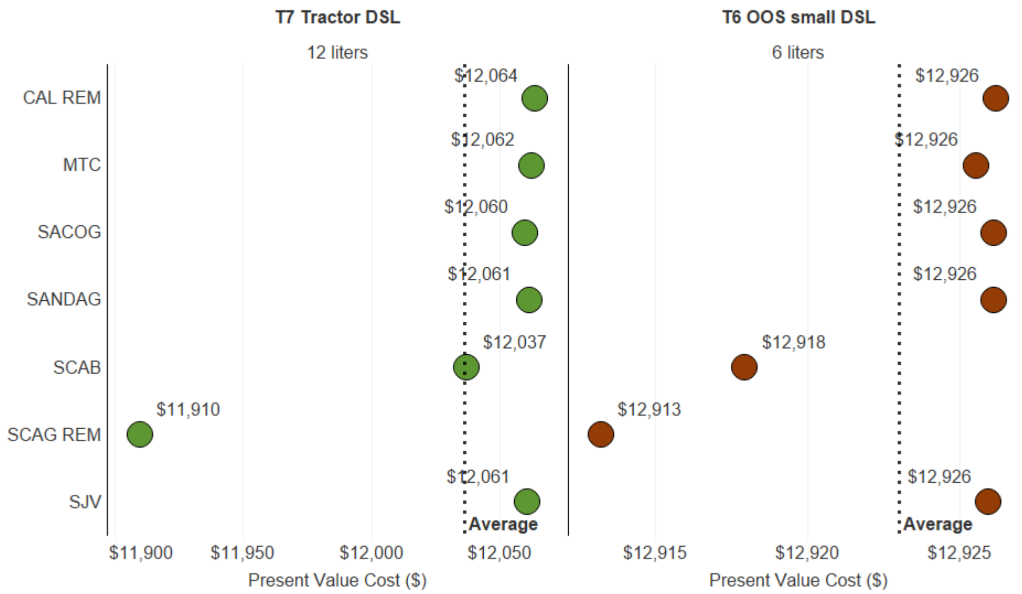

Figure 16. Present value cost for the T7 Tractor and T7 OOS small trucks with diesel engine aftertreatment technologies designed for current FUL as a function of region

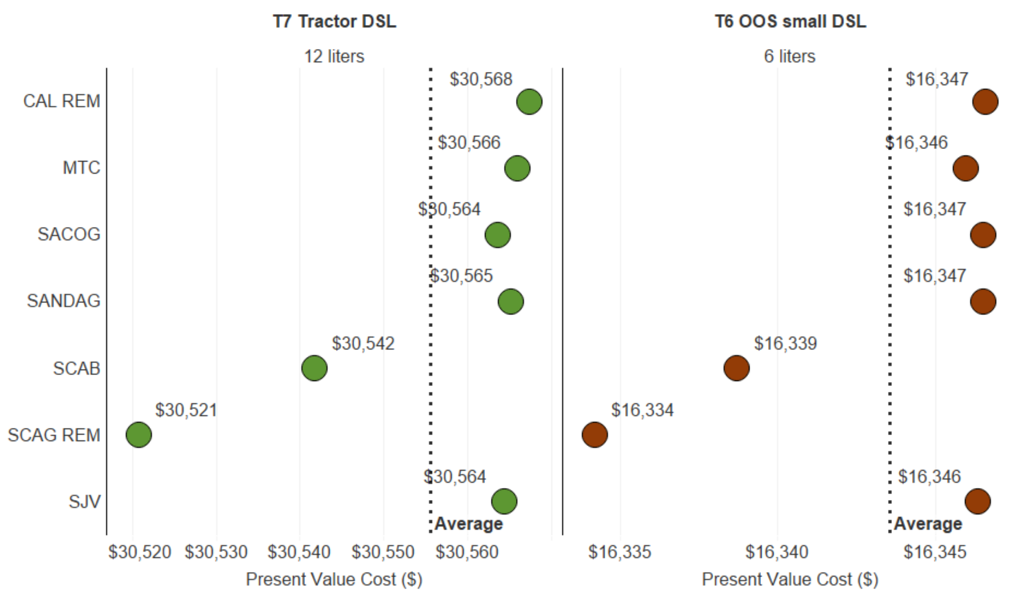

Figure 17. Present value cost for the T7 Tractor and T7 OOS small trucks with diesel engine aftertreatment technologies designed for extended FUL and warranty as a function of region

2.4.2 Scenario Analysis Results

This section presents results from a cost analysis of the three different cost scenarios depicted in Table 23. The scenario analysis results are summarized for the three different metrics discussed in Section 2.3.1:

1. Life-cycle costs for each vehicle/displacement/fuel/vocation/region combination

2. A vehicle sales weighted-average life-cycle cost across all vehicle/displacement/fuel/vocation/region combinations
3. A life-cycle cost across the full California fleet.

2.4.2.1 Vehicle-Specific Life-Cycle Costs

The life-cycle cost was calculated for each EMFAC vehicle, engine displacement, fuel technology, EMFAC vocation, and region within each of the low-, mid-, and high-cost scenarios. This provides vehicle-specific data and can be used to demonstrate the potential life-cycle costs that could be realized for each vehicle owner.

For the low-cost scenario (defined in section 2.3.1), the resulting distribution of vehicle life-cycle costs are shown in Figure 18 for each fuel and engine displacement evaluated in this study. Each EMFAC vehicle is plotted within a density plot that shows the relative proportion of vehicle types that have the associated life-cycle cost. It should be noted that this plot does not account for the projected vehicle sales and how those may differ across vehicle types (e.g., the density shown does not reflect the number of vehicles in California that will have that cost, but rather the number of EMFAC vehicle types that have that cost).

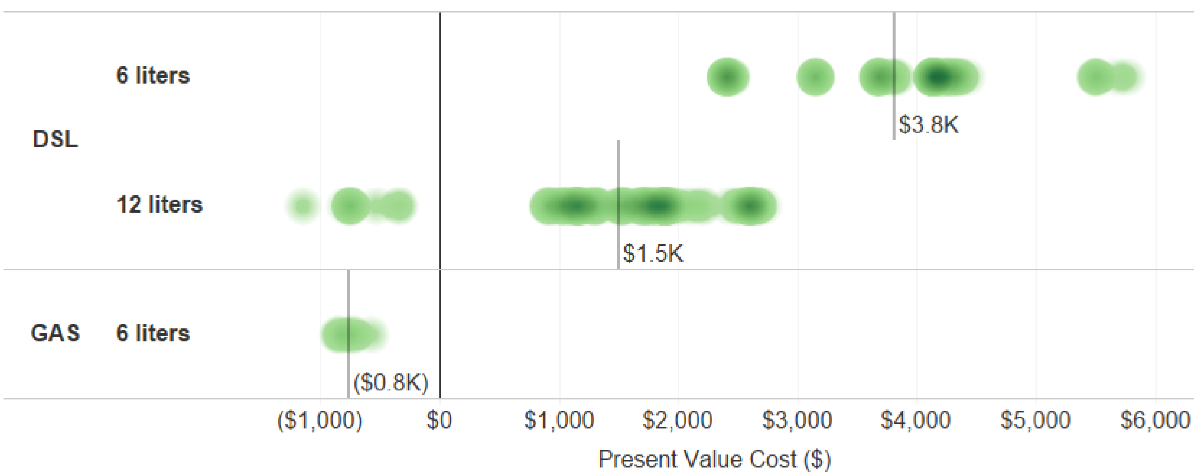

Figure 18. Present value life-cycle cost for all EMFAC vehicles in the low-cost scenario, segmented by fuel type and engine displacement (DSL = diesel, GAS = gasoline)

As seen in Figure 18, some life-cycle costs in the low-cost scenario are negative, indicating the fuel economy benefit outweighs the marginal cost of the aftertreatment package. Additionally, the spread in life-cycle costs is around ~$4,000 for both diesel engine displacements and is primarily due to the different vehicle-miles-traveled profiles across the EMFAC vehicle types. Life-cycle costs for natural gas are not shown, as there was only a single-point estimate of $3,000 for the incremental aftertreatment cost rather than low/high bounds, so natural gas was only evaluated for the mid-cost scenario.

Figure 19 shows the present value life-cycle costs for the mid-cost scenario for all three fuel types. As seen in Figure 19, there could be a significant potential spread in life-cycle costs within a single fuel type and engine displacement category. This is primarily due to the different mileage requirements for certain vehicles combined with the aftertreatment maximum useful life assumption. For the diesel engines, the potential spread in life-cycle costs could be ~$12,000 depending on which EMFAC vehicle type is evaluated. The spread is significantly lower for gasoline and natural-gas engines because there are very few vehicle types defined in EMFAC that use these fuels.

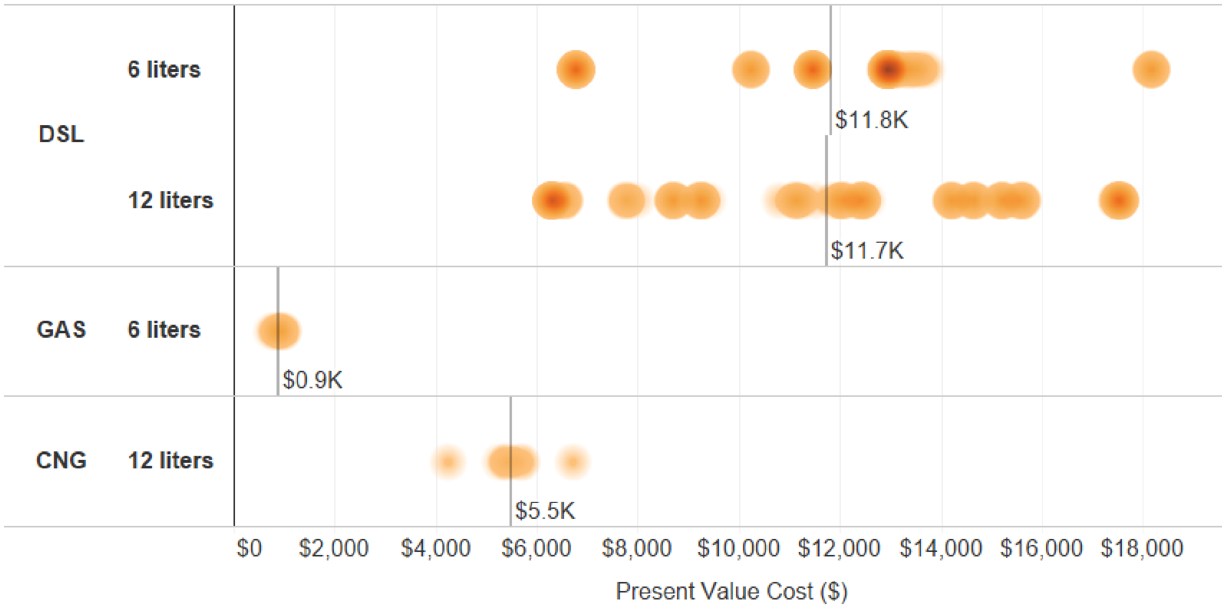

Figure 19. Present value life-cycle cost for all EMFAC vehicles in the mid-cost scenario, segmented by fuel type and engine displacement (DSL = diesel, GAS = gasoline, CNG = compressed natural gas)

The present value life-cycle costs for the high-cost scenario for diesel are shown in Figure 20. Only diesel is shown because this scenario uses the extended useful life cost data, which are not available for gasoline or natural gas. As seen in Figure 20, the life-cycle costs for a vehicle with a 6-L diesel engine in this scenario ranges from ~$18,000 to nearly $30,000. The life-cycle cost for a vehicle with a 12-L diesel engine ranges from ~$50,000 to $88,000 under this high-cost scenario. As seen previously, these higher costs are due to the high incremental cost of the aftertreatment package with both an extended maximum useful life and warranty combined with the assumption that they are replaced after the vehicle mileage exceeds the maximum useful life. The clear definition of two groups of costs in both the 6-L and 12-L engine displacements seen in Figure 20 shows that if the aftertreatment package does not need to be replaced, the life-cycle cost will be on the lower end of each range. However, if the aftertreatment package is replaced (for vehicles that travel more than the extended useful life), the life-cycle cost increases significantly to the upper end of the range.

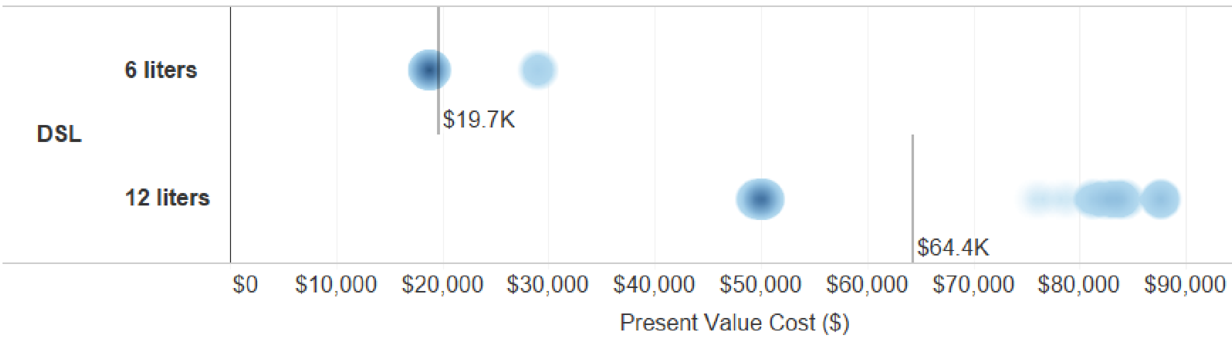

Figure 20. Present value life-cycle cost for all EMFAC vehicles in the high-cost scenario, segmented by fuel type and engine displacement (DSL = diesel)

2.4.2.2 Vehicle Sales Weighted Average Costs

As seen in Section 2.4.2.1, each EMFAC vehicle has a unique life-cycle cost. To combine these into a single, typical life-cycle cost to evaluate, a vehicle sales weighted average can be completed. Figure 21 shows the vehicle sales weighted-average results for the 6–7-L and 12–13-L engine aftertreatment technologies. The analysis shows a significant spread in potential cost between the three 12–13-L engine cases, ranging from roughly $1,500 all the way up to $71,400.[5] Most of this spread is associated with the difference between current and extended full useful life as discussed in Section 2.4.2.1. These sensitivities are discussed in the following section.

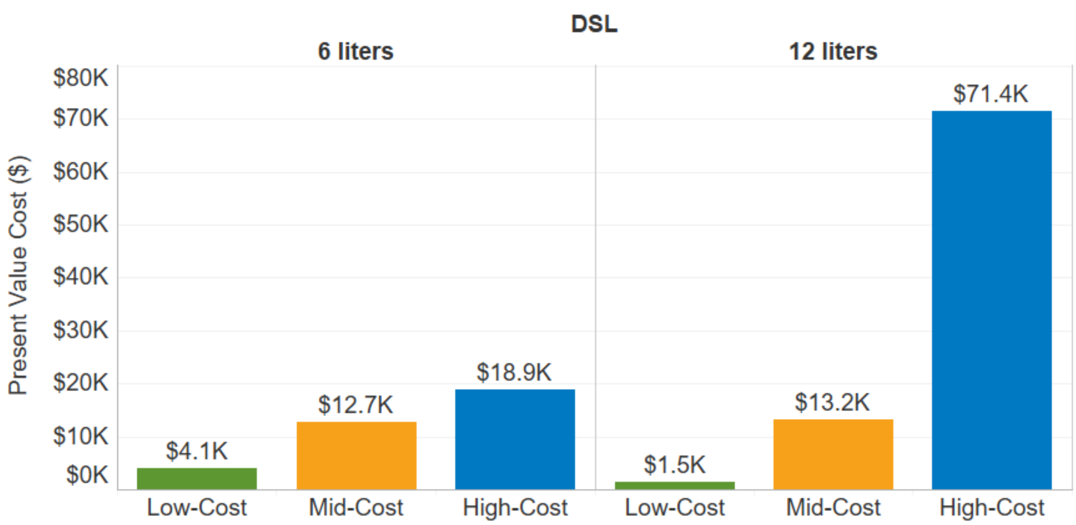

Figure 21. EMFAC vehicle sales-weighted average present value cost for 6-L and 12-L diesel engine technologies under the three cost scenarios described in Table 23

Figure 22 shows the scenario analysis for a 12-L compressed natural-gas engine and a 6-L gasoline engine. The compressed natural-gas costs are based on NREL estimates and do not reflect actual OEM data (only a single-point incremental cost of $3,000 for the aftertreatment

[5] These vehicle sales weighted averages are different than the average values shown in the figures in Section 2.4.2.1 because those averages are simple averages across EMFAC vehicle types without regard to how many of those vehicle types are actually sold in California.

package). The gasoline engine data are based on a small number of OEM estimates with limited spread in upfront cost. As a result, the differences between cases are small. Interestingly, for the low-cost scenario of the gasoline engine, the fuel economy benefits effectively cancel out the incremental aftertreatment package costs, resulting in a near-zero life-cycle cost.

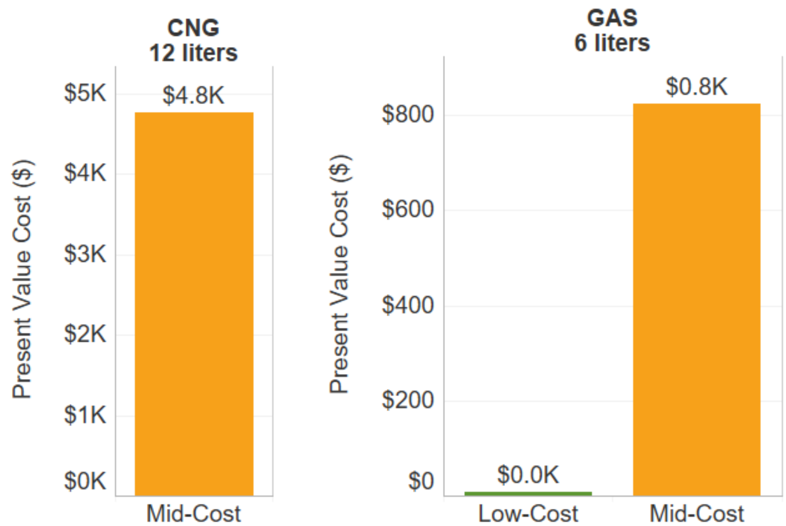

Figure 22. Scenario analysis for a 12-liter compressed natural-gas and 6-liter gasoline engine

2.4.2.3 California Fleet Life-Cycle Costs

The life-cycle cost across the full California fleet was evaluated to better understand what the total cost to all vehicle owners in California would be. As described in Section 2.3.1, this fleet calculation accounts for vehicle attrition over time because not all vehicles in the fleet will last through 2050.

Figure 23 shows the total California fleet costs for MY 2027 for each scenario evaluated in this study. The fleet costs aggregate all fuel types and engine displacements into a single cost metric. As seen in Figure 23, the total fleet life-cycle cost for the MY 2027 vehicles could range from $92 million to $1.2 billion depending on the scenario. As seen before, the large spread in costs across scenarios is primarily due to the higher incremental costs for the aftertreatment extended useful life and extended warranty, which are used in the high-cost scenario.

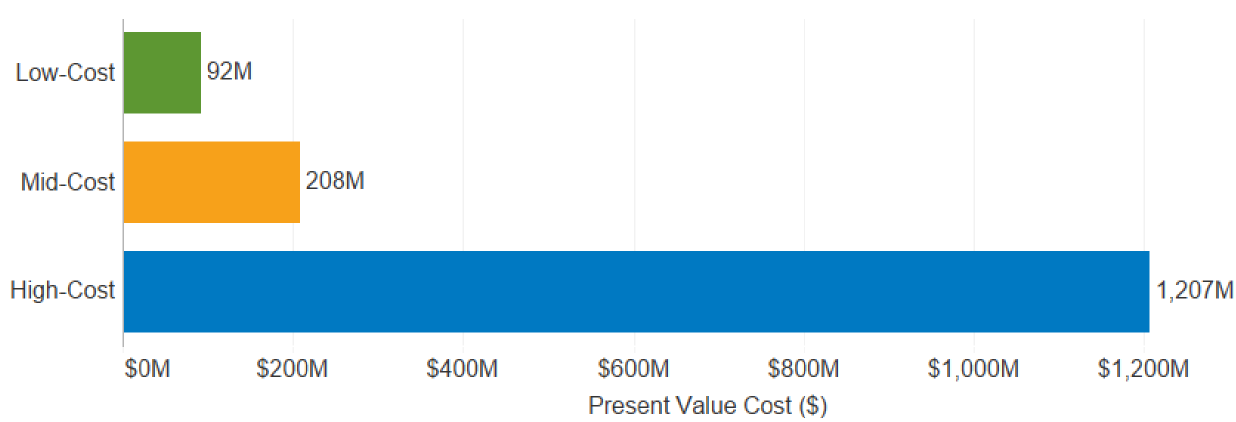

Figure 23. Total California fleet life-cycle cost for the MY 2027 vehicles for each scenario analyzed

2.4.3 Sensitivity Analysis Results

To better understand how each particular parameter assessed in this study impacts the vehicle's incremental life-cycle cost, a sensitivity analysis was completed. The vehicle sales weighted average for the mid-cost scenario (see Section 2.4.2.2 for details) was used as the starting (central) point for the sensitivity analysis.

Figure 24 shows the sensitivity analysis results for the diesel 6–7-L and 12–13-L engines. The sensitivity results are relative to the vehicle sales weighted-average costs of $12,700 and $13,200 for the 6–7-L and 12–13-L engines, respectively. For the 12-L engine, the most influential parameter is manufacturing volume, but this is based on a very limited feedback in the cost survey (Section 1.3.2) and thus was not used outside of this sensitivity analysis. Extended full useful life is the next most significant parameter, which also includes the cost associated with the extended warranty. Figure 24 shows the impact of the extended useful life along with 25% increments between the current useful life and extended useful life (linear interpolation of costs from the two data points). Each step helps illustrate how the cost increases as the full useful life increases up to the extended full useful life mileage.

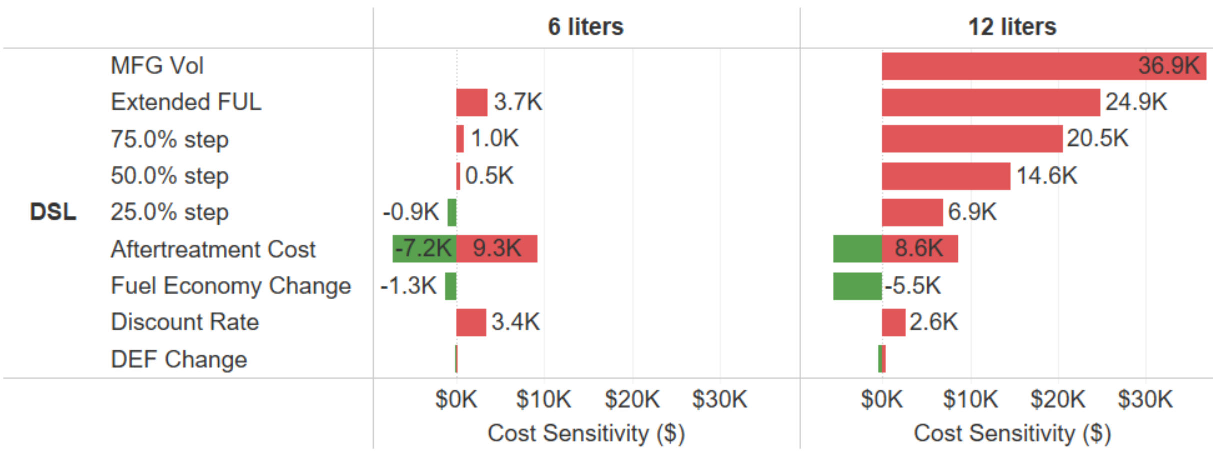

Figure 24. Sensitivity diagram for the diesel 6–7-L and 12–13-L engines relative to the mid-cost scenario

The influence of the incremental aftertreatment technology cost (Task 1 data) is relatively small compared to the aforementioned factors and has the potential to be nearly offset by fuel economy improvements. Discount rate and DEF consumption have minimal influences on the life-cycle cost. For the 6–7-L diesel engine, the aftertreatment cost (incremental cost data from Task 1) was the most influential sensitivity parameter for which data were available. Manufacturing volume may be more significant, as seen in the 12–13-L engine case, but no data were available for California-only manufacturing volume costs for the 6–7 L.

Because no cost data were available for the effect of manufacturing volume or extended useful life, the sensitivity plots for gasoline and natural gas engines have fewer parameters. Figure 25 shows the sensitivity analysis results for gasoline engines. As seen in Figure 25, the gasoline engine life-cycle cost is impacted most by the fuel economy change and incremental aftertreatment cost parameters. This indicates that if the fuel economy benefit is realized, it will likely fully offset the incremental aftertreatment costs.

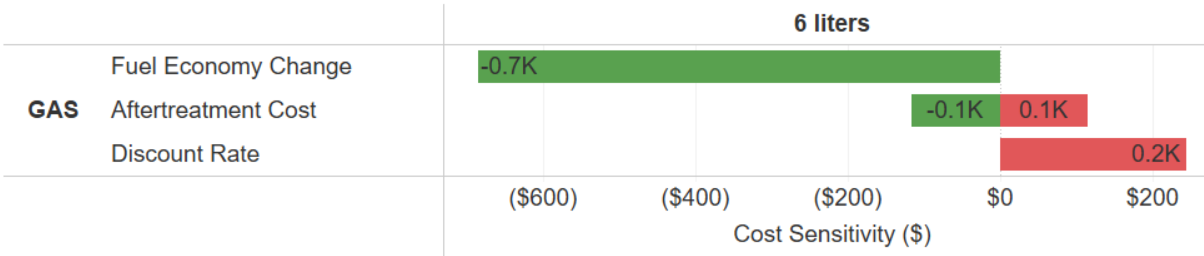

Figure 25. Sensitivity diagram for the gasoline 6-L engine relative to the mid-cost scenario

Figure 26 shows the sensitivity analysis results for the natural-gas engine. Fuel economy impacts and discount rate are approximately equal in magnitude but opposite in the direction of their influence.

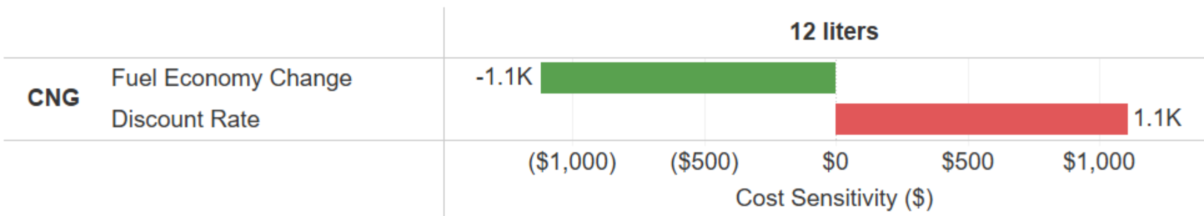

Figure 26. Sensitivity diagram for the natural-gas 12-L engine relative to the mid-cost scenario

2.5 Life-Cycle Cost Analysis Summary and Conclusions

The life-cycle cost analysis seeks to incorporate all direct and indirect incremental costs associated with the different engine aftertreatment technologies over the life of the vehicle. Three scenarios were defined and evaluated to estimate the life-cycle cost across vehicles in California under different conditions.

The scenario results suggest that the life-cycle cost incurred to each vehicle owner depends significantly on the vehicle type and scenario evaluated. Within a given scenario, the spread in life-cycle costs incurred ranges from $4,000 in the low-cost scenario up to nearly $40,000 in the high-cost scenario. Drilling down to the specific EMFAC vehicle definitions (e.g., T7 Tractor), the incremental replacement costs and potential cost savings associated with improved engine fuel economy are two dominant parameters. Because each vehicle has a different mileage profile over its lifetime, the replacement costs and fuel economy savings can vary substantially between vehicles. For example, extending the aftertreatment package's full useful life from current mileages to proposed mileages has the potential to significantly reduce, if not eliminate, the need for aftertreatment technology replacements through 2050 for some vehicles, but not others. Additionally, this extension results in little, if any, reduction in present value cost for the 6–7-L diesel engines and increases present value cost substantially for the 12–13-L diesel engines.

The scenario results also showed that the total California fleet life-cycle costs for the MY 2027 vehicles could be between $92 million and $1.2 billion depending on the scenario realized. Again, the largest factor differentiating scenarios was whether the current or extended full useful life costs were used.

Next, the vehicle sales weighted-average costs provide an approximate, representative per-vehicle life-cycle cost for each scenario. For the mid-cost scenario, the life-cycle cost could be $12,700 and $13,200 for the diesel 6–7-L and 12–13-L engines, respectively. For the mid-cost scenario, the natural gas life-cycle cost is estimated to be $4,800 while the gasoline engine life-cycle cost is $800.

Lastly, the life-cycle cost results suggest that regional impacts across California are minimal, while manufacturing volume could have a significant impact on present value cost. Very little data were available for California-only manufacturing volumes, but the data available suggest the costs could be 4–5 times more than if a national manufacturing volume was realized.

3 Conclusions

The incremental cost analysis was constructed to bracket a range of potential incremental costs associated with achieving 0.02 g/bhp-hr NO_x emissions over certification cycles, including a new proposed LLC. Diesel engines were the primary consideration, as they comprise the majority of HD engines. Incremental cost bracketing included three diesel engine and aftertreatment technology packages, two diesel engine displacements, MY 2023 versus 2027 introduction, U.S. versus California-only implementation, and current FUL versus extended FUL and warranty. Direct and indirect incremental costs were broken down to as discrete a level as possible while maintaining data confidentiality. The calculation of incremental costs was limited by the small number of respondents. Engine OEM participation was crucial, as only they could provide estimates for indirect costs, which represented a significant portion of the total cost.

The average incremental cost for the 6–7-L diesel engines for MY 2023 with current FUL ranged from $3,685 to $5,344, but the absolute low and high bounds were between ~$2,000 and over $9,000. Extending FUL and warranty moved the average incremental costs to a range of $15,370 to $16,245, with tighter low and high bounds (constrained in part by the limited number of responses). The average incremental cost for the 12–13-L diesel engines for MY 2023 with current FUL ranged from $5,340 to $6,063, but the absolute low and high bounds were between ~$3,000 and over $10,000. Extending FUL and warranty moved the average incremental costs to a range of $28,868 to $47,042, with much wider low and high bounds (driven in part by the limited number of responses). The natural gas 12-L engine application was unable to be studied in detail, but OEM feedback anticipated that the incremental cost for natural-gas engines and aftertreatment technology is within 10% of the low-cost diesel technology package for equivalent displacement, specifically due to possibly requiring a moving average window method to assess emission compliance. The gasoline engine 6-L application was also unable to be studied in detail, but comparatively low incremental costs were estimated.

Incremental costs are largely driven by indirect costs associated with engineering research and development costs, plus warranty. Those indirect costs, in turn, are driven by production volumes and amortization.

The life-cycle cost analysis incorporates all direct and indirect incremental costs associated with the different engine aftertreatment technologies over the life of the vehicle. The life-cycle costs depend on the vehicle type (mileage), region, fuel, engine displacement, maximum useful life, fuel economy change, diesel exhaust fluid consumption change, and discount rate. The primary drivers of life-cycle cost were the incremental aftertreatment replacement costs and fuel economy benefits.

For the three scenarios evaluated (low-cost, mid-cost, high-cost), the life-cycle costs were evaluated for each EMFAC vehicle type, aggregated to a representative average, and also calculated across the vehicle fleet for the model year 2027 vehicles. The analysis showed that EMFAC vehicles can have significantly different life-cycle costs, and that spread depends on the scenario evaluated: approximately a $4,000 spread across vehicle types in the low-cost scenario, while the high-cost scenario had nearly a $40,000 difference. This large spread was found to be due to the number of aftertreatment package replacements needed throughout the vehicle lifetime. The aggregated, representative average life-cycle costs for the mid-cost scenario were estimated to be $12,700 for the 6–7-L diesel engine, $13,200 for the 12–13-L diesel engine, $4,800 for the 12-L natural-gas engine, and $800 for the 6-L gasoline engine. The total life-cycle cost to California vehicle owners for the model year 2027 vehicles was estimated to range between $92 million and $1.2 billion depending on the scenario (low-cost or high-cost) realized.

The sensitivity analysis indicated that the manufacturing volume may be the most important parameter impacting the life-cycle cost; however, limited data were received from the external stakeholders surveyed. The next most important parameter was the assumption of extended useful life and extended warranty, as the increase in aftertreatment lifetime may not exceed the vehicle's travel requirement, which results in larger replacement costs over the vehicle's life. The aftertreatment cost bound (low/high error bars on the incremental cost data), fuel economy improvement, and discount rate were found to have a moderate impact on the life-cycle cost. Lastly, the region and DEF consumption change were found to have minimal influence on the life-cycle cost.

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Appendix A. Selected Results for Specific EMFAC Vehicles of Interest to CARB

In addition to the life-cycle costs presented in this report, the California Air Resources Board (CARB) indicated a specific interest in the following EMission FACtor (EMFAC) vehicles (CARB 2018b):

Table A1. EMFAC Vehicles of Interest to CARB

EMFAC Vehicle	EMFAC Description (GVWR = Gross Vehicle Weight Rating)
T7 Tractor	Heavy Heavy-Duty Diesel Tractor Truck
T7 Single	Heavy Heavy-Duty Diesel Single Unit Truck
T7 POLA	Heavy Heavy-Duty Diesel Drayage Truck near South Coast
T6 OOS Heavy	Medium Heavy-Duty Diesel Out-of-State (OOS) Truck with GVWR > 26,000 lb
T6 OOS Small	Medium Heavy-Duty Diesel Out-of-State Truck with GVWR ≤ 26,000 lb

Per the CA Vision 2.1 model, the vehicle-miles-traveled profiles for these vehicles with a model year (MY) of 2027 in the South Coast Air Basin (SCAB) region are shown in Figure A1.

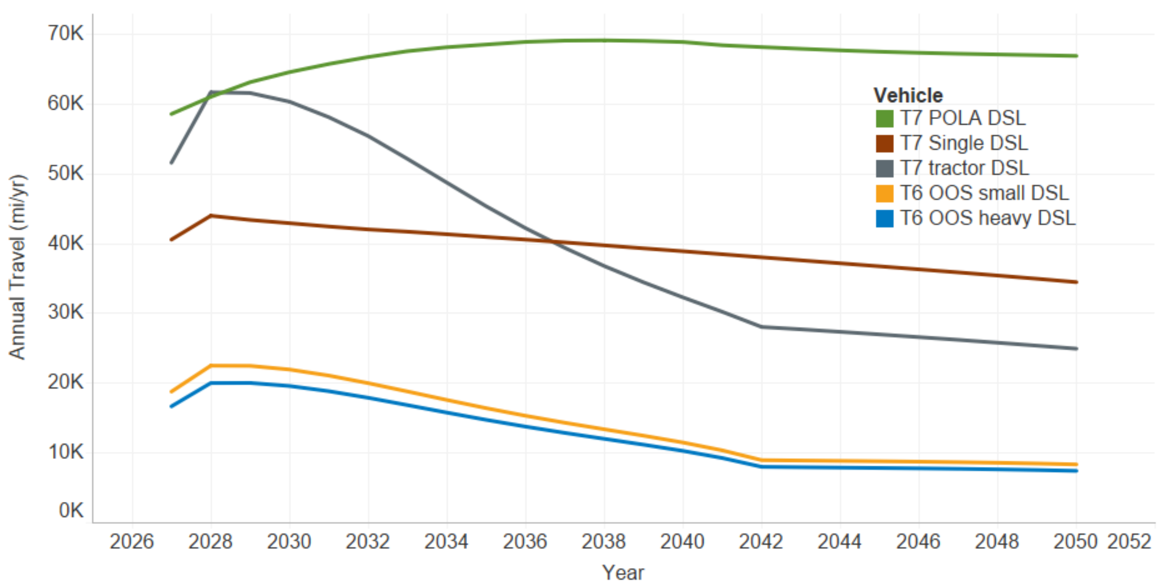

Figure A1. Selected EMFAC vehicle miles traveled for MY 2027 in the SCAB region

For these vehicles, the life-cycle costs for each scenario evaluated (low-cost, mid-cost, and high-cost) are shown in the following figures. Figure A2 shows the life-cycle costs for the low-cost scenario, Figure A3 shows the results for the mid-cost scenario, and Figure A4 shows the results for the high-cost scenario. These results are aggregated for each vehicle, which accounts for the costs incurred from the aftertreatment package as well as any potential fuel economy benefit associated with the scenario.

Of note, the individual vehicle life-cycle cost results are very close to the representative life-cycle costs estimated using the vehicle sales weighted average shown in Figure 21 in Section 2.4.2.2.

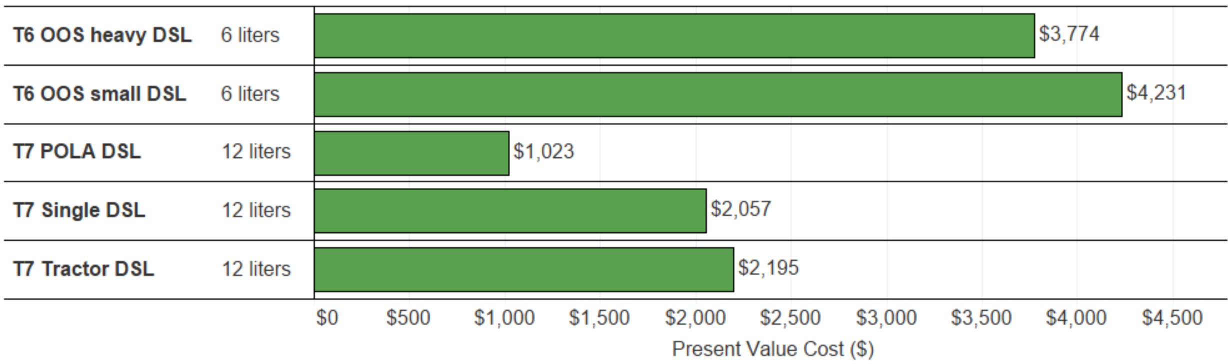

Figure A2. Present value life-cycle cost for selected EMFAC vehicles (MY 2027 in the SCAB region) for the low-cost scenario

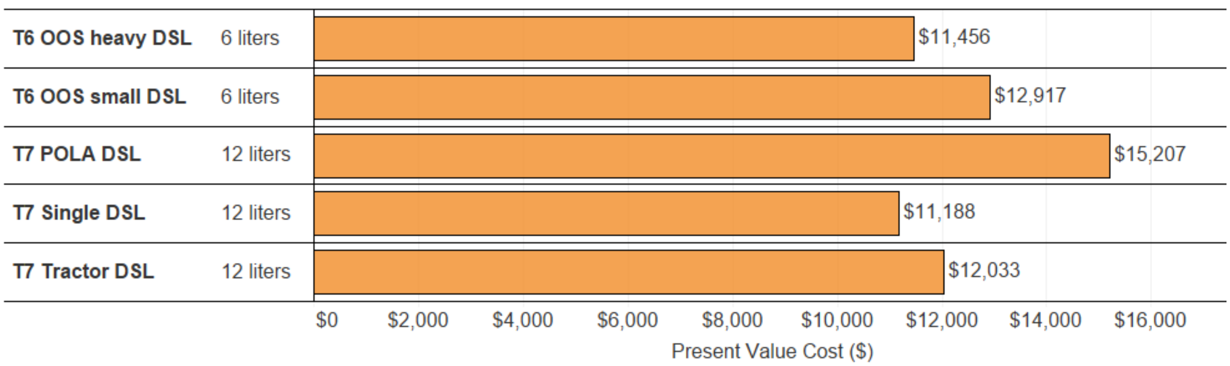

Figure A3. Present value life-cycle cost for selected EMFAC vehicles (MY 2027 in the SCAB region) for the mid-cost scenario

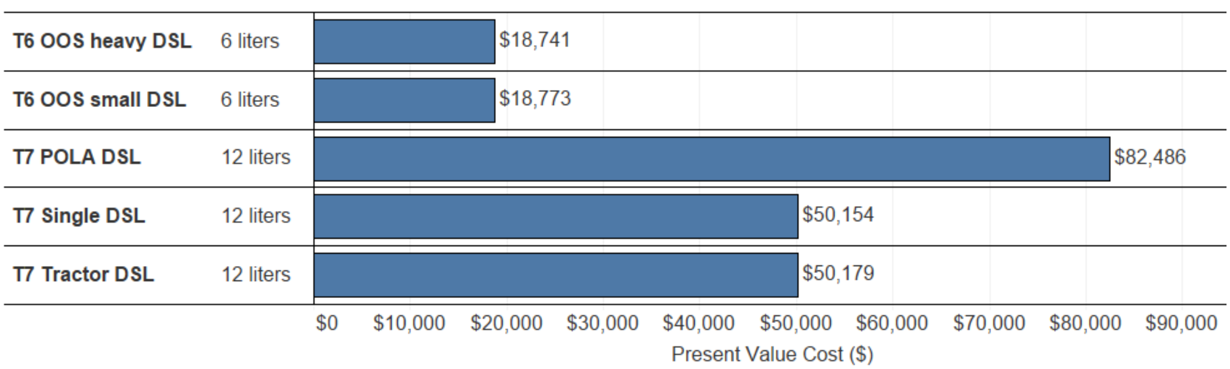

Figure A4. Present value life-cycle cost for selected EMFAC vehicles (MY 2027 in the SCAB region) for the high-cost scenario

Appendix B. EMFAC Vehicle Disaggregation

The EMFAC vehicles needed to be broken down into the appropriate fuel and engine displacement categories. The IHS Markit (formerly Polk) Department of Motor Vehicles registration database was used to disaggregate the EMFAC vehicles. The same disaggregation was used for each CA Vision region and the first few results are summarized in Table B1, while the full table is provided in a separate file.

Table B1. EMFAC Vehicle Disaggregation Results

EMFAC 2011 Vehicle	Displacement (L)	GVWR Class	Fraction (veh/veh)
MH	12	7	0.6008
MH	15	7	0.3992
T6 Ag	6	4	0.3302
T6 Ag	9	4	0.0063
T6 Ag	6	5	0.1554
T6 Ag	9	5	0.0095
T6 Ag	6	6	0.1936
T6 Ag	9	6	0.0995
T6 Ag	6	7	0.0975
T6 Ag	9	7	0.1081
T6 CAIRP heavy	6	7	0.4743
T6 CAIRP heavy	9	7	0.5257
T6 CAIRP small	6	4	0.4156
T6 CAIRP small	9	4	0.0079
T6 CAIRP small	6	5	0.1956
T6 CAIRP small	9	5	0.0119
T6 CAIRP small	6	6	0.2437
T6 CAIRP small	9	6	0.1253
T6 instate construction heavy	6	7	0.4743
T6 instate construction heavy	9	7	0.5257
T6 instate construction small	6	4	0.4156
T6 instate construction small	9	4	0.0079
T6 instate construction small	6	5	0.1956

EMFAC 2011 Vehicle	Displacement (L)	GVWR Class	Fraction (veh/veh)
T6 instate construction small	9	5	0.0119
T6 instate construction small	6	6	0.2437
T6 instate construction small	9	6	0.1253
T6 instate heavy	6	7	0.4743
T6 instate heavy	9	7	0.5257
T6 instate small	6	4	0.4156
T6 instate small	9	4	0.0079
T6 instate small	6	5	0.1956
T6 instate small	9	5	0.0119
T6 instate small	6	6	0.2437
T6 instate small	9	6	0.1253

Exhibit D

RESPONSE TO STANDARDIZED REGULATORY IMPACT ANALYSIS FOR PROPOSED CARB HEAVY-DUTY EMISSIONS REGULATIONS

PREPARED FOR:

TRUCK & ENGINE MANUFACTURERS ASSOCIATION

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CHICAGO, ILLINOIS ▪ 60606

July 31, 2020

ACT Research Company (ACTR) appreciates the opportunity to submit the following comments in response to the Standardized Regulatory Impact Assessment (SRIA) associated with the *Proposed Heavy-Duty Engine and Vehicle Omnibus Regulation and Associated Amendment* that the California Air Resources Board published on June 23, 2020, which was amended on July 10, 2020.

ACTR is a boutique research firm focused on surface transportation dynamics and commercial vehicle demand. ACTR's customers include leading MD and HD vehicle manufacturers, the commercial vehicle industry's supply base, investors in transportation and machinery companies, transportation companies, and other groups of stakeholders who need to understand the impact of economic activity on trucking industry profitability, and by extension, demand for medium- and heavy-duty on-highway vehicles.

ACTR's decision to provide comments on the CARB SRIA relates to a study the company undertook at the behest of the Engine Manufacturers Association (EMA) in early 2020. The resulting study was an upfront cost and total cost of ownership (TCO) analysis relating to the impact of the California Air Resource Board's (CARB) proposed Omnibus Low-NOx Regulation and the U.S. Environmental Protection Agency's (EPA) advanced notice of proposed rulemaking (ANPRM) published in the Federal Register on January 21, 2020, entitled "Control of Air Pollution from New Motor Vehicles: Heavy-Duty Engine Standards." Given the similarities in the CARB and EPA proposals surrounding NOx standards and warranty extensions, we believe our analysis adds to the discourse surrounding CARB's proposed Regulation.

ACTR has been and will continue to be a supporter of CARB and EPA efforts to improve air quality. We applaud the 99% and 98% reductions in particulates and NOx, respectively, that have occurred over the past quarter-century. And in contrast to the costly final mandates that reduced PM and NOx, the more recent GHG Phase 1 and Phase 2 (to date) regulations have pushed industry stakeholders to deliver tremendous advances in on-highway fuel economy at nominal cost, thereby benefitting both the environment and the buyers of new commercial vehicles.

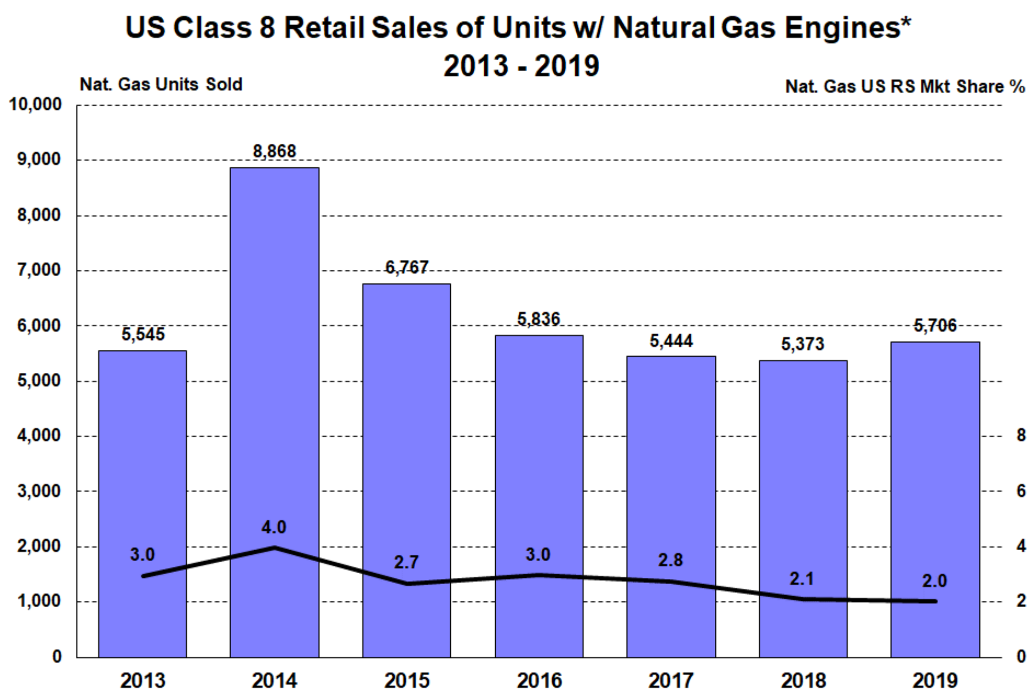
While we at ACTR recognize the need to continue reducing emissions levels from all sources, we also believe that accuracy in accounting is needed for regulators to make the best decisions possible in plotting the way forward on emissions regulations. It is in that spirit that we believe a better accounting needs to be made in regard to CARB's current proposal to improve air quality. Based on our modeled conclusions, it is ACTR's opinion that CARB's accounting for the cost impact of the proposed Omnibus Regulation is incomplete on several fronts, including:

1) Market sizing
2) Accounting for Pre-buy/No-buy impacts
3) R&D accounting
4) Useful life and warranty accounting

Over the course of this submission, ACTR will lay out where we believe the accounting as presented in the SRIA fails to capture the true costs of CARB's regulatory proposal. If our analysis is correct, the CARB regulation is likely to cause significant market disruptions as trucking companies actively work to minimize their exposure to new vehicles that would leave them at an operating cost disadvantage compared to their competition.

Market Size and Structure. Although we do not have a fully transparent understanding of the sales projections driven by CARB's EMFAC model, we disagree with the use of 2013 as the year from which to draw conclusions about the current and future commercial vehicle market size and structure.

- Based on OEM data, we estimate natural gas vehicles had a Class 8 market share nationally of 3%-4% in 2013-2014, which has since trended down to 2% in the past two years (see chart). Of course, we recognize that California represents an out-sized proportion of natural gas truck sales, but in the SRIA, CARB assumes HD Otto-cycle engines, including natural gas, were 43.6% of the heavy heavy-duty (Class 8) market in 2013. The market share has fallen considerably in the years since, and a more current weighting of the Class 8 market would increase the diesel units subject to the low-NOx standards, which would increase overall costs in the resultant calculations.



* Transit bus data estimated. All other data as reported by OEMs to ACT Research.
Source: ACT Research Co., LLC

- We agree with CARB's earlier sales volume methodology which took into account the smaller market outlook resulting from the implementation of the Advanced Clean Truck (ACT) Regulation. But we disagree with the changes made, as recommended by the California Department of Finance (page IX-7), to adhere to a legal baseline that does not include the mandated zero-emissions vehicles under the ACT Regulation. That may have mixed implications for cost outputs, but suggests per-unit costs are understated. The cost study conducted by ACTR used the smaller market size resulting from the ACT Regulation, which lowered overall costs but raised per-unit costs, though the targets in the ACT Regulation have been raised even further since our study was conducted.

- CARB's SRIA Does not Consider the Likelihood of Pre-buy/No-buy. We agree with the need to include increased DEF consumption costs and financing costs, as CARB did in the SRIA. Costs to truckers were not included in ACTR's manufacturing cost analysis, but were included in our Pre-buy/No-buy analysis. In our view, the largest blind-spot in CARB's SRIA is the failure to consider the industry's anticipated avoidance-response to the prospect of costly and risky new emission-control regulations.

- The higher DEF consumption rate is just one of several additional cost factors that should be considered for the trucking industry, separate from manufacturing costs. Those include the taxes on the higher cost of a truck, which is a 12% Federal Excise Tax plus state taxes, and the costs to insure the more expensive vehicles, typically 5% of the purchase price per year.

- As a result, for every $1 increase in the purchase price of the vehicle, the equipment costs to the operator are likely to rise by $1.40 - $1.75, depending on the assumptions about the operating lifecycle. Hence, we think DEF costs are a very small fraction of the non-manufacturing costs of the Omnibus Low-NOx rulemaking proposal, which would be borne by the trucking industry.
    - In the cost study ACT Research performed for the EMA, we considered how the foregoing costs plus the higher base vehicle prices would impact the trucking industry. Instead of arguing about assumptions, we took a macroeconomic approach.

- We concluded that in this highly fragmented and cyclical industry, which is largely dependent upon market freight rates, a significant pre-buy is likely, with elevated demand for equipment built before the Omnibus regulations take effect. Trucking is a low-margin industry which abhors risk. Considerable historical precedent shows any significant price increase and technological change likely will drive a pre-buy in this industry. This will add excess capacity to the market and drive down freight rates, with a material adverse effect on earnings for the trucking industry. We have expertise in those freight rate sensitivities through *Freight Forecast* service, and we estimate the subsequent decline in truckload rates would cost the industry between $6.5 billion and $8.6 billion in the 2027-2028 timeframe. Further, the combination of the effects of the pre-buy and the cost of lower freight rates would materially reduce the industry's ability and willingness to purchase new vehicles after the Omnibus regulations take effect, thereby delaying the benefits of the regulation. The significant pre-buy/no-buy impacts are missing from the CARB SRIA.

R&D. CARB's SRIA assigns minimal Research and Development (R&D) costs to the implementation of its proposals, ranging from $78-$85 per unit for Medium Heavy-Duty (MHD) vehicles to $354-$356 per unit for Heavy Heavy-Duty (HHD) vehicles (ISOR page IX-10). The underlying sales figures from CARB's EMFAC model are not clear, and the total R&D costs are not broken out in CARB's aggregate table IX-32.

- The Original Equipment Manufacturer (OEM) study conducted by ACT Research yielded an estimate of $603 million of R&D costs to meet the HHD MY2027 standards proposed for California, only modestly less than the $715 million estimated for a full nationwide program. While the core processes are unchanged regardless of whether it is a California-only or national standard, the OEMs intend to reduce the offerings available in California to achieve those modest savings.

- Based on OEM feedback that these costs would be amortized over three- to four-year product cycles, that translates to about $38,000 per unit for the HHD market beginning in MY2027. CARB's SRIA does not explain how it arrived at its significantly lower R&D figure, though we acknowledge there is significant managerial accounting discretion to extend the amortization period and lower the per unit costs. Extending the regulations to a natiowide basis reduces those per-unit costs to just under $2,800 per unit in our model, even keeping with the OEMs' three- to four-year amortization periods, which highlights the benefit of harmonized national standards over regional ones.

<u>Useful Life.</u> Producing aftertreatment systems to meet tighter standards, increasing the Useful Life (UL) of those systems, and providing a warranty on those systems are three of the distinct challenges presented by the proposed Omnibus Low-NOx regulations. CARB's assertion that increased UL is included in the Technology Costs is not realistic because, for example, Cylinder Deactivation technology is not currently commercially viable and likely will require at least one full replacement in order to meet the UL proposal.

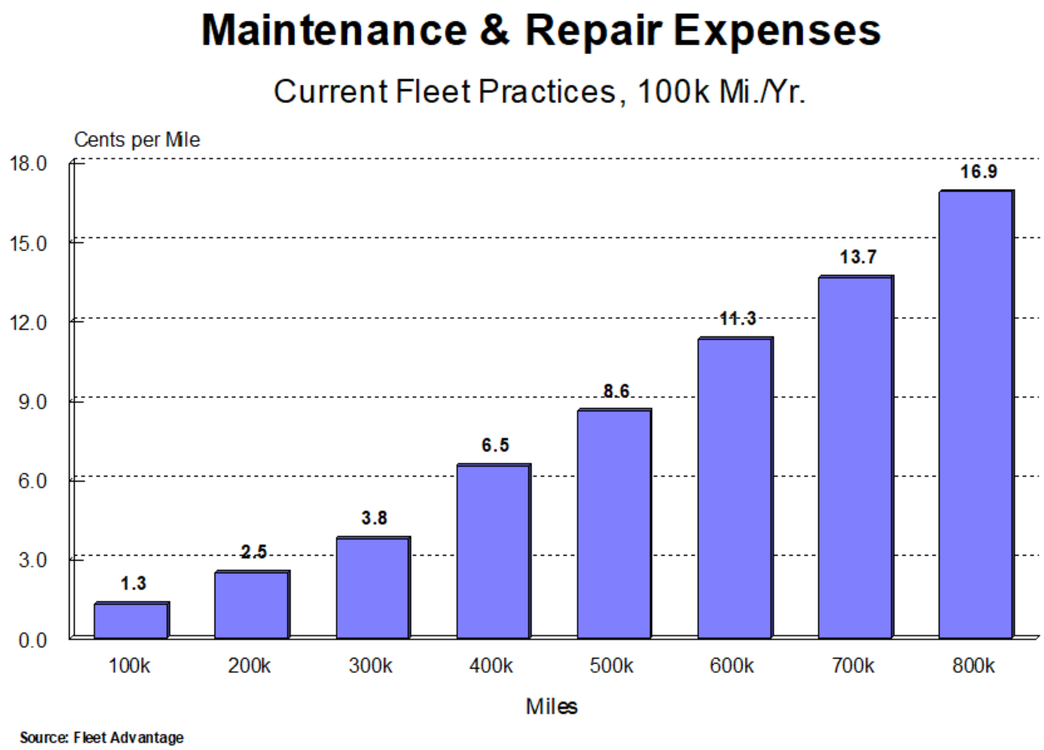
- The OEM survey conducted by ACT Research, which accounted for all major manufacturers, yielded an estimate of $176 million of indirect costs to meet the MY2027 UL provisions in the CARB regulatory proposal for Heavy Heavy-Duty (HHD) vehicles, which added $11,178 of cost per vehicle under our market sizing parameters. It also yielded a similar result for MY2031, with smaller cost figures for medium-duty vehicles. Those costs are missing from the CARB SRIA.

<u>Warranty.</u> In assigning $930 of incremental repair costs for HHD vehicles in order to extend warranties from 350,000 miles to 600,000 miles in MY2031, where no warranty data exists, CARB's warranty analysis (SRIA, page IX-19 to IX-25) materially contradicts the results of both the ACT Research and the NREL (May 2020) cost analyses. The $159 estimate for incremental repair costs beginning MY2027 for HHD vehicles also is deeply flawed, again considering the unproven nature of the new technologies expected to be employed, particularly cylinder deactivation.

- The feedback from manufacturers used as input for both the ACT Research and NREL studies is that the extended warranty provisions would effectively require the manufacturers to account for almost a full aftertreatment system replacement for every vehicle, or about $8,000 per HHD unit. NREL's average cost scenario for 12-13L engines included a $23,424 per unit incremental warranty cost, but this appears to include the extended useful life provisions as well, which we detailed separately.

- We do not agree with CARB's linear extrapolation of warranty costs into the extended warranty periods based on MY2013 data.
    - Those data represent significantly lower-cost MY2013 emissions systems, not the more costly systems envisioned in the Omnibus regulation. Thus, we believe that extrapolation methodology fails to account for the warranty costs on the added components.
    - We also believe CARB's assumption (page IX-22) "that components would continue to fail at the same rate for the duration of the lengthened warranty period" is flawed. Based on feedback from manufacturers during our survey, our experience analyzing the trucking industry, and the *Fleet Advantage* study charted below, it appears to us to be common knowledge that maintenance costs increase significantly over time. In addition, the

Southwest Research Institute (SwRI) Low-NOx Stage 3 testing program only tested the prototype engine system up to 435,000 miles (page III-7). Thus, CARB's SRIA does not include accurate UL or warranty costs.



- CARB's warranty mileage baseline is not realistic, in our view, and ignores the costs incurred by the trucking industry for extended warranties above the regulatory baseline. CARB's methodology understates warranty costs for California, and would understate warranty costs even more on a national basis where the baseline is below CARB's Step 1 baseline.
  - For MY2027, CARB assumed 40% of HHD trucks would be purchased with 500,000-mile warranties, reducing the distance to the extended 600,000-mile warranty proposal. That ignores the considerable costs some fleets pay for extended warranties and overstates current industry practice. Our research suggests that extended warranties are typically for 400,000 miles, and that the take-rate is likely less than 40%.
  - In reality, the industry-standard base warranty is 250,000 miles, and the EPA regulatory baseline is 100,000 miles. Because those are significantly lower than the 350,000-mile CARB Step 1 baseline, which will be in effect as of 2022, that is a material difference when considering extending those provisions to the national level. Incremental warranty costs per unit on a national basis from the proposed regulations would be significantly higher than the estimates in CARB's SRIA.
  - Based on CARB's assumption (however questionable) that it can calculate warranty costs linearly, and our view that the incremental warranty costs should be based on the 350,000-mile Step 1 baseline, CARB should be accruing for an incremental 250,000 miles of warranty coverage, not 190,000 miles in its analysis (adding the 40% at 500,000 miles raises the baseline to 410,000 miles). Thus, CARB's analysis misses about 24% of the increase in regulatory warranty costs.

Technology path. The direct engine and aftertreatment component cost output of $11,347 from the ACTR Study, which combined MY2024 and MY2027, was well above the comparable figure from CARB's SRIA of $6,429 ($1,611 in MY2024 and $4,818 in MY2027). The main source of difference is that the manufacturers surveyed by ACT Research did not all choose the same technology path, and so did not all choose the path laid out in CARB's proposal. Since CARB's proposals are supposed to be technology neutral, with no picking of winners or losers, an estimate of costs that considers more than one technology path is preferable in our view.

Other. We do not purport to be experts in the management of large manufacturing companies, as our expertise is primarily in data analysis and forecasting for the transportation and commercial vehicle industries. However, we question CARB's assumptions throughout the SRIA cost analysis that the important work of compliance with these Omnibus emission regulations would be relegated to a single junior engineer earning just $70 per hour. Including internal management oversight, which seems important from our perspective, would add further incremental compliance costs. In addition, we took particular exception to the doubts CARB cast on the NREL study (page IX-73) by questioning its quality because of a small sample size. CARB knows well the number of major truck OEMs, and while the same could be said of our study, it covered every OEM of consequence. Moreover, the results of the ACTR study fell very close to the NREL study, both in stark constrast to the CARB SRIA.

To conclude, ACTR's analysis suggests that, in 2019 dollars, the new purchase price of an HHD vehicle will rise by $69,930 in MY2027 from the current baseline in a California-only scenario, which would fall to $25,825 on a nationwide basis. CARB's SRIA does not add up the estimated costs to present them on a per-unit basis in total, which seems very pertinent in our view. Nonetheless, adding up the costs in CARB's SRIA, we reach roughly $10,000 per unit for MY2027, though this is not clear given the lack of transparency on market sizing (note: we combined the MY2024 proposals into our MY2027, as the MY2024 timeframe was deemed infeasible from a planning and testing perspective). CARB's numbers do not account for the higher total-cost-of-ownership burden that will be borne by the trucking industry (including ACTR CA-only cost estimates of $8,392 from 12% FET, $5,070 from 7.25% state taxes, etc.) and ultimately by consumers, nor does it realistically reflect the likely Pre-buy/No-buy, R&D, UL and warranty cost impacts of the proposed Omnibus regulations. If we are even "ballpark" correct in our cost assessment, the cost increases at issue have the potential to meaningfully move the trucking industry away from vehicles that meet CARB's proposed mandates, thereby reducing the regulations' potential benefits for several years, especially if the regulations requiring significantly more expensive trucks align with the peak of an economic cycle. If that happens, we can expect an even larger pre-buy ahead of the mandate, and an extended post-mandate delay, which would invalidate much of CARB's cost analysis and delay the anticipated benefits.

EMFAC SCENARIOS
Revised with Latest CARB California-Original Vehicle Splits

John Grant, Yesica Alvarez

RAMBOLL

OCT, 25 2019

OUTLINE

- Purpose
- Scope and methods
  - Scenarios
  - NOx reduction methods
  - CA-Original vehicles
- Results
- Summary of findings

PURPOSE

- To provide an updated estimate of the effect on California state-wide and South Coast Air Basin NOx emissions of various heavy duty diesel truck emission standard policy scenarios in calendar years 2031, 2035 and 2045.
- Update is based on recent CARB-developed methodology to estimate CA-original purchased vehicles.
- Emission Standard Scenarios
  - Base Case (EMFAC2017 default; based on 0.2 g/bhp-hr standard for NOx)
  - Scenario 1a (CARB White Paper, least stringent, California Fleet Only)
  - Scenario 1b (CARB White Paper, most stringent, California Fleet Only)
  - Scenario 2 (EMA Voluntary Nationwide)

SCOPE & KEY ASSUMPTIONS

- **Pollutant**: NOx
- **Calendar** Years (CY): 2031, 2035, 2045
- **Geographies**:
  - California state-wide
  - South Coast Air Basin (SoCAB)
- **Applicable Vehicles**: Diesel MHD and HHD Trucks
- **Applicable Model Years (MYs)**: Engine model years 2024, 2025 and 2026 (corresponding to EMFAC vehicle model years 2025, 2026, and 2027)
- Base case emission rates and vehicle activity are from the CARB EMFAC2017 model
- Emissions Reductions Applied by Mode
  - Running exhaust:
    - Zero Mile Rate (ZMR)
    - Deterioration (no reductions estimated for this mode)
  - Extended Idling Exhaust
  - Start Exhaust

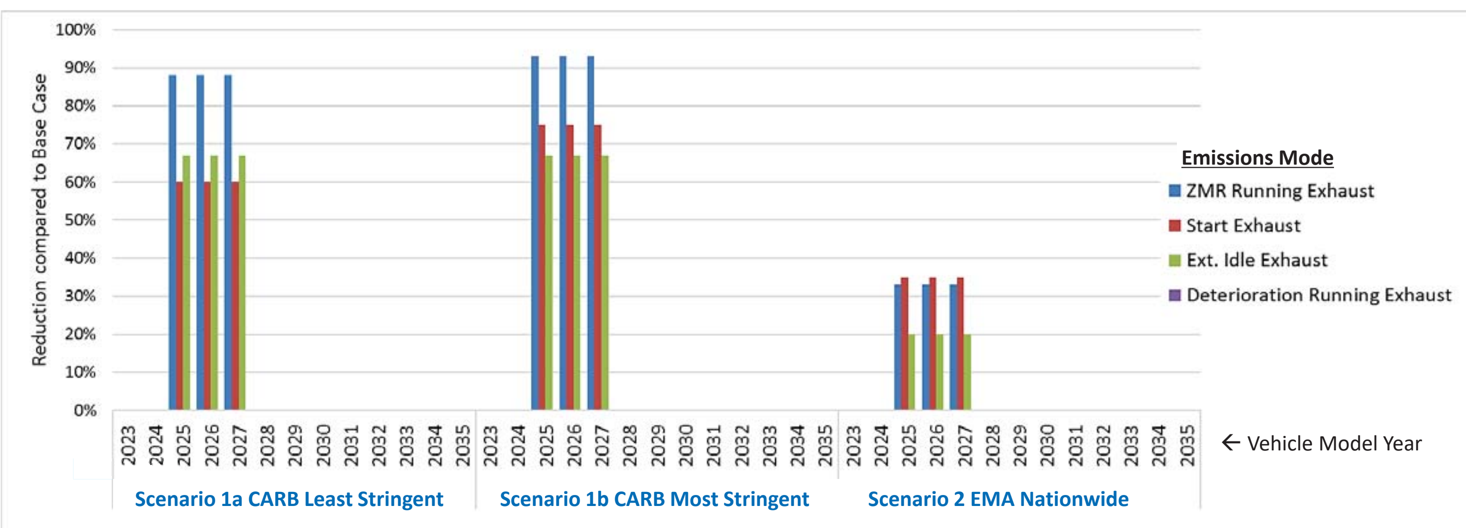
EMFAC SCENARIOS SUMMARY

NOx Emission Component	Scenario 1a CARB White Paper, Least Stringent California MHD & HHD Trucks Only[1,2]	Scenario 1b CARB White Paper, Most Stringent California MHD & HHD Trucks Only[1,2]	Scenario 2 EMA Voluntary Nationwide All MHD & HHD Trucks[2]
Running Exhaust: ZMR	88%	93%	33%
Running Exhaust: Deterioration	0%	0%	0%
Extended Idling Exhaust	67%	67%	20%
Start Exhaust	60%	75%	35%

[1] Scenario applies only to CA-original (purchased) vehicles

[2] Emission reductions applied only to engine model years 2024, 2025 and 2026 corresponding to EMFAC vehicle model years 2025, 2026, and 2027

REDUCTIONS TO NOX EMISSION STANDARD BY SCENARIO

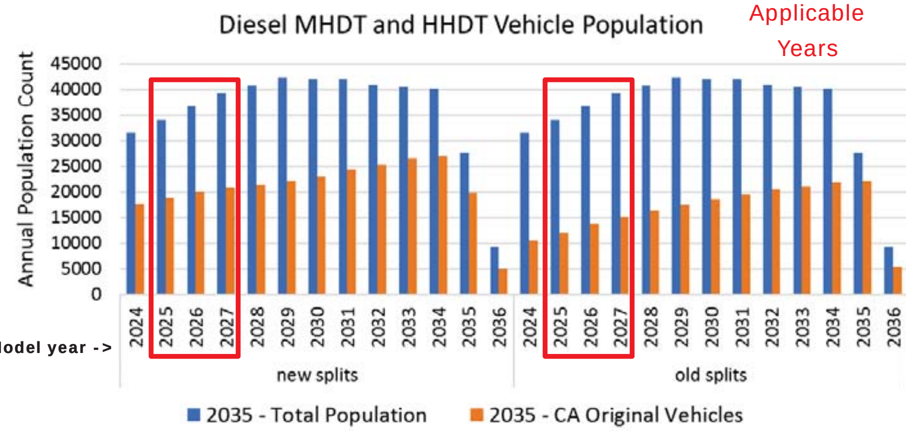


- Additional Scenario Specific Factors
  - Scenarios 1a and 1b include CA-only standards which apply to only vehicles originally purchased in CA. Vehicles originally purchased in CA would meet the CA standard; vehicles that migrate into the CA fleet from out-of-state would meet the current Federal standard
  - Reductions apply to vehicle model years (2025, 2026 and 2027). Emissions for other model years are equivalent to the EMFAC2017 Base Case

Exhibit E

CALIFORNIA-ORIGINAL VEHICLE POPULATION EXAMPLE: CALENDAR YEAR 2035

- New CARB methodology was used to estimate population splits (i.e., vehicles purchased in CA and vehicles purchased outside CA)
- CARB First Sold Fractions have been applied to population and VMT to estimate percentage of CA-original vehicle activity
- Based on new method, percentage of CA-original HD vehicles is larger than previous estimates



% CA-Original Population for Diesel MHDT and HHDT

Model Year	Old Method	New CARB Splits		
	2035	2031	2035	2045
2025	35%	53%	55%	62%
2026	37%	55%	54%	62%
2027	39%	58%	53%	61%

ESTIMATED NOX REDUCTIONS BY SCENARIO

Scenarios	Diesel HD Truck* NOx Emissions (tpd)			Total NOx Emissions Reduction from Base (tpd)		
	2031	2035	2045	2031	2035	2045
California State-wide						
Base Case	208	214	237			
Scenario 1a CARB White Paper Least Stringent, CA-only fleet	200	208	235	7.91	6.16	2.35
Scenario 1b CARB White Paper Most Stringent, CA-only fleet	200	208	235	8.50	6.63	2.56
Scenario 2 EMA Voluntary Nationwide	200	208	236	7.66	5.81	1.80
South Coast						
Base Case	62	63	70			
Scenario 1a CARB White Paper Least Stringent, CA-only fleet	60	61	70	2.42	2.01	0.85
Scenario 1b CARB White Paper Most Stringent, CA-only fleet	59	61	70	2.62	2.17	0.92
Scenario 2 EMA Voluntary Nationwide	60	62	70	2.08	1.72	0.62

* Diesel MHDT and HHDT trucks

ESTIMATED NOX REDUCTIONS BY SCENARIO

Scenarios	Diesel HD Truck* NOx Emissions (tpd)			% Reduction from Base		
	2031	2035	2045	2031	2035	2045
California State-wide						
Base Case	208	214	237			
Scenario 1a CARB White Paper Least Stringent, CA-only fleet	200	208	235	3.8%	2.9%	1.0%
Scenario 1b CARB White Paper Most Stringent, CA-only fleet	200	208	235	4.1%	3.1%	1.1%
Scenario 2 EMA Voluntary Nationwide	200	208	236	3.7%	2.7%	0.8%
South Coast						
Base Case	62	63	70			
Scenario 1a CARB White Paper Least Stringent, CA-only fleet	60	61	70	3.9%	3.2%	1.2%
Scenario 1b CARB White Paper Most Stringent, CA-only fleet	59	61	70	4.2%	3.4%	1.3%
Scenario 2 EMA Voluntary Nationwide	60	62	70	3.3%	2.7%	0.9%

* Diesel MHDT and HHDT trucks

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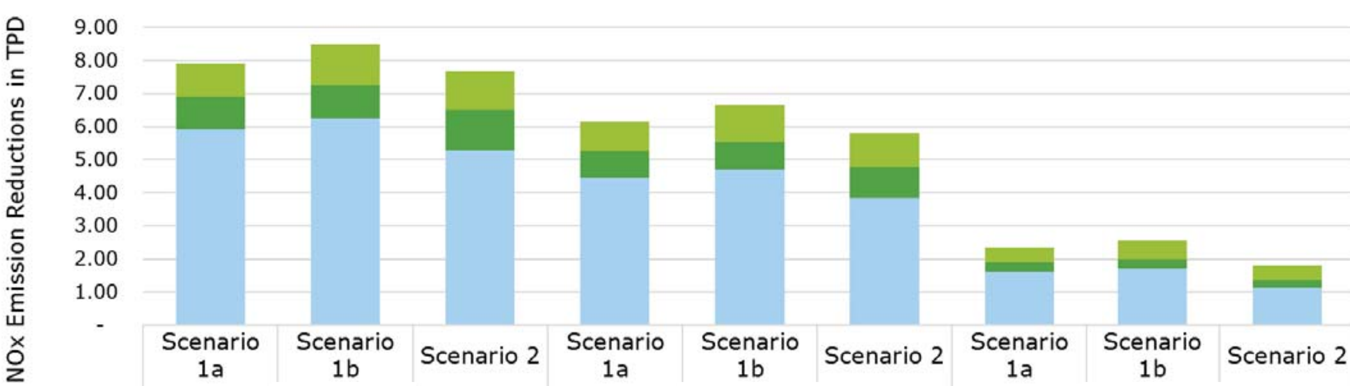
STATE-WIDE NOX REDUCTIONS (TPD) BY EMISSIONS MODE



	Scenario 1a (2031)	Scenario 1b (2031)	Scenario 2 (2031)	Scenario 1a (2035)	Scenario 1b (2035)	Scenario 2 (2035)	Scenario 1a (2045)	Scenario 1b (2045)	Scenario 2 (2045)
Start Exh.	1.00	1.25	1.15	0.89	1.12	1.03	0.44	0.56	0.44
Ext. Idle Exh.	1.00	1.00	1.22	0.82	0.82	0.94	0.28	0.28	0.25
Running Exh.	5.91	6.25	5.30	4.45	4.70	3.84	1.63	1.72	1.12
Total	**7.91**	**8.50**	**7.66**	**6.16**	**6.63**	**5.81**	**2.35**	**2.56**	**1.80**

Scenario 1a = CARB White Paper, Least Stringent, CA-only fleet
Scenario 1b = CARB White Paper, Most Stringent, CA-only fleet
Scenario 2 = EMA Voluntary Nationwide
* Running exhaust emission reductions result from ZMR reductions, deterioration rates were unchanged from the EMFAC2017 base case

SUMMARY OF FINDINGS (STATE-WIDE AND SOUTH COAST)

- Estimated NOx emission reductions from 1.8-8.5 TPD state-wide and 0.6-2.6 TPD in the South Coast across all three scenarios and three calendar years (2031, 2035, 2045)
  - Emission reductions from MYs2025-2027 cohorts decrease over time (2031 shows largest reductions while 2045 has lowest reductions) due to decreases in vehicle activity as vehicles get older
- New CARB CA-original vehicle splits generates larger reduction for CARB White Paper Scenario than previous CA-split method
  - California-registered heavy duty vehicles represent about 53-55% of state-wide heavy duty vehicles of MY2025-2027, therefore CA-specific standards affect a larger portion of the NOx inventory under the new methodology

		Comparative Ratio of Reductions (EMA Proposal / CARB Proposal)			
Region	Scenario	2031	2035	2045	2031 (CARB Estimate)
State-wide	Scenario 1a CARB Least Stringent	0.97	0.94	0.77	n/a
	Scenario 1b CARB Most Stringent	0.90	0.88	0.71	0.8
South Coast	Scenario 1a CARB Least Stringent	0.86	0.86	0.73	n/a
	Scenario 1b CARB Most Stringent	0.79	0.79	0.67	n/a

END

CY2031 & 2045 REDUCTIONS ON AFFECTED FLEET: (SCENARIOS 1A AND 1B CARB WHITE PAPER)

State-wide MHD and HHD NOx emissions reductions for affected vehicle model years for Scenarios 1a and 1b for the **CA-original Diesel MHDT and HHDT fleet**:

NOx Emission Reduction from Base Case (%) [a]						
Scenario		Vehicle Model Year	Running exhaust[b]	Idling exhaust	Start exhaust	Total
Scenario 1a CARB Least Stringent **CA-Only Fleet**	**2031**	2025	47%	67%	60%	50%
		2026	49%	67%	60%	52%
		2027	53%	67%	60%	55%
	2045	2025	38%	67%	60%	43%
		2026	38%	67%	60%	44%
		2027	39%	67%	60%	44%
Scenario 1b CARB Most Stringent **CA-Only Fleet**	**2031**	2025	49%	67%	75%	54%
		2026	52%	67%	75%	56%
		2027	56%	67%	75%	59%
	2045	2025	40%	67%	75%	47%
		2026	41%	67%	75%	47%
		2027	41%	67%	75%	48%

[a] *Emissions reductions for CA-only MHD and HHD trucks*
[b] *Emission reductions for running exhaust are based on zero-mile emission rate reductions with no change to deterioration*

Emissions from non-CA vehicles in Scenarios 1a and 1b are unchanged from the Base Case and are not included in the table above

CY2031 & 2045 REDUCTIONS ON AFFECTED FLEET: (SCENARIO 2 EMA NATIONWIDE)

State-wide NOx emission reductions are shown below for applicable vehicle model years over the entire fleet (**CA-original + out-of-state**):

NOx Emission Reduction from Base Case (%)[a]						
Scenario		Vehicle Model Year	Running exhaust[b]	Idling exhaust	Start exhaust	Total
Scenario 2 EMA Nationwide	2031	2025	17%	20%	35%	19%
		2026	18%	20%	35%	20%
		2027	19%	20%	35%	21%
Scenario 2 EMA Nationwide	2045	2025	14%	20%	35%	18%
		2026	14%	20%	35%	18%
		2027	15%	20%	35%	18%

[a] *Emissions reductions over Base Case (Fleet of Diesel MHD and HHD trucks; CA-original and OOS)*

[b] *Emission reductions for running exhaust are based on zero-mile emission rate reductions with no change to deterioration*

CALIFORNIA FLEET ACTIVITY TREND

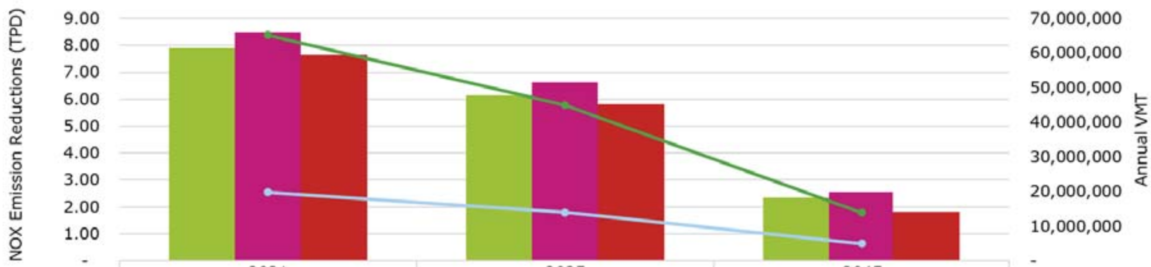
The decrease in EMA/CARB emission reductions over time is consistent with changes to vehicle activity affected by each program

		Ratio of Reductions (EMA Proposal / CARB Proposal)		
Region	Scenario	2031	2035	2045
State-wide	Scenario 1a CARB Least Stringent	0.97	0.94	0.77
	Scenario 1b CARB Most Stringent	0.90	0.88	0.71

Vehicle Miles Traveled for Diesel MHDT and HHDT

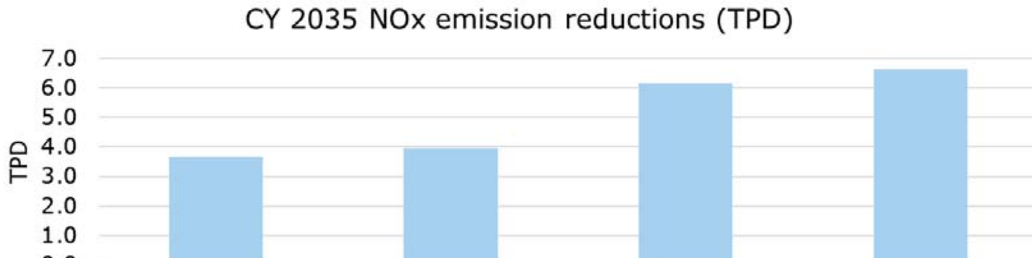
	VMT TOTAL / [VMT CA-Only]		
Model Year	CY2031	CY2035	CY2045
2025	2.42	2.15	1.77
2026	2.33	2.22	1.79
2027	2.17	2.30	1.81
All 3 MYs	2.30	2.23	1.79

Decrease over time



	2031	2035	2045
Scenario 1a CARB (least)	7.91	6.16	2.35
Scenario 1b CARB (most)	8.50	6.63	2.56
Scenario 2 EMA	7.66	5.81	1.80
Total VMT	45,417,781	31,083,818	8,877,656
CA-original VMT	19,765,723	13,946,773	4,956,678

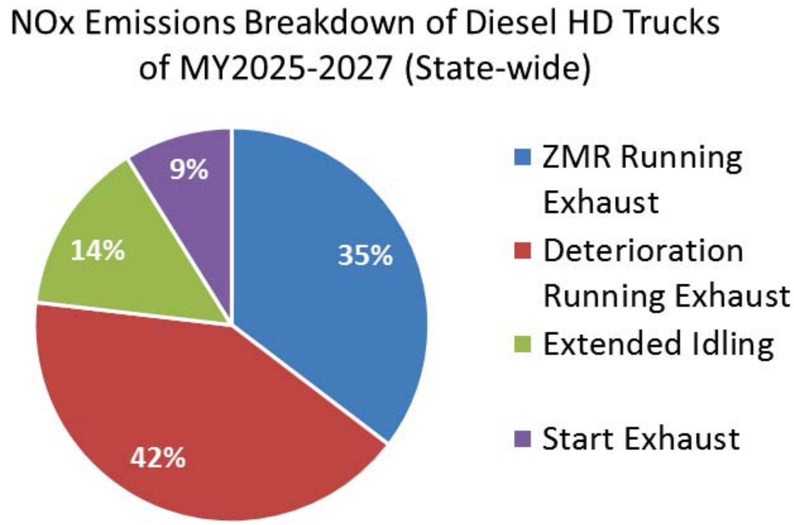
COMPARISON OF STATE-WIDE 2035 REDUCTIONS: CA-SPLITS OLD VS. NEW



	CARB old method		CARB new splits	
	Scenario 1a CARB White Paper (Least Stringent)	Scenario 1b CARB White Paper (Most Stringent)	Scenario 1a CARB White Paper (Least Stringent)	Scenario 1b CARB White Paper (Most Stringent)
2035	3.7	4.0	6.2	6.6
% NOx Reduction from Base	1.7%	1.9%	2.9%	3.1%
% CA-original CY2035 Population*	37%		54%	

* Diesel MHDT and HHDT trucks

CALIFORNIA STATE-WIDE CALENDAR YEAR 2035 NOX EMISSION CONTRIBUTIONS FOR MHD AND HHD TRUCKS BY MODE FOR VEHICLE MODEL YEARS 2025-2027



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ENVIRONMENT
& HEALTH

RECENT EVIDENCE OF CONTINUED NOX DISBENEFITS IN LOS ANGELES

Ozone is formed in the troposphere (e.g., ground-level) through a complex set of chemical reactions involving Volatile Organic Compounds (VOC) and oxides of nitrogen (NO_X) in the presence of sunlight. It is well known that in areas of concentrated NO_X emissions, such as within point source plumes and in some urban areas, reducing NO_X emissions can increase ozone concentrations. This “NO_X disbenefit” ozone effect has been known for many years. In fact, 50 years ago the 1970 Clean Air Act (CAA), Section 182(f)[1] had provisions to exempt NO_X RAQC control requirements in an area if they are determined to not produce a net ozone air quality benefit because of the concerns that implementing NO_X controls in urban areas could exacerbate the ozone problem.

Past Observed Ozone NO_X Disbenefits in the SoCAB

The NO_X disbenefit in urban areas, and in the South Coast Air Basin (SoCAB) in particular, was confirmed using ozone observations based on the “weekend effect.” Early in this century, numerous peer-reviewed studies found that high ozone concentrations in the SoCAB (and some other urban areas) were more likely to occur on weekend days than weekdays (Marr and Harley, 2002a,b; Heuss et al., 2003; Lawson, 2003; Fujita et al., 2003; Blanchard and Tanenbaum, 2003; Chinkin et al., 2003; Yarwood et al., 2003; Pun and Seigneur, 2003; Steadman, 2004; Murphy et. al., 2006; Tonse et al., 2008) The lower NO_X emissions on weekend days resulted in a higher frequency of occurrence of elevated ozone concentrations than on weekdays, when more mobile source NO_X emission due to the commute and other activities produced higher NO_X emissions. The “weekend effect” was especially apparent in the SoCAB and firmly established the fact that NO_X emissions reductions would increase ozone concentrations that would have to be counteracted with more VOC emission reductions.

Kim et al., (2016) modeled the SoCAB during the 2010 CalNex field study and found higher observed and modeled ozone concentrations were still more likely on weekend days than weekdays in 2010.

Current Observed Ozone NO_X Disbenefits in the SoCAB

Over the last several decades there have been large amounts of reductions in VOC and NO_X emissions in the SoCAB resulting in reduced ozone concentrations. Morris and co-workers (1998; 1999) examined the trends in ozone and ozone precursors in the SoCAB from 1980 onward and found large reductions in both (Figure 1). However, trends have shown increases in ozone design values in more recent years (i.e., 2016, 2017 and 2018).

[1] https://www.govinfo.gov/content/pkg/USCODE-2013-title42/html/USCODE-2013-title42-chap85-subchapI-partD-subpart2-sec7511a.htm

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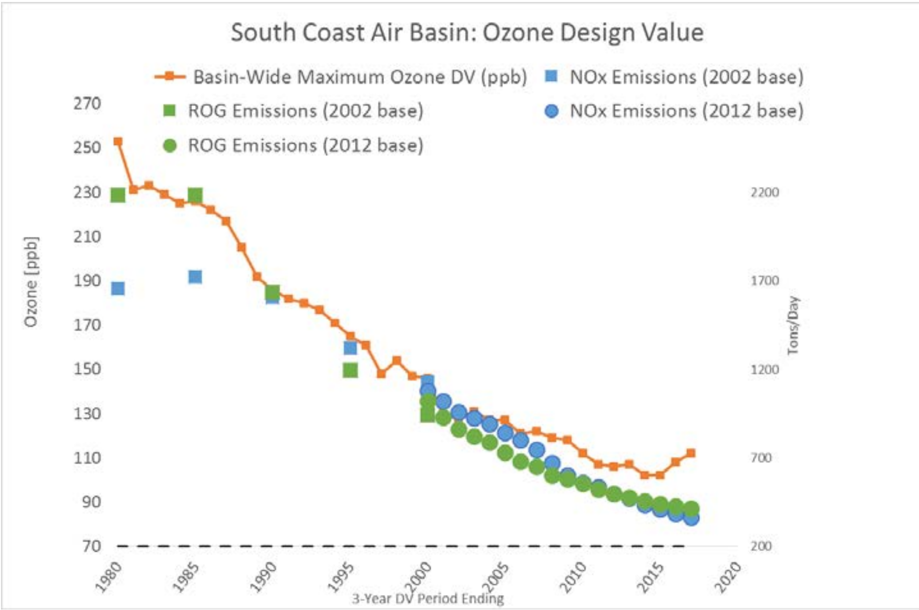

Figure 1. Trends in SoCAB maximum ozone design values (orange) and basin-wide VOC/ROG (green) and NO_X (blue) emissions from 1980 to 2018 (Source: Morris et al., 2019).

The chemistry in the SoCAB is changing and there is a lessening of the ozone NO_X disbenefit effect as evidenced by a weakening of the weekend ozone effect (Baidar et al., 2015). Baider and co-workers examined the ozone weekend effect in the SoCAB for 1996 to 2014 and found most of the ozone reductions during that period were due to reductions in ozone concentrations on weekends in the eastern part of the SoCAB where there was a lessening of the weekend effect, but as one gets closer to central Los Angeles (CELA; e.g., North Main Street site), the ozone reductions and lessening of the weekend effect was reduced with the weekend effect still very evident in CELA. Figure 2 shows the trends in weekday and weekend ozone concentrations (April through October average MDA8) averaged over the three design value sites in the SoCAB (i.e., inland) from 2000 to 2018, which indicate that the downward ozone trends are due to ozone reductions on weekends. Figure 3 shows weekday/weekend ozone averaged over April – October for the most recent 5 years at the Los Angeles North Main location where the weekend effect persists, and ozone is 4.8 ppb higher on weekend days than weekdays due to the lower NO_X emissions on weekend days.

Thus, in the eastern part of SoCAB the weekend effect is lessening and is the primary driver for decreasing design values in the SoCAB over the last 2 decades. However, it is also the case that the weekend effect still persists near and at the CELA monitoring site where over the last 5 years ozone levels on weekends continue to be substantially higher than on weekdays.

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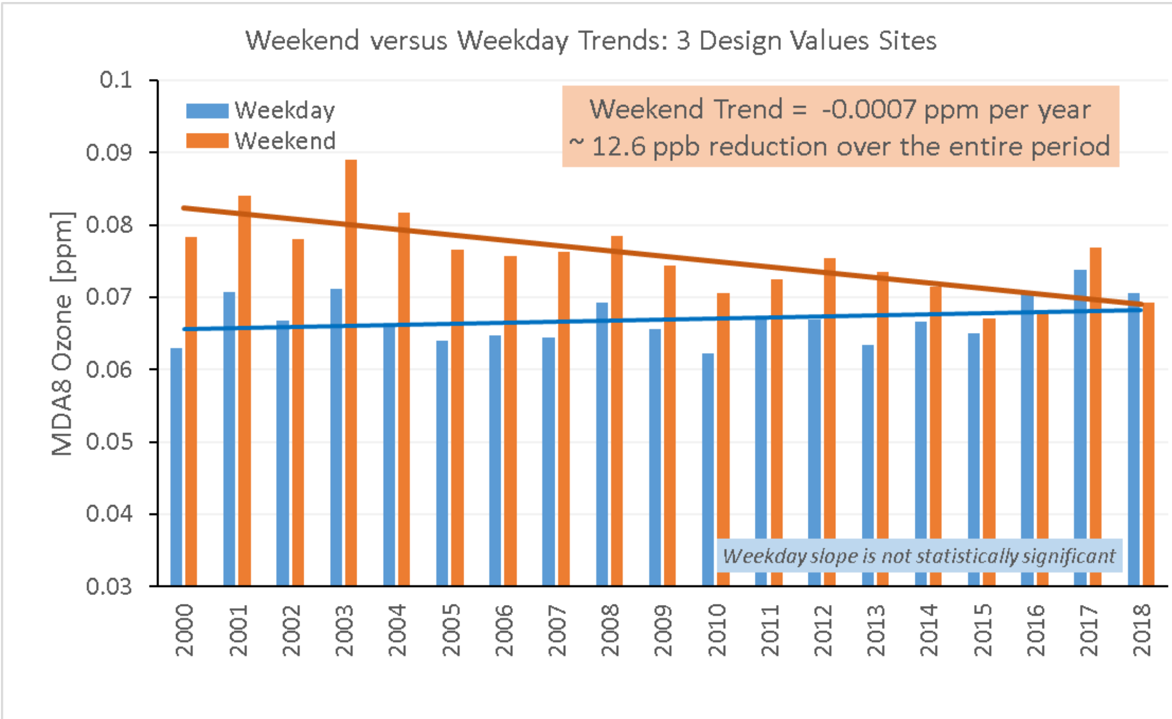

Figure 2. 2000 to 2018 trends in highest ozone at ozone design value sites (i.e., inland) in the SoCAB.

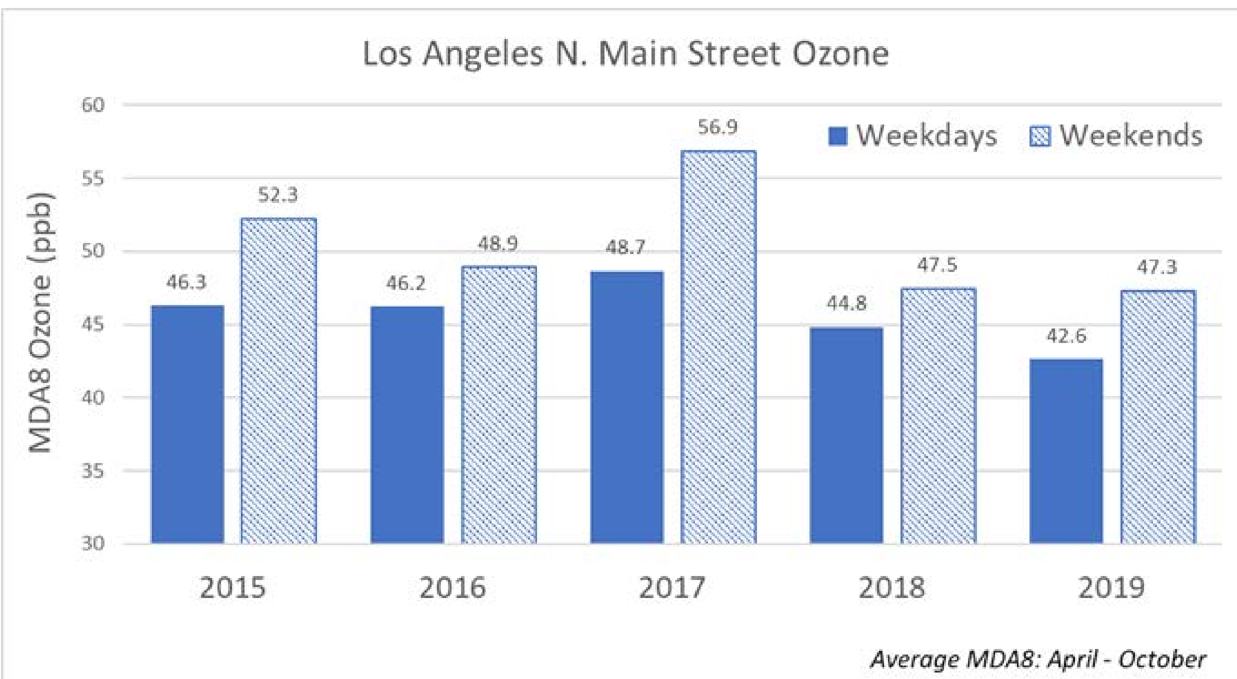

Figure 3. 2015 – 2019 MDA8 ozone at CELA by weekday and weekend.

Future Year NO_X Disbenefits in the SoCAB

Three recent studies assessed recent and/or future year ozone chemistry in the SoCAB and all indicate the persistence of the NO_X disbenefit in the central part of the SoCAB including CELA: Fujita et. al. (2016), EPA (Simon et. al., 2020) and Collet et. al. (2018). These three studies are summarized below.

Fujita et. al. (2016) used the CMAQ model to examine the effects of varying NO_X and VOC emissions on the magnitudes and peak ozone levels within the SoCAB. They predict that a −61% reduction (from 2008 emissions) of NOx (from 723 to 284 tons per day, tpd) and −32% reduction of ROG (from 639 to 437 tpd) would lead to average daily maximum 8-hr ozone concentration increases of 18, 14, 11 and 10 percent at Los Angeles, Azusa, Pomona, and Upland, respectively, and decreases of 2 and 4 percent at Rubidoux and Crestline, respectively. In addition, they predict that design values would be exceeded at all six SoCAB sites they considered, including the CELA site. However, they also address uncertainty in VOC emissions and note that if SoCAB VOC emissions are currently underestimated (as they believe they are) and VOC reductions are made, then the predicted NO_X disbenefits may not occur. In terms of the location of peak ozone, Fujita et. al. predict that: “The Basin maximum ozone site will shift westward to more populated areas of the Basin and will result potentially in greater population-weighted exposure to ozone with even a relatively small shortfall in the required NOx reductions unless accompanied by additional VOC reductions beyond 2030 baseline levels.”

EPA personnel gave a recent presentation at the June 2020 Air and Waste Management Association annual conference entitled “Changes in Ozone Photochemistry across the U.S. between 2007 and 2016: An Integrated Modeling Assessment” (Simon et.at, 2020). They showed that although the spatial extent of VOC-limited areas where NO_X disbenefits occur has shrunk between 2007 and 2016, in the SoCAB there is a large area from the coast to San Bernardino County that is still VOC-limited in 2016. Figure 4 shows the area of VOC-limited ozone formation in the SoCAB as estimated by Simon et al., (2020) that corresponds to a high population density area stretching from LAX/Long Beach through Los Angeles, Pasadena, Azusa and Upland into Upland and Fontana in San Bernardino County, where NO_X emission reductions will cause increases in ozone concentrations.

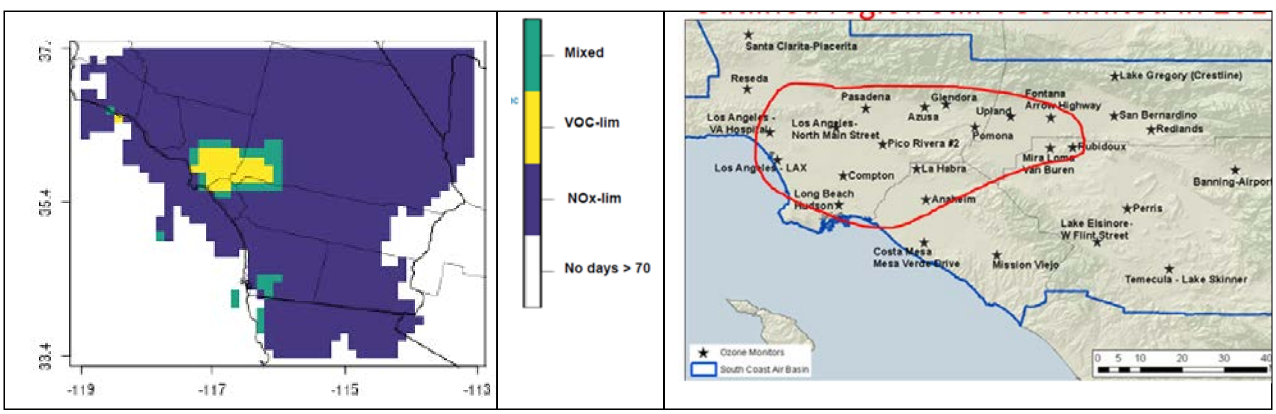

Figure 4. Area of VOC-limited ozone formation (yellow) in the SoCAB where NO_X disbenefits can occur (Source: Simon et al., 2020).

The objective of the study by Collet and co-workers (2018) was to use the CAMx photochemical grid model to develop future-year (2030) VOC-NO_X ozone isopleth diagrams of the 4th highest daily maximum 8-hour ozone concentrations at monitors of interest in the SoCAB and San Joaquin Valley (SJV) in California. A 2030 emissions scenario was developed based on EPA's 2025 emissions with on-road mobile sector emissions developed for 2030 using the latest on-road emissions models. To generate the VOC-NO_X ozone isopleths, a set of reduction factors (20%, 40%, 60% and 80%) was applied to NO_X, anthropogenic VOCs, and CO emissions, and 25 future-year simulations were performed with CAMx for the California region. The results of the simulations were smoothed using interpolation to create the isopleth diagrams and the results for six SoCAB sites are shown in Figure 5. Ozone isopleths for the most inland sites (Banning and Crestline; Figure 5, bottom) indicate that 2030 ozone chemistry is generally NO_X-limited. Whereas, the mid-basin sites (Glendora, Pasadena and Anaheim) are indicated to be in a transitional ozone chemistry regime in 2030 where NO_X control alone will likely increase ozone. The Los Angeles - North Main site indicates that at 2030 emissions conditions (i.e. upper right corner of isopleth) the site is VOC-limited and therefore a decrease in NO_X without a corresponding decrease in VOCs will lead to an increases in ozone (i.e. the upper right corner is below the ozone ridgeline in the isopleth diagram). This work illustrates that even though ozone is relatively low at the Los Angeles – North Main site, increases could potentially occur in this highly populated region if emission reduction strategies focus solely on NO_X-controls.

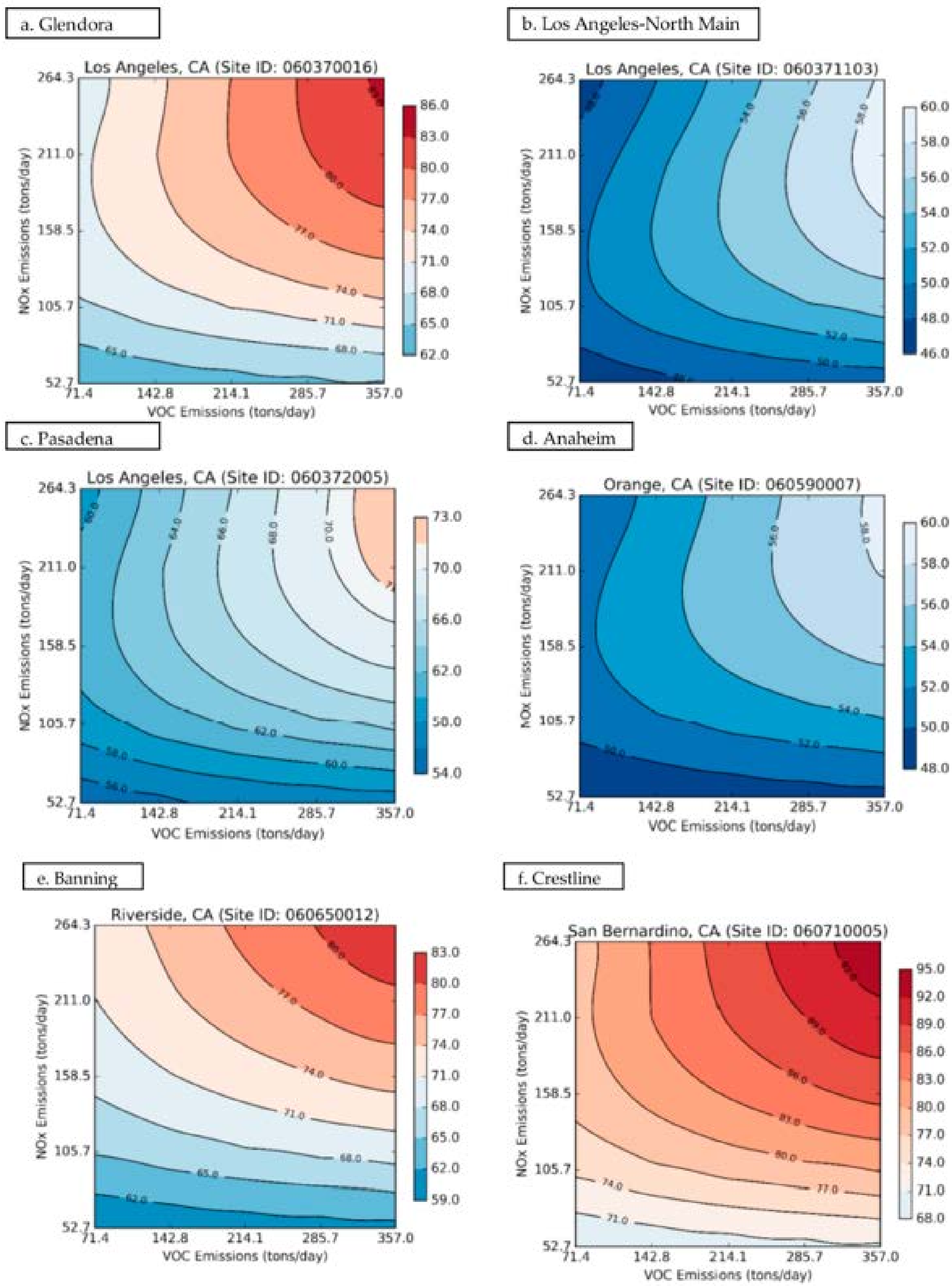

Figure 5. Future-year (2030) VOC- NO_X isopleth diagrams of the 4th highest maximum MDA8 ozone at monitors of interest in the SoCAB. (Source: Collet et al., 2018).

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Conclusions on NO_X Disbenefits in the SoCAB

The occurrence of NO_X disbenefits in the SoCAB is well established. Over the years there has been a lessening of the weekend effect and resulting NO_X disbenefits in the eastern part of the SoCAB. But current year ozone observations and future year modeling out to 2030 indicate the NO_X disbenefit ozone effect is still occurring in the central part of the SoCAB include Los Angeles.

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Exhibit G

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Report

CARB 3B-MAW Analysis and Results

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August 11th, 2020

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1 INTRODUCTION

This document presents a detailed analysis of the three-bin moving-average windowing (3B-MAW) method that the California Air Resources Board (CARB) has proposed for analyzing real-world data collected from heavy-duty, on-highway (HDOH) vehicles operated over random, but vehicle-category-specific activity patterns during their regular daily usage by a fleet. Data from a total of 110 vehicles, distributed across different vehicle categories as shown in Table 1, was collected over a period of approximately 30 days per vehicle using telemetry systems and independent NO_x and NH_3 sensors. The distribution by vehicle class, weight category, and model year of the recruited vehicles was modeled based on EMFAC2014 model outputs for the vehicle fleet operating within the South Coast Air Basin (SoCAB)[1] as well as the market share of individual engine original equipment manufacturers (OEM) within the California HDOH market.

The independent NO_x sensors (i.e., Continental Gen 2 NO_x) were employed in order to avoid the shortcomings of vehicle on-board NO_x sensors that are only operational above a certain threshold temperature in order to preserve them from mechanical damage at lower exhaust gas temperatures. The independently installed sensors were turned on as soon as a vehicle key-on event was detected and were subjected to a ~30sec to 1min maximum heat-up time before being ready to measure NO_x concentrations in the exhaust stream. This allowed for the capture of cold-start engine operation as well as exhaust concentrations during extended idle and low-load operation, which would be missed with a traditional vehicle on-board NO_x sensor when it turns off at exhaust temperatures below ~180℃.

Table 1: Summary of vehicle categories, specific vocations/applications, EMFAC class definition, and number (i.e., count) of test vehicles for given categories.

Category	Vocation	EMFAC Class	Vehicle Count
1a	Long haul	T7 NNOOS, NOOS, CAIRP	26
1b	Short haul	T7 tractor	23
2a	Port Drayage	T7 POLA	17
3a	Tractor construction heavy	T7 single construction	5
3b	Cement mixer	T7 single construction	6
4	Tractor construction	T7 tractor construction	8
6a	Food/Beverage Distribution	T6 instate small	8
6b	Moving / Towing	T6 instate heavy	15
7a	Goods distribution	T7 Single	1
7b	Moving / Towing	T7 Single	1

1.1 Dataset characteristics and analysis methods

Vehicles selected for this study, and instrumented with the telemetry systems, spanned six heavy-duty engine manufacturers, and ranged from model year (MY) 2015 to 2019, and were equipped with 6.7 to 15 Liter (i.e., total engine displacement) engines. Also, all vehicles were within '*useful life*' and maintenance records were consulted before data-logging commenced to

[1] *Ramboll Environ, "EMFAC2014 Heavy Duty Diesel Fleet Characterization for the South Coast Air Basin (SCCAB)," Draft Memorandum, July 19th, (2017).*

assure that vehicle maintenance standards were followed and that there had been no tampering with any vehicle/engine components.

Vehicles in categories 1a and 1b are characteristic of long- and short-haul goods movement operations, respectively. There is no technological difference between EMFAC class T7 NNOOS, T7 NOOS, T7 CAIRP and T7 tractor heavy-heavy-duty trucks (HHDT), regardless of which state they are registered in and operating. However, based on the daily VMT sub-categories established from EMFAC2014 for vehicle classes operating within the SoCAB, T7 tractors are predominantly used for short hauling activity, whereas vehicles from the other three HHDT classes are utilized for long-haul operations. Thus, for this program it was assumed that category 1b vehicle activity is characterized by goods movement operations with a radius of ~200miles of their origin, with trucks returning to their fleet yard at the end of the shift-day (e.g., delivery of goods from central warehouses to large grocery stores across Southern California, transfer of domestic garbage from collection facilities to distant landfills, etc.). Category 1a vehicles, on the other hand, were assumed to be trucks used for goods movement over longer distances, possibly into neighboring states, but still having their origin in Southern California to where they would return after delivering their loads.

Category 2a includes vehicles used in port drayage activities with frequent access to the Ports of Los Angeles and Long Beach, and characterized by extended idle and low-speed creep activity while operating withing the port to receive their loads, and to move over congested port access routes and freeway segments between the ports and inland warehouses and railhead facilities.

Categories 3a and 3b are both single-unit trucks used in construction applications, with category 3a comprising dump-truck-type vehicles that could haul an additional bucket-trailer. Category 3b vehicles are cement trucks and were grouped in a separate category due to their very different duty-cycles, including significant power takeoff (PTO) operation to continuously turn their drum, while carrying fresh cement from the production facilities to construction sites. Category 4 vehicles are tractors related to construction operations that are used to haul different processed raw-materials (e.g., powder-trucks transporting ground stone powder between quarries and production facilities) or to transport construction equipment between construction sites (e.g. pulling flatbed trailer with excavator or similar machineries). ‘Powder-trucks’ also typically experience PTO operation during the unloading process, while using low-pressure air flow from a PTO-driven pump to empty their stone-powder vessels.

Categories 6a and 6b are medium-heavy-duty trucks that are utilized for food/beverage distribution, moving, and towing applications, and are split between vehicles with a GVWR > 26,000lbs (EMFAC T6 interstate heavy) and a GVWR ≤ 26,000lbs (EMFAC T6 interstate small), respectively. Those two vehicle categories are predominantly characterized by the highly transient urban/city driving environment with increased traffic density and frequent stop/go operation.

Categories 7a and 7b comprise single-unit heavy-heavy-duty trucks used for goods distribution (i.e. 7a) and moving/towing operations (i.e. 7b), respectively.

1.1.1 CARB 3B-MAW analysis method

Data analyzed is this report utilized the 3B-MAW analysis method released by CARB in March[2] and June[3] 2020. Since the data originated from telemetry-based ECU data-streams, CO_2 mass rates required for the 3B-MAW calculations had to be derived from engine fueling rate parameters and assumed fuel properties for typical Diesel fuel sold in Southern California. A detailed outline of the CO_2 mass rate calculation is provided in Appendix I, Section 3.2 of this report.

Data exclusions were applied as described in Appendix B-1, §6.2 of CARB's proposed 3B-MAW method and are listed in Table 2 below (i.e., §6.2.1 through §6.2.2). In addition, any vehicle activity with an active malfunction indicator light (MIL) was flagged and excluded from the analysis presented in this document. As will be discussed in detail in the results and analysis section (Section 2), *engine-off* events during a shift-day due to loading/unloading of the vehicle, driver lunch break, etc., also were considered as additional exclusions although that is not currently the case under CARB's 3B-MAW method.

In addition, since NO_x emissions were quantified using an independent NO_x sensor for the telemetry-based datasets discussed in this report, the *PEMS-zeroing* exclusion (§6.2.1) was modified to account for the NO_x sensor stability criteria. The Continental NO_x sensors used in our analysis provided a qualitative parameter to assess sensor measurement validity and stability during their operation. Based on that parameter a '*Boolean*' state-variable was defined, analogous to the *PEMS-zeroing* flag, indicating the periods of valid NO_x concentration measurements. As mentioned earlier, the independent NO_x sensor was turned on at any vehicle key-on event and experienced a warm-up duration on the order of ~30seconds to 1minute, depending on its previous temperature state. During that period for example, the NO_x sensor data valid '*Boolean*' parameter would be set to zero, which would trigger the *PEMS-zeroing* exclusion (§6.2.1).

Table 2: Permitted 3B-MAW exclusions (Appendix B-1, §6.2).

Exclusion	Definition	Paragraph
PEMS-Zeroing (*NO_x-Stable*)	Data collected during PEMS-zeroing events (*Data collected during NO_x sensor not stable events*)	6.2.1
P_{amb}	Atmospheric pressure < 82.5kPa	6.2.2
T_{amb}	Ambient temperature < 19°F (-7°C)	6.2.3
Altitude	Altitudes > 5,500ft (*above sea level*)	6.2.4
T_{amb_upper}	Ambient temperature upper limit, $T = f(altitude)$	6.2.5
ECT	ECT < 70°C or ECT stable ±2°C over 5min at shift-day start	6.2.6
Engine-off	Engine-off events during loading/unloading, etc.,	Not listed
MIL active	Active malfunction indicator light	Not listed

[2] *State of California Air Resources Board, Draft Omnibus Regulation, Proposed Amendments to the Test Procedures, Appendix, Paragraph 6, March, (2020).*

[3] *State of California Air Resources Board, Proposed Heavy-Duty Engine and Vehicle Omnibus Regulation, Proposed Amendments to the Exhaust Emissions Standards and Test Procedures for 2024 and Subsequent Model Year Heavy-Duty Engines and Vehicles, Appendix B-1, Proposed Amendments to the Diesel Test Procedures, Paragraph 6, June 23rd, (2020).*

Since data was collected for approximately 30 shift-days for each test vehicle, continuously collected data was segregated into individual daily datasets for the 3B-MAW analysis based on the timestamps available from the raw data. This allowed us to assess the 3B-MAW method with typical single shift-day datasets as described in the proposed approach for PEMS datasets. Ultimately, this resulted in about 30 individual datasets for each of the 110 vehicles included in this study.

Each daily dataset was evaluated using the proposed 3B-MAW method and the validity of each shift-day was determined based on the three test validity criteria proposed by the CARB 3B-MAW method in Appendix B-1, §6.3, and shown in Table 3 below. A dataset must meet all three test validity criteria in order to be considered a *valid* test. If a daily dataset did not comply with the test validity criteria, it was flagged and not included in the overall data summary. Since the telemetry-based data used in this work originated from actual real-world vehicle operation by a fleet, not every shift-day would comply with the minimum of 3 hours valid non-idle operation ($t_{Non\text{-}Idle}$) due to shorter routes or limited operation, etc., on a given day of the week. Thus, those shift-days would be invalidated based on the test validity requirements outlined in the 3B-MAW method.

Table 3: Required 3B-MAW test validity criteria (Appendix B-1, §6.3).

Criteria	Definition	Paragraph
ECT	ECT at start of test ≤ 30°C	6.3.1
$t_{Non\text{-}Idle}$	Min. valid window requirement of 3 hours non-idle operation	6.3.2
P_{min}	Min. daily avg. power of valid operation ≥ 10% of $P_{peak,\ engine}$	6.3.3

2 CARB 3B-MAW ANALYSIS AND RESULTS

This section will discuss the results of our data analysis using the 3B-MAW method and will primarily focus on concerns and possible shortcomings, as well as the limitations of the proposed 3B-MAW procedure for use in CARB's heavy-duty in-use testing program. The analysis is split into specific areas of concern that will be discussed in detail in individual sub-sections.

2.1 Multiple counting of datapoints and unequal data weighting within bin-transitions

Figure 1 indicates how individual single datapoints are concurrently distributed among multiple bins using the proposed 3B-MAW approach due to the 300sec averaging window method. Data is presented for 9 different HDOH vocations and vehicle categories. On average across all vehicle categories considered in this study, the percentage count of single data points attributed to a single bin only is approximately 55.6%, whereas on average ~26.6% of individual data points are added to two bins at the same time and, about 1.4% of datapoints are concurrently present in all three bins. In addition, approximately 16.4% of all datapoints are excluded from the 3B-MAW analysis due to specified data-exclusions or short-duration vehicle operation that results in consecutive data of less than 5min (i.e., the required window size is 300sec). That is especially pronounced for the delivery truck category 6b, which includes vehicles with frequent engine stop/start events for goods deliveries in urban areas where fleets require their drivers to turn off the engine as soon as they stop to unload goods. Table 4 summaries the averages of multiply-binned datapoints for each individual vehicle category along with the global averages across all categories. The percentage count distributions for each individual vehicle category are given in Appendix I, Section 3.1 in Figure 37 through Figure 41 along with tabulated percentiles and averages in Table 11 through Table 15.

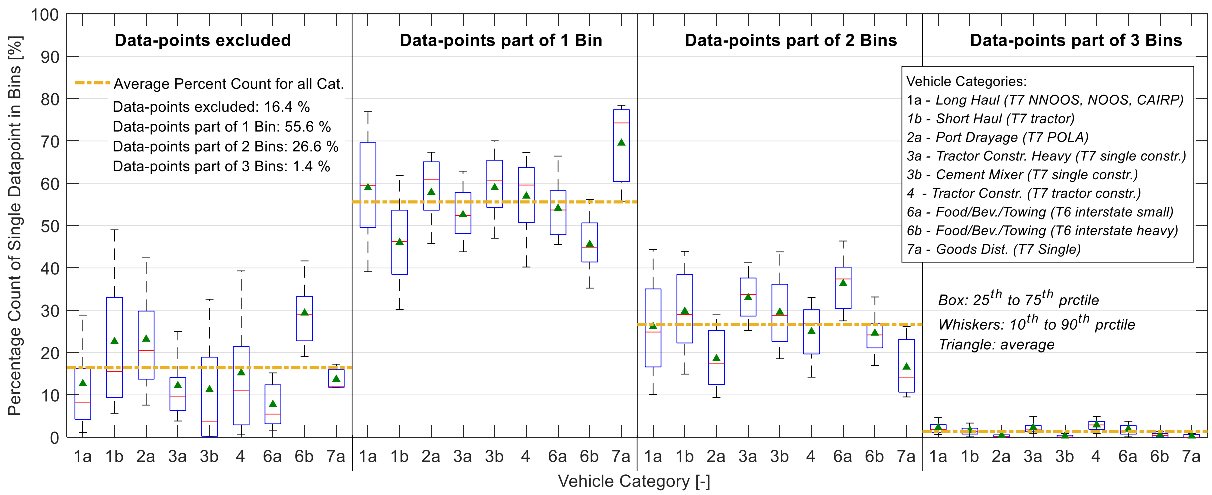


Figure 1: Percentage count of single data points appearing in either none (i.e., excluded), 1, 2 or 3 bins at the same time for all vehicle categories evaluated.

Table 4: Average percentage count per vehicle category of single data points appearing in either none (i.e., excluded), 1, 2 or 3 bins at the same time; global mean represents average across all vehicle categories.

Distribution of Data-points	Mean of Distributions [%]									Global Mean
[-]	1a	1b	2a	3a	3b	4	6a	6b	7a	[%]
Excluded Data (Part of 0 Bins)	12.66	22.61	23.15	12.19	11.23	15.20	7.73	29.33	13.68	16.42
Data-points Part of 1 Bin	58.94	46.03	57.85	52.57	58.95	56.96	54.07	45.55	69.49	55.60
Data-points Part of 2 Bins	26.15	29.75	18.56	32.97	29.54	24.94	36.31	24.58	16.56	26.60
Data-points Part of 3 Bins	2.25	1.62	0.44	2.27	0.28	2.90	1.89	0.54	0.28	1.38

Figure 2 provides an example of the data overlap between the three bins and the resulting unequal weighting of individual datapoints inside a given bin due to the transitioning of the 300sec windows between different bins. The data shown in Figure 2 was collected from a category 1a long-haul truck, and provides a 1 hour and 40 minute snapshot of a single operating day, representative of typical activity of a vehicle in this category operating in California.

The upper graph in Figure 2 shows the number of times a single datapoint is counted in a respective bin (i.e., Bins 1 through 3). It becomes obvious from Figure 2 that transitioning between different bins results in unequal weighting of an individual datapoint in a given bin as it becomes a reducing member of the previous bin and, at the same time, an increasing member of a subsequent bin. That systemic drawback of unequal weighting of individual datapoints becomes less and less pronounced as the averaging window size is reduced.

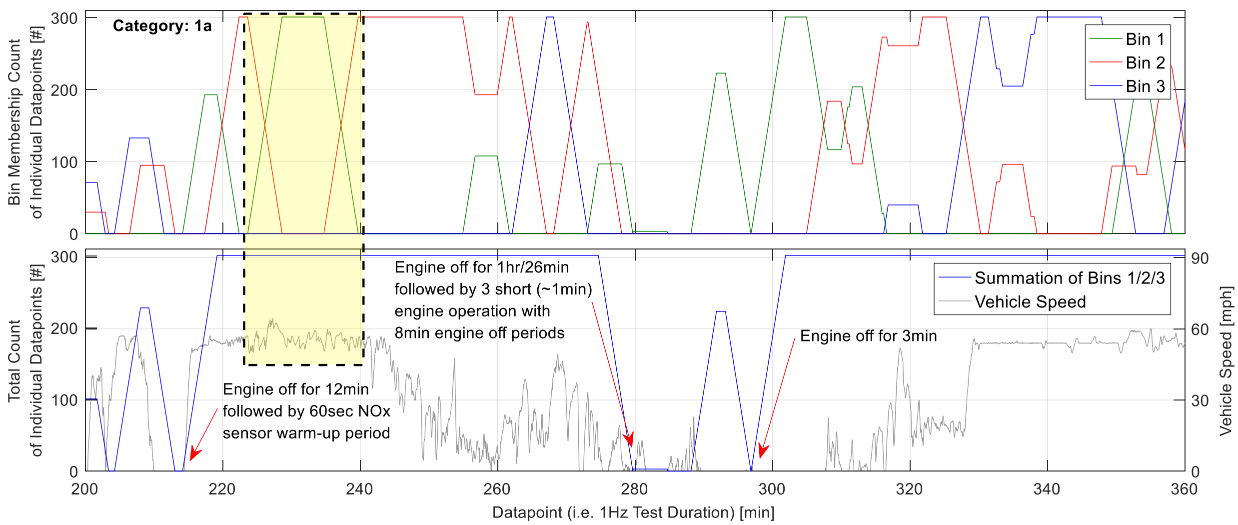


Figure 2: Bin membership count of individual datapoints to either of the three bins for a category 1a vehicle (i.e., long haul); data represents excerpt of single day operation.

Table 5: Percentage count single data points appearing in either none (i.e., excluded), 1, 2 or 3 bins at the same time for a single category 1a vehicle; data represents a single day of operation.

Data-points excluded [%]	Data-points part of 1 Bin [%]	Data-points part of 2 Bins [%]	Data-points part of 3 Bins [%]
6.79	43.81	42.94	6.45

The bottom graph presents the total count of a single datapoint for the 3B-MAW analysis. Ideally, a single datapoint would be used 300 time. However, due primarily to *engine-off* events (e.g. during loading/unloading of the vehicle, driver lunch break, etc.), and to a lesser extent due to the range of exclusions specified under the 3B-MAW approach, some datapoints are not used at all for the 3B-MAW analysis, or are unequally weighted as individual datapoints appear in a diminishing number of windows at the start (*engine-on* event) and end (*engine-off* event) of a vehicle's operation. It can be noticed from Figure 2, that at the 280-minute mark only a single window exists due to very short engine operation (on the order of 300sec), and that no 300sec windows can be established during a series of intermittent, short (on the order of ~8min) engine operations while the vehicle is maneuvered to different locations at its warehouse yard to load goods. However, as expected for a longer- or long-haul vehicle application, the number of datapoints excluded due to *engine-off* events during the shift-day is less as seen from Table 5 above with only ~6.8% of the data being excluded. Nevertheless, the frequency of such *engine-off* events is strongly related to the specific route and operation of the vehicle. In comparison, Figure 3 below, which shows data representative of a category 6b vehicle performing food/beverage distribution in an urban/city environment, has a significantly higher fraction of data excluded (see Table 6) due to frequent engine start/stop events, resulting in a more pronounced unequal weighting of datapoints, as can been seen from the lower graph in Figure 3 showing the total count of individual datapoints used in the analysis.

Figure 2 also highlights another significant deficiency of the 3B-MAW approach. The yellow-shaded area is vehicle operation at constant ~55-60mph speed on a highway (see grey vehicle-speed line in lower graph). However, from inspection of the top graph of individual bin-membership count, this data is actually being attributed to Bin 1, the idle bin. It is clear from the vehicle speed trace that this type of operation is definitively <u>not</u> typical idle operation that should be compared to the idle emissions standard. During the yellow-highlighted portion of the route, the vehicle also is operating over a highway section with extended downhill operation (i.e., 600m over 20min operation at ~55-60mph), which results in very low engine power demand and thus, the normalized CO_2 mass rate drops on average below the 6% level over the 300sec MAWs, causing this operation to be inappropriately binned as '*idle*' operation. Overall, for the category 1a vehicle shown in Figure 2, data is allocated 13.3% to Bin 1, 39.5% to Bin 2 and 47.3% to Bin 3 over the entire shift-day.

As noted, data presented in Figure 3 for a food/beverage distribution application in an urban environment shows a highly transient vehicle-activity behavior based on vehicle speed. Based on vehicle dynamics, the dataset comprises 18.1% idle, 46.8% urban (<31mph), 11.6% rural (>31 and <46mph), and 23.6% highway (>46mph) operation. However, despite the significant fraction of idle and urban stop/go-type driving patterns, Bin 1 does not get populated <u>at all</u> for this vehicle. In fact, a majority of the data, 59%, is attributed to Bin 3. The remaining 41% falls into Bin 2. Similar results have been observed for all category 6a and 6b vehicles that are subject to highly transient urban vehicle operation with frequent stop/go and heavy acceleration between idle events. Furthermore, vehicles in this category experience frequent *engine-off* events during unloading/delivery of food/beverages, leaving insufficient time for accumulation of a 300 second window, resulting in a large fraction of the stop/go operation being discarded from the analysis (see red arrows in lower graph pointing to operations with no 300-second windows). Shortening the MAW duration would aid in addressing the incorrect assignment of datapoints to bins that are not intended to cover that specific type of vehicle operation. That is especially pronounced for highly transient operation such as observed for the delivery-type applications in categories 6a/6b.

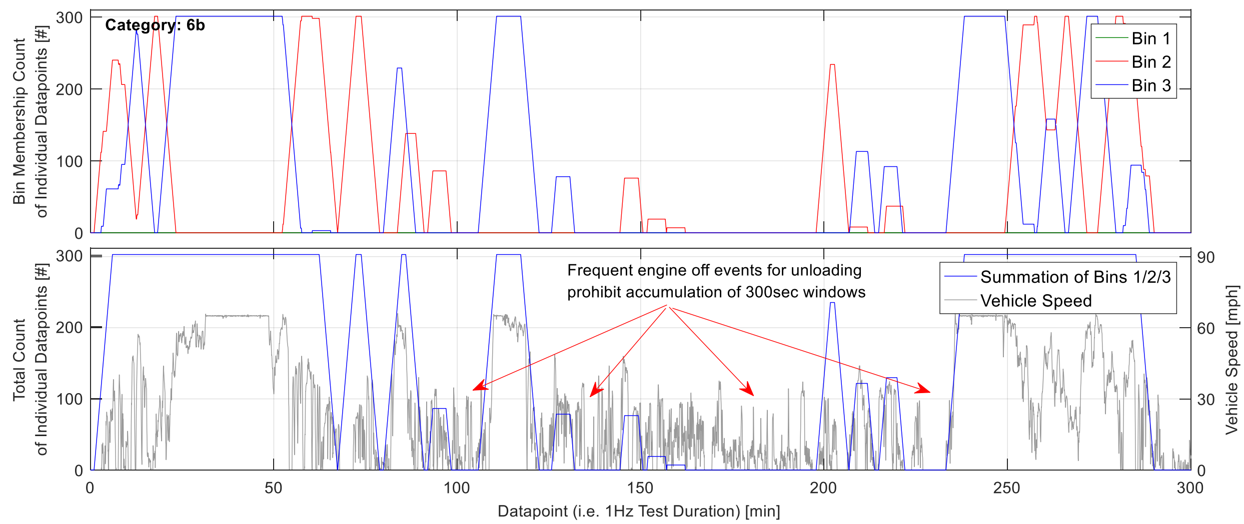


Figure 3: Bin membership count of individual datapoints to either of the three bins for a category 6b vehicle (i.e., beverage/food distribution); data represents excerpt of single day operation.

Table 6: Percentage count single data points appearing in either none (i.e., excluded), 1, 2 or 3 bins at the same time for a single category 6b vehicle; data represents a single day of operation.

Data-points excluded [%]	Data-points part of 1 Bin [%]	Data-points part of 2 Bins [%]	Data-points part of 3 Bins [%]
28.6	45.4	26.0	0.0

2.2 Data '*smearing*'-effects of the same emission datapoints across the proposed bins

This section analyzes how emissions data originating from different vehicle and engine operating conditions is distributed among bins leading to an overall '*smearing*'-effect across bin boundaries. This behavior of the 3B-MAW method is especially challenging for the effective optimization of emissions control-strategies, which are typically tailored towards specific engine operating conditions (e.g., idle operation will require a different strategy to efficiently reduce emissions as compared to high-load operation that experiences elevated exhaust gas temperatures). Thus, if an in-use data analysis method combines emissions rates from such different operating conditions into a single bin it will be, on one hand, difficult to design an adequate control strategy for a given engine operating condition, and on the other hand, problematic to evaluate the efficacy of that strategy using the in-use binning method.

To investigate those aspects of the 3B-MAW method, data within Bin 2 and Bin 3 were segregated based on the MAW-averaged (i.e., t_{MAW} = 300sec) vehicle speed into urban, rural and highway operation. Those three distinct speed ranges were established previously by the European Union's heavy-duty in-service conformity (ISC) testing program, and are frequently used to discern specific vehicle operating activity. *Urban* operation is defined as vehicle speeds ≤31mph; *rural* operation as vehicle speeds between > 31mph and ≤ 46.6mph; and finally, *highway* operation for vehicle speeds above 46.6mph. Figure 4 shows the distribution of daily sum-over-sum brake-specific NO_x emissions for a category 1a long-haul vehicle over 40 valid shift-days, and segregated into the three vehicle-speed-based activity domains. The top portion of Figure 4 presents results for Bin 3 (the medium/high load bin), whereas the bottom portion of Figure 4 shows results for Bin 2 (the low-load bin). The graphs also include the time-weighted route fractions within a given bin. The individual boxplots represent the 10th and 90th percentiles (whiskers), the 25th and 75th percentile (blue box), and the median (red line) and mean (black triangle) of the distribution of the 40 daily sum-over-sum *bs*NO_x emissions rates within a given activity domain. The light-green triangles represent the daily-averaged sum-over-sum *bs*NO_x for the entire bin, whereas the single dark-green triangle shows the global averaged sum-over-sum *bs*NO_x emissions rates over all 40 shift-days for the category 1a long-haul vehicle.

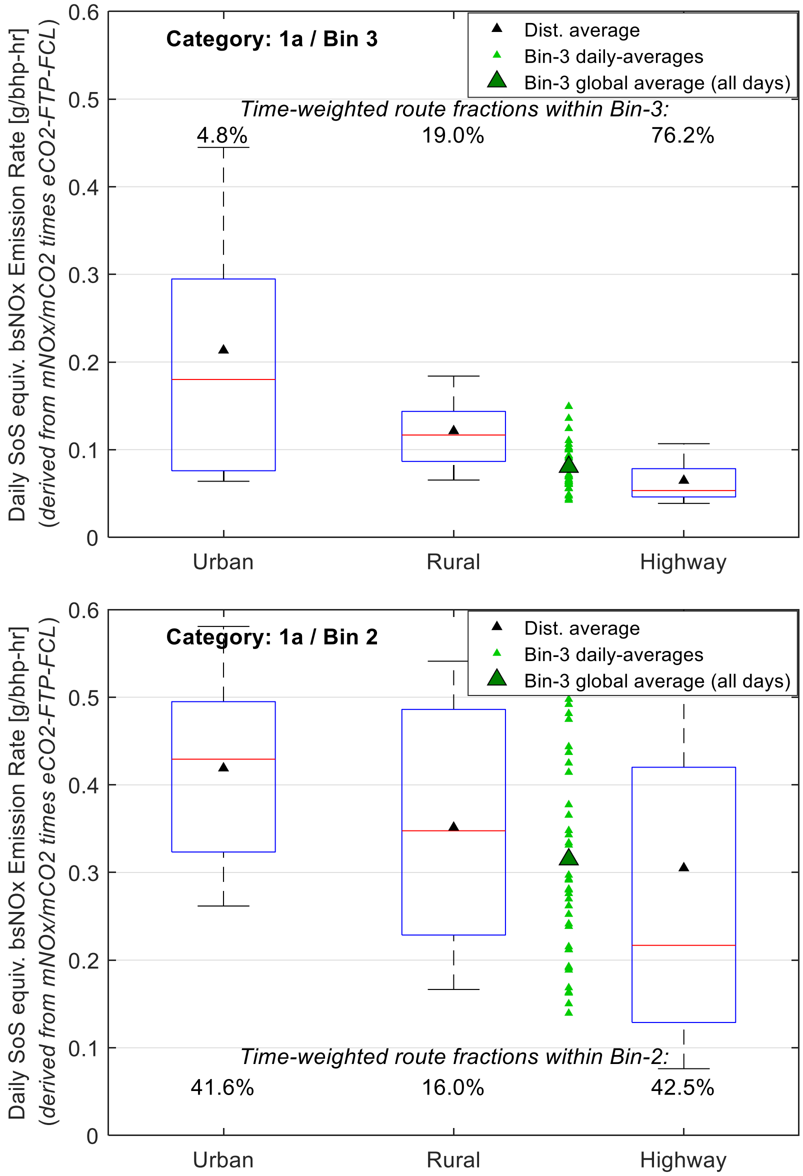


Figure 4: Bin-3 (top) and Bin-2 (bottom) emissions for 40 shift-days of a category 1a long-haul vehicle divided into urban, rural, highway operation based on MAW-averaged vehicle speed; box - 25th and 75th percentile, whiskers - 10th and 90th percentile, red line - median, black triangle - mean; light-green triangles - daily SoS-*bs*NO_x emissions; dark-green triangle - average of all daily SoS-*bs*NO_x emissions; urban ≤31mph, rural >31 & ≤ 46.6mph, highway > 46.6mph.

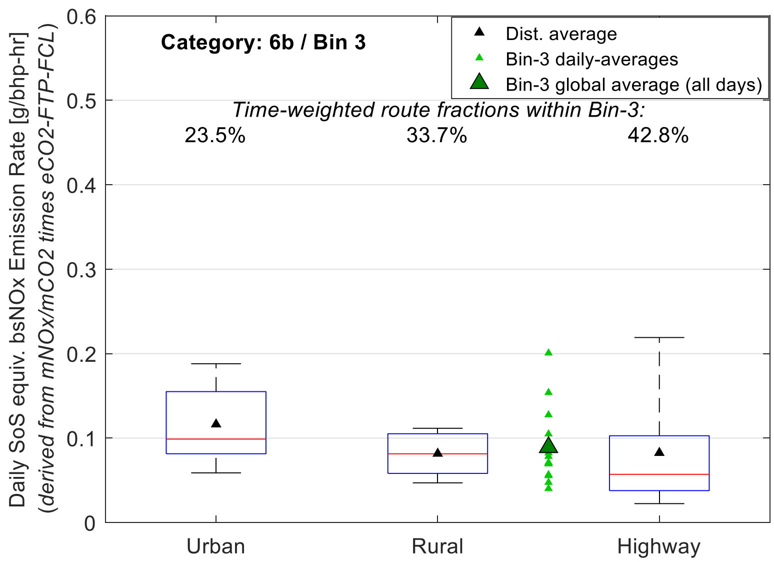
It is evident from Figure 4 (along with Table 7 below) that there is a significant amount of different vehicle dynamic activity that results in significantly different emissions rates, which nonetheless are ultimately all lumped together into a single bin by the 3B-MAW method. For example, Bin 3 includes up to ~5% of urban vehicle operation that exhibits nearly 3.5 times the *bs*NO_x emissions rates measured for the highway operation. In contrast, Bin 2, the low-load bin, includes 42.5% highway operation at vehicle speeds > 46.6mph, which exceeds the amount of urban operation at lower vehicle speeds (≤ 31mph) that would be considered typical of low-load vehicle operation. Furthermore, the spread observed in the daily sum-over-sum *bs*NO_x emissions rates, indicated by the light-green triangles, clearly shows the significant variability in bin-specific emissions results (i.e. 10^{th} to 90^{th} percentile range for Bin 2: 0.321g/bhp-hr and Bin 3: 0.056g/bhp-hr) that the same vehicle is experiencing between day-to-day operation when analyzed with the 3B-MAW method.

Table 7: Statistics for Bin-3 (left) and Bin-2 (right) emissions for 40 shift-days of a category 1a long-haul vehicle divided into urban, rural, highway operation based on MAW-averaged vehicle speed.

Bin 3 Results		Urban	Rural	Highway	Bin 3		Bin 2 Results		Urban	Rural	Highway	Bin 2
		[g/bhp-hr]	[g/bhp-hr]	[g/bhp-hr]	[g/bhp-hr]				[g/bhp-hr]	[g/bhp-hr]	[g/bhp-hr]	[g/bhp-hr]
Average	[%]	0.213	0.121	0.065	0.080		***Average***	[%]	0.419	0.351	0.305	0.315
10th Percentile	[%]	0.064	0.065	0.039	0.052		***10th Percentile***	[%]	0.262	0.167	0.076	0.166
25th Percentile	[%]	0.076	0.087	0.046	0.063		***25th Percentile***	[%]	0.323	0.229	0.129	0.214
50th Percentile	[%]	0.180	0.117	0.053	0.080		***50th Percentile***	[%]	0.429	0.348	0.217	0.292
75th Percentile	[%]	0.295	0.144	0.078	0.093		***75th Percentile***	[%]	0.495	0.486	0.420	0.396
90th Percentile	[%]	0.445	0.184	0.107	0.108		***90th Percentile***	[%]	0.581	0.541	0.635	0.487

Similarly, Figure 5 below (along with Table 8) shows sum-over-sum brake-specific *bs*NO_x emissions results for 14 shift-days of operation of a category 6b food/beverage delivery vehicle, which is characterized by lower-speed operation with increased transient and frequent stop/go activity, for Bin 3 (top of figure) and Bin 2 (bottom of figure). While Bin 2 shows a majority of the vehicle activity (~77%) in the urban speed domain, the medium/high-load Bin 3 only contains ~43% of data in the highway speed range, and about a quarter of the time (23.5%) is in the lowest urban speed domain. This is attributed to the previously discussed drawback of the 300-second MAW, which sorts highly transient vehicle operation into Bin 3. Figure 5 also shows large variability in sum-over-sum *bs*NO_x emissions rates over different shift-day operations of the same vehicle in both Bin 2 and Bin 3.

Overall, these results highlight again that the 3B-MAW method is not an effective approach for discriminating between different types of vehicle operations from a NO_x emissions perspective.



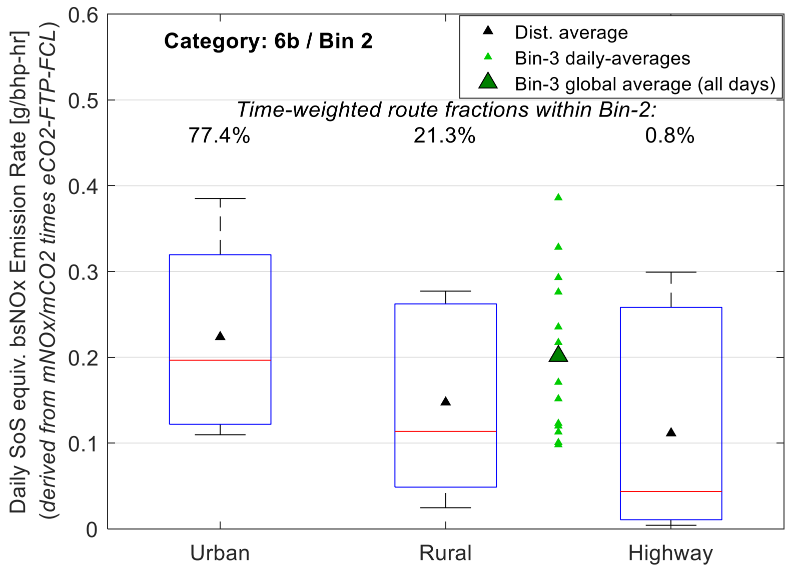


Figure 5: Bin-3 (top) and Bin-2 (bottom) emissions for 14 shift-days of a category 6b food/beverage distribution vehicle divided into urban, rural, highway operation based on MAW-averaged vehicle speed; box - 25th and 75th percentile, whiskers - 10th and 90th percentile, red line - median, black triangle - mean; light-green triangles - daily SoS-*bs*NO_x emissions; dark-green triangle - average of all daily SoS-*bs*NO_x emissions; urban ≤31mph, rural >31 & ≤ 46.6mph, highway > 46.6mph.

Table 8: Statistics for Bin-3 (left) and Bin-2 (right) emissions for 40 shift-days of a category 6b food/beverage distribution vehicle divided into urban, rural, highway operation.

Bin 3 Results		Urban	Rural	Highway	Bin 3	Bin 2 Results		Urban	Rural	Highway	Bin 2
		[g/bhp-hr]	[g/bhp-hr]	[g/bhp-hr]	[g/bhp-hr]			[g/bhp-hr]	[g/bhp-hr]	[g/bhp-hr]	[g/bhp-hr]
Average	[%]	0.116	0.082	0.083	0.089	*Average*	[%]	0.224	0.148	0.111	0.201
10th Percentile	[%]	0.059	0.047	0.022	0.047	*10th Percentile*	[%]	0.110	0.025	0.004	0.100
25th Percentile	[%]	0.081	0.058	0.038	0.057	*25th Percentile*	[%]	0.122	0.049	0.011	0.120
50th Percentile	[%]	0.099	0.081	0.057	0.075	*50th Percentile*	[%]	0.196	0.114	0.044	0.189
75th Percentile	[%]	0.155	0.105	0.103	0.105	*75th Percentile*	[%]	0.320	0.262	0.258	0.276
90th Percentile	[%]	0.188	0.112	0.219	0.159	*90th Percentile*	[%]	0.385	0.277	0.299	0.334

Figure 6 and Figure 7 show a comparison of MAW-averaged *bs*NO_x emissions rates versus MAW-normalized CO_2 mass rates for Bin 2 and Bin 3 from a category 1a long-haul, and a category 1b food/beverage delivery vehicle, respectively, operated over a single shift-day (i.e. with t_{MAW} = 300sec). Individual MAW emissions results are colored by the MAW-averaged after-treatment outlet temperatures ($T_{AT\text{-}out}$), where $T_{AT\text{-}out}$ below 200°C are shown in green, $T_{AT\text{-}out}$ between 200°C and 250°C in blue, and $T_{AT\text{-}out}$ above 250°C in red.

As can be seen, the window-specific *bs*NO_x levels vary widely with no discernable patterns and with no clear trends between exhaust temperatures and NO_x levels for a specific Bin. This indicates that the 3B-MAW method is <u>not</u> effective at segregating different types of vehicle operations from a NO_x perspective, but instead leads to a random spreading or smearing of NO_x emissions data across the bins.

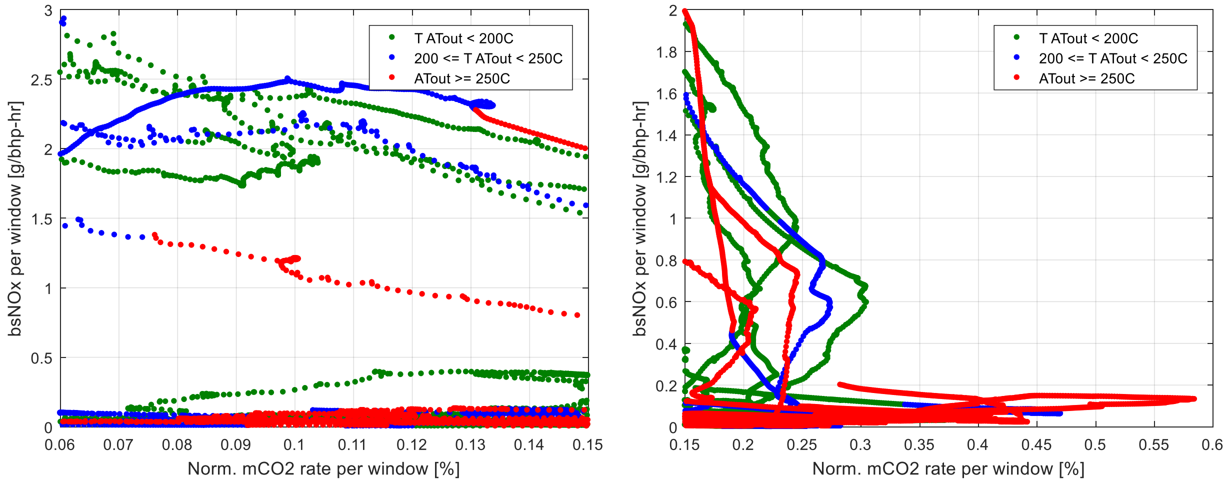


Figure 6: Comparison of window-averaged *bs*NO_x emissions rate vs. window-normalized CO_2 mass rate for a category 1a (i.e., long haul) vehicle; left - Bin 2, right - Bin 3 results; operation over a single shift-day; colors indicate MAW-averaged AT-out temperatures.

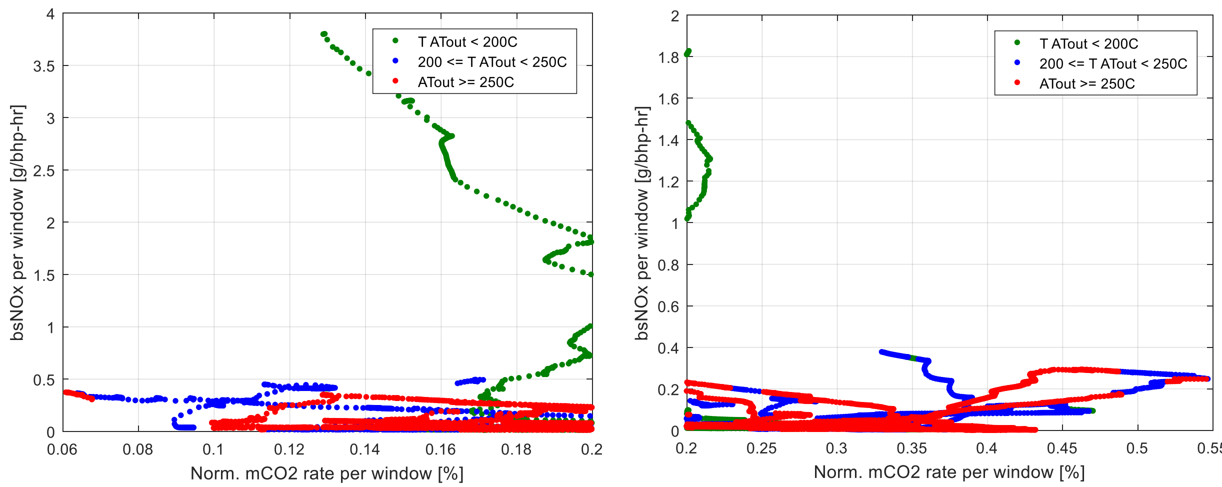


Figure 7: Comparison of window-averaged *bs*NO_x emissions rate vs. window-normalized CO_2 mass rate for a category 6b (i.e., food/beverage delivery) vehicle; left - Bin 2, right - Bin 3 results; operation over a single shift-day; colors indicate MAW-averaged AT-out temperatures.

2.3 Disproportional weighting of certain emission results over others

Figure 8 represents the cumulative count of individual datapoints in windows for each of the three bins versus the normalized route duration for a single long-haul vehicle of category 1a operating over a single day. The data shows that only 30% of the individual datapoints falling into Bin 3 (the medium/high load bin) are weighted equally in the 300 windows that are part of Bin 3. In comparison, 40% of the individual datapoints are only present in 150 of the windows that fall into Bin 3 and, thus, those datapoints are only weighted half as much as the datapoints that fall into 300 individual windows as part of Bin 3. Overall, Figure 8 shows that for this specific vehicle, representative of typical category 1a-type operation, only 6.9%, 14.6%, and 30%, of datapoints in Bins 1, 2, and 3, respectively, are weighted equally 300 times. The rest of the datapoints are weighted differently and disproportionally.

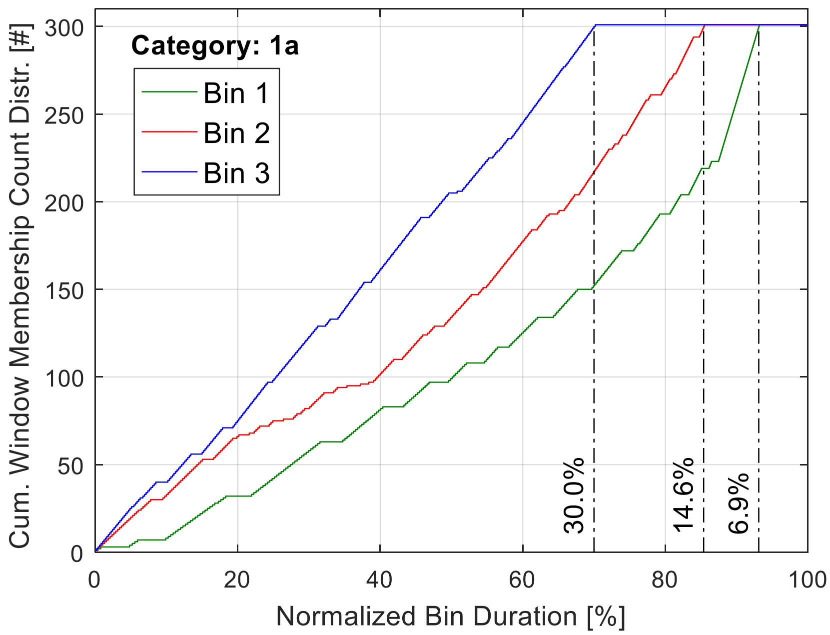


Figure 8: Cumulative count of window membership of individual datapoints for the three different bins over the normalized route duration for a single vehicle of category 1a; data represents a single day of operation.

Similarly, Figure 9 shows the cumulative count of individual datapoints in windows for each of the three bins for a category 6b vehicle used for food/beverage distribution. Even though the vehicle spends 18.1% of the time idling and approximately 46.8% of the time under urban driving conditions (<31mph), no data was included in Bin 1 over the entire 5-hour operating period of this vehicle. Similar to the category 1a vehicle (see Figure 8), only 31.8% of the datapoints are equally weighted 300 times in Bin 3, and only a fraction of the datapoints falling into Bin 2, specifically 6.3%, are equally weighted. Remarkably, assessment of idle bin emissions (i.e., Bin 1) would not be possible for this delivery-vehicle due to the complete lack of data in Bin 1, despite the dataset meeting the 3B-MAW validity criteria of a minimum of three hours of non-idle operation and average engine power over the shift-day above a 10% minimum level.

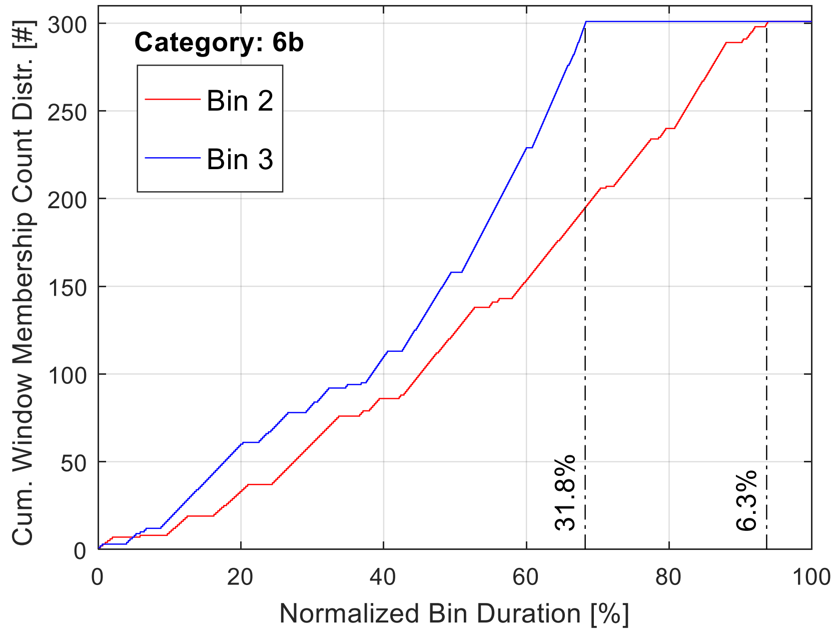


Figure 9: Cumulative count of window membership of individual datapoints for the three different bins over the normalized route duration for a single vehicle of category 6a; data represents a single day of operation; no data allocated to Bin 1 (i.e., Idle Bin).

Figure 10 and Figure 11 present a '*Monte-Carlo*'-like simulation for two sample vehicles, one each from category 1a and 6b. The analysis was conducted by combining the beginning and end of the datasets to form a continuous vector of data, and then randomly selecting a starting point for the moving-averaging window procedure. The objective behind this approach is to demonstrate whether a given dataset always provides the same result, since the same amount of emissions-mass and work produced is being analyzed. A rational MAW-based approach would exhibit that consistent behavior. Conversely, possible deficiencies in a MAW-analysis method could lead to an increased variability in results, as opposed to a stable and repeatable result no matter where in the dataset the analysis is started.

To assess those aspects of CARB's 3B-MAW, 1000 MAW staring points within a single-day dataset were randomly selected and the 3B-MAW binning process was performed. In addition, the MAW size (i.e., t_{MAW}) was altered between 30, 60, 180, and 300 seconds to assess the impact of the data averaging time. Figure 10 shows the results for the category 1a long-haul vehicle for the three bins and the four MAW durations. It can be seen from the boxplots for this vehicle that the interquartile distributions are narrow and, in most cases, coincide with the median of the distributions (exceptions are Bin 2, 180/300sec and Bin 3, 180/300sec). In general, there is a trend in reducing the 10th to 90th percentile range (the whiskers of the plotted data) as the MAW duration is shortened from 300sec to 30sec for all three bins. However, that is most pronounced for Bin 1, and to a lesser degree for Bin 2. Bin 3 does not show significant variability as a function of MAW starting point in the dataset, and the distributions mostly converge on the sample average. On the other hand, it was observed that the time-weighted membership for Bin 1 increased from 3.8% to 12%, while it was reduced from 63% to 54% for Bin 3 when the MAW duration was shortened between 300sec to 30sec. Overall, this analysis shows that the 3B-MAW method does not yield stable and repeatable results.

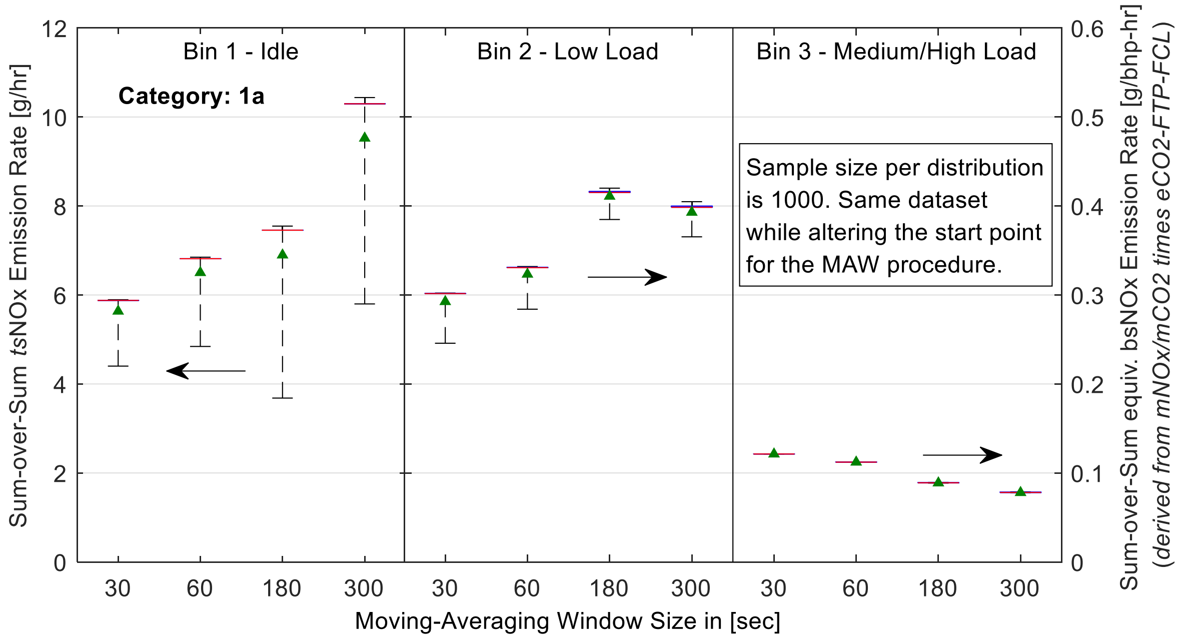


Figure 10: 'Monte-Carlo' simulation of single day operation of category 1a (long-haul) vehicle while randomly selecting starting point for moving-averaging window; sample size is 1000; MAW size varying between 30, 60, 180, and 300sec; red line - median, blue box - 25th-75th, whiskers - 10th-90th.

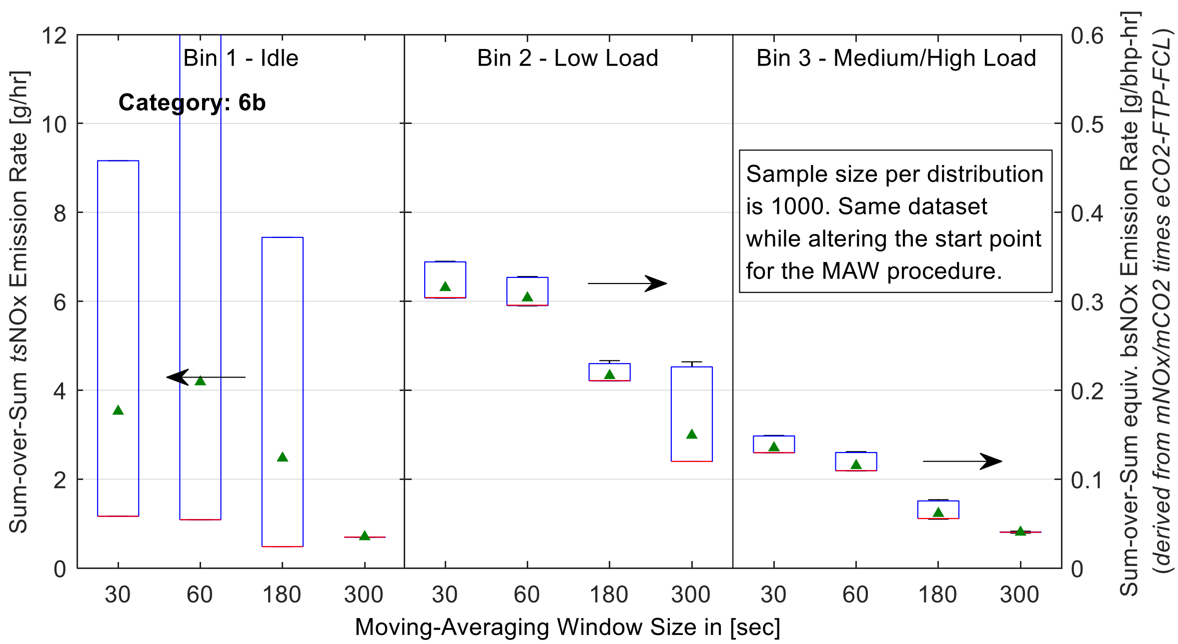


Figure 11: 'Monte-Carlo' simulation of single day operation of category 6b (food/beverage delivery) vehicle while randomly selecting starting point for moving-averaging window; sample size is 1000; MAW size varying between 30, 60, 180, and 300sec; red line - median, blue box - 25th-75th, whiskers - 10th-90th.

Figure 11 above shows the same analysis for a single operating day of a category 6b food/beverage delivery vehicle. For this vehicle, even more variability was observed, especially for the Bin 1 and Bin 2 results. The 300-second MAW shows highest variability for the Bin 2 results with sum-over-sum *bs*NO_x emission rates ranging between 0.12 to 0.24g/bhp-hr. Shortening the MAW duration resulted in a reduction in the interquartile range for Bin 2, but also resulted in a slight increase in average emission rates. The largest variability as a function of MAW starting point in the dataset was observed for Bin 1, with emissions ranging from 0.5g/hr to 13g/hr. The time-weighted membership for Bin 1 increased from 1.2% to ~9% with durations ranging between 300 seconds and 30 seconds, while at the same time, operation in Bin 3 was reduced from ~76% to ~49% for the same change in MAW duration. Again, in this case, the 3B-MAW method did not yield stable and repeatable results.

2.4 Concatenating of data across engine-off events and exclusions will result in unrepresentative binning of data yielding wide spreads in the binned results

Figure 12 below highlights the limitations and concerns relating to data concatenation over extended '*engine-off*' events, including due to vehicle loading/unloading and driver lunch breaks, etc., during a shift-day. Depending on the vehicle application, such events can be on the order of 1-hour or more during which the after-treatment system will be cooling down substantially. The current 3B-MAW approach would concatenate data during the specified exclusion conditions, such as PEMS-zeroing events and '*engine-off'* events, if they occur during the shift-day where emissions are collected. Figure 12 shows data from a portion of a single shift-day operation of a category 1a vehicle, highlighting exhaust gas temperature measured at the SCR outlet location (see red line) along with the total count of times an individual datapoint is used in the 3B-MAW method. It is evident from Figure 12 that during the three '*engine-off*' events as the vehicle was unloaded, which lasted on the order of 25 to 48minutes, the exhaust thermal state varied significantly with exhaust temperatures dropping on the order of 100°C between engine shut-off and restart. Data concatenation and continuous MAW calculations across these types of discontinuities would lead to very different engine and after-treatment conditions being lumped and averaged together into single MAW windows. That is not a scientifically reasonable approach.

Concatenating data for short (~<1min) discontinuity events (e.g. short '*engine-off*' events) can be useful to reduce the impact of unequal data-weighting before and after an '*engine-off*' event (shown as the down- and up-ramps seen from the total membership count, as represented by the blue line). During those short '*engine-off*' events, the thermal state of the engine and after-treatment will not change appreciably. However, during extended '*engine-off*' events, as experienced for loading/unloading or for driver lunch breaks, etc., data concatenation will lead to a combination of wholly dissimilar states.

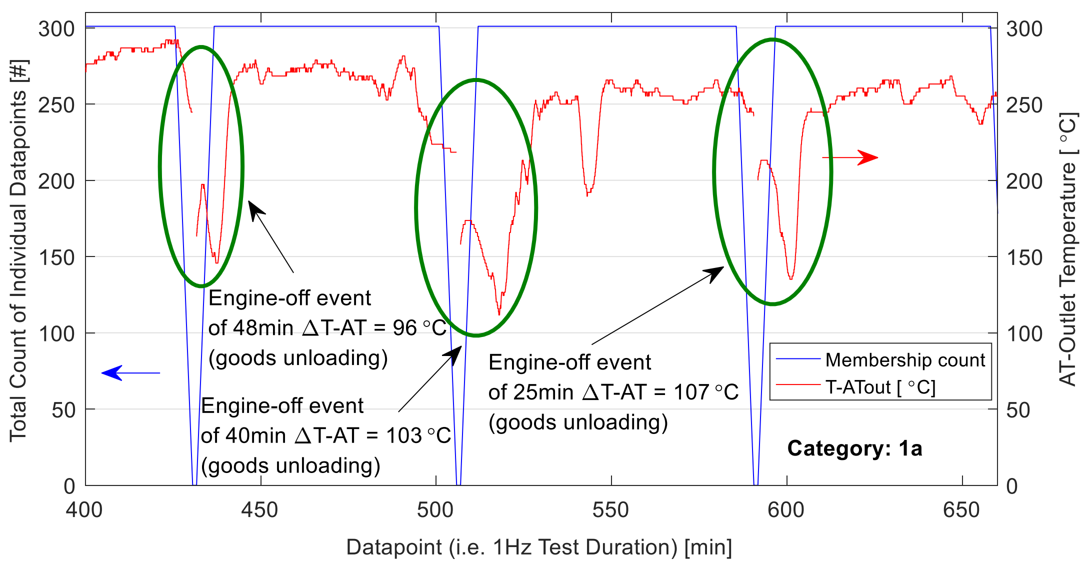


Figure 12: Exhaust after-treatment temperature (i.e., SCR catalyst outlet) as a function of 3B-MAW transitions during time-limited engine-off events (i.e., unloading/loading of vehicle).

2.5 Lack of discernable correlation among the datapoints that end-up binned together under the 3B-MAW approach

The following box and whisker plots (Figure 13 through Figure 27) show the lack of any consistent correlation among the binned data for any of the assessed vehicle categories, even if different MAW durations between 300 seconds and 30 seconds are applied. No consistent trends in the data are evident and the range of data variability between different days of operation for a given vehicle, as well as across vehicles and categories, was observed to be extremely high. These data provide additional evidence that the 3B-MAW method does not yield consistent or reasonably corelated binned emission results suitable for the application of separate emission standards.

In general, the figures shown in this section represent the distribution of individual shift-day sum-over-sum results on a time-specific (*ts*NO_x) basis for Bin 1, and on a brake-specific (*bs*NO_x) basis for Bin 2 and Bin 3. The blue box represents the 25th to 75th percentile, the whiskers the 10th to 90th percentile, the red line the median, and the green triangles the mean of the data distribution across vehicles in all categories (for Figure 13 through Figure 15) or for specific vehicle categories (Figure 16 through Figure 27).

Specifically, Figure 13 through Figure 15 show a comparison of normalized NO_x emissions results for the 3B-MAW method for Bin 1, Bin 2, and Bin 3, respectively, across the different vehicle categories. Data is presented in a 'normalized' way, using the globally-averaged shift-day sum-over-sum NO_x emissions rate for all test vehicles as the normalization factor. This is done to assess the relative variability of bin results within a vehicle category and to compensate for vehicle-to-vehicle changes in absolute emissions rate levels. In addition, the MAW duration is varied between t_{MAW} = 300sec (white background), t_{MAW} = 30sec (red background), and binning of individual 1Hz datapoints (green background). The analysis shows that significant variabilities exist within categories for all three 3B-MAW bins, for all vehicle categories evaluated, and that no clear trends are observable as a function of category-specific activity patterns.

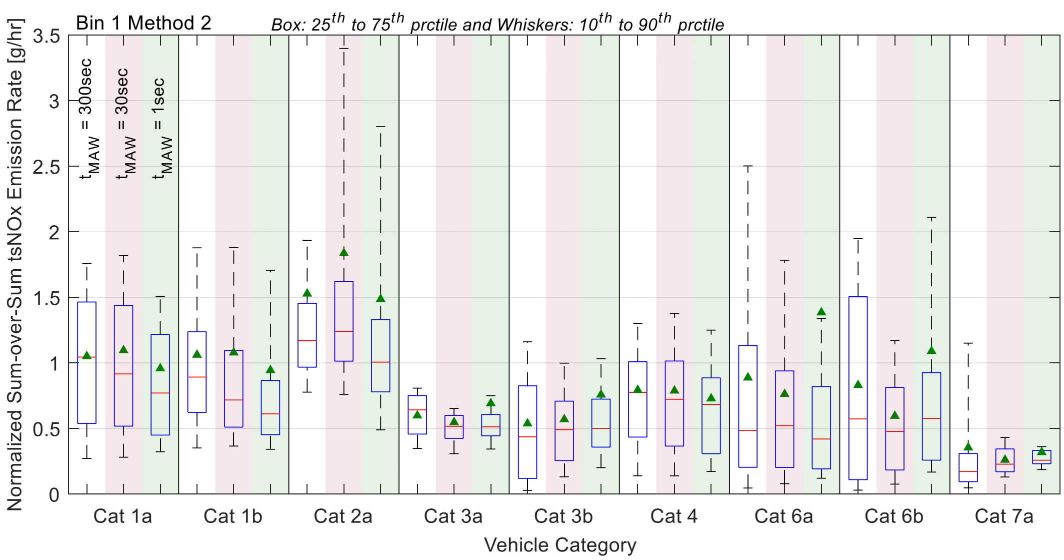


Figure 13: Comparison of normalized Bin 1 emissions results across vehicles in all categories with MAW durations between 300 (white) and 30sec (red) as well as no windowing (1Hz-binning, green).

Variabilities are found to be highest for vehicle categories 2a, 3b, 6a, and 6b, which are characterized by increased idle operation, PTO operation (category 3b), and transient and frequent stop/go activity, particularly for Bin 2 and Bin 3. That is consistent with the previously mentioned observation that highly transient activity can lead to significant portions of data moving to the medium/high-load Bin 3 from idle Bin 1 due to the 300-second MAW approach. The same vehicle categories also show the highest ‘skewness’ with the distribution means being significantly larger than the medians, indicating that diverse daily activity can lead to drastically different emissions rates for the same vehicle.

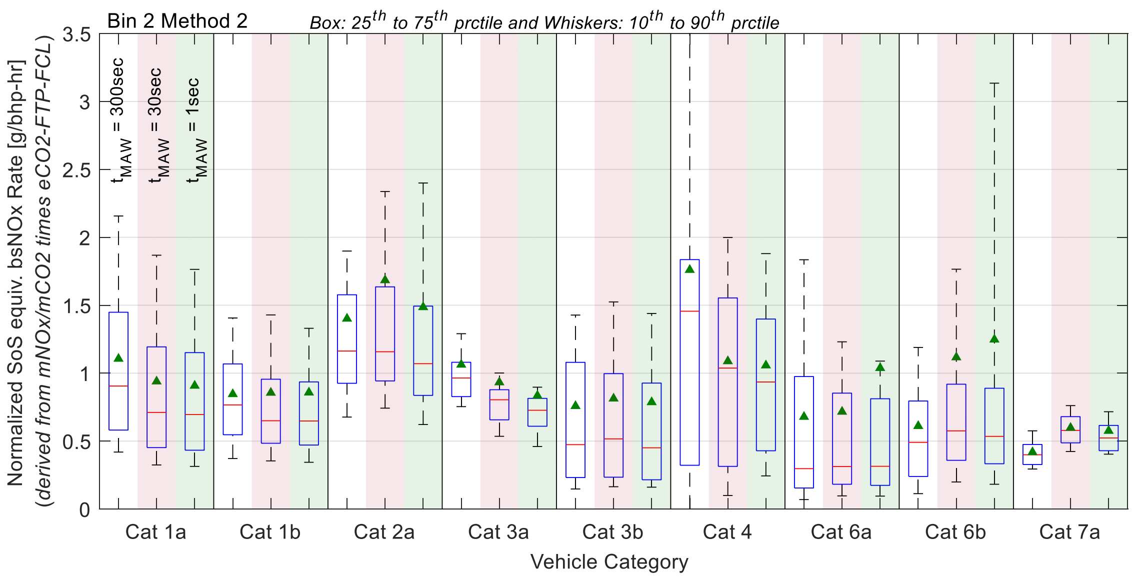


Figure 14: Comparison of normalized Bin 2 emissions results across vehicles in all categories with MAW durations between 300 (white) and 30sec (red) as well as no windowing (1Hz-binning, green).

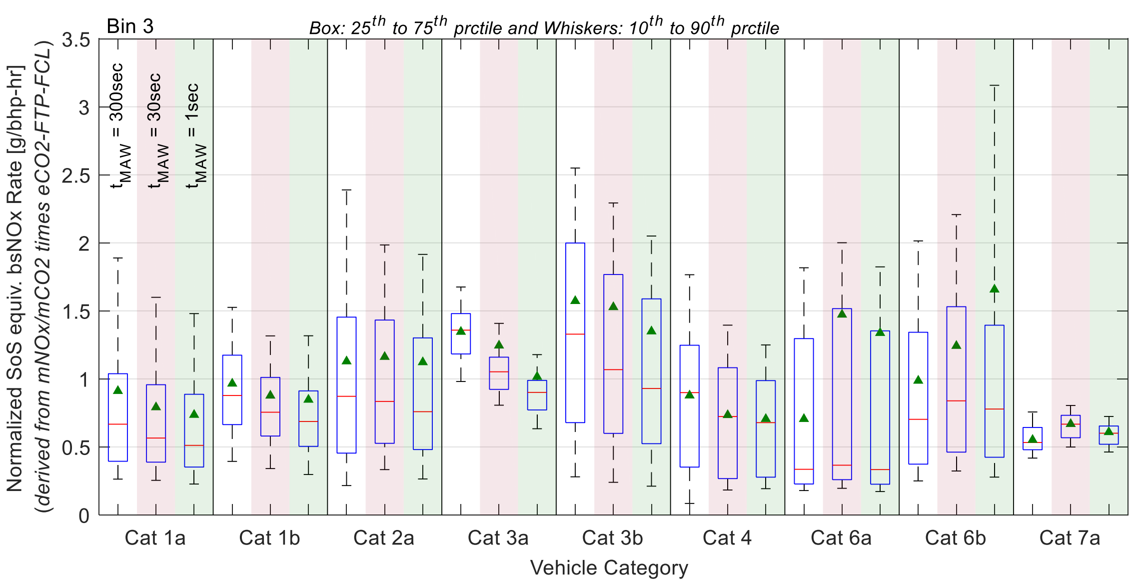


Figure 15: Comparison of normalized Bin 3 emissions results across vehicles in all categories with MAW durations between 300 (white) and 30sec (red) as well as no windowing (1Hz-binning, green).

Figure 16 through Figure 18 show a comparison of bin-specific sum-over-sum NO_x emissions rates for category 1a long-haul vehicles for three different MAW durations between 300 seconds and 30 seconds. Data shown in these graphs are presented on an absolute scale.

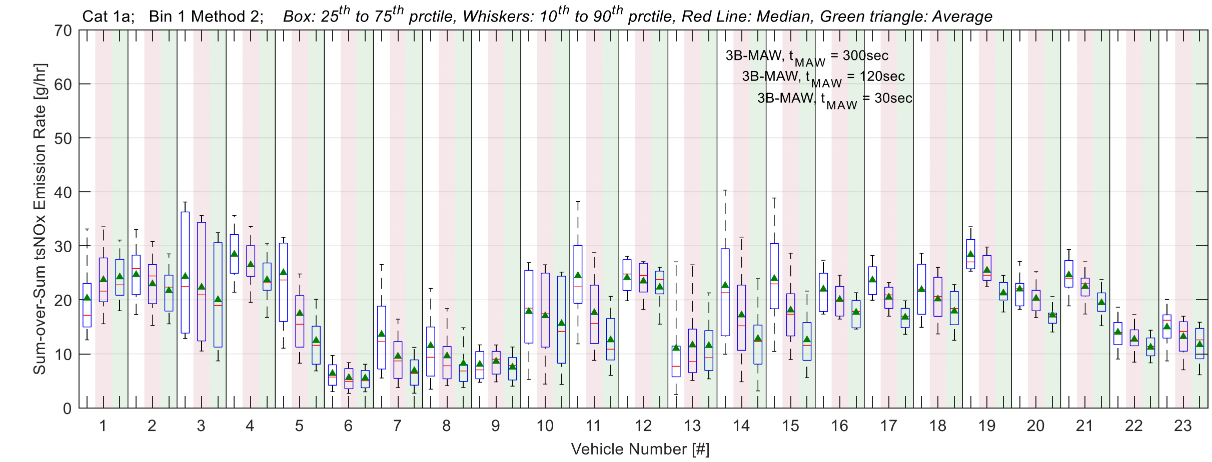


Figure 16: Comparison of Bin 1 emissions results for vehicles in category 1a (i.e., long haul); Analysis methods: t_{MAW} =300sec (white), t_{MAW} = 120sec (red), t_{MAW} = 30sec (green).

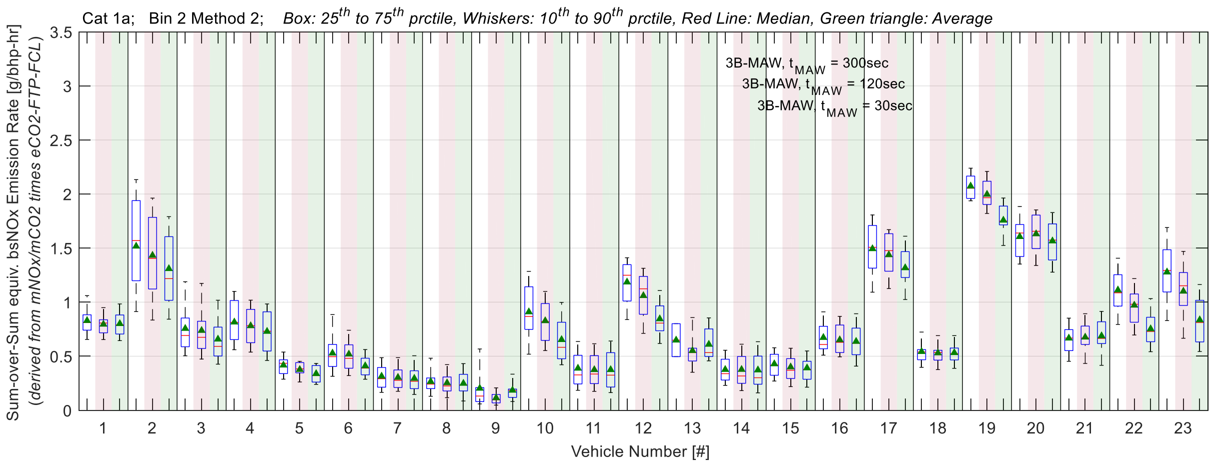


Figure 17: Comparison of Bin 2 emissions results for vehicles in category 1a (i.e., long haul); Analysis methods: t_{MAW} =300sec (white), t_{MAW} = 120sec (red), t_{MAW} = 30sec (green).

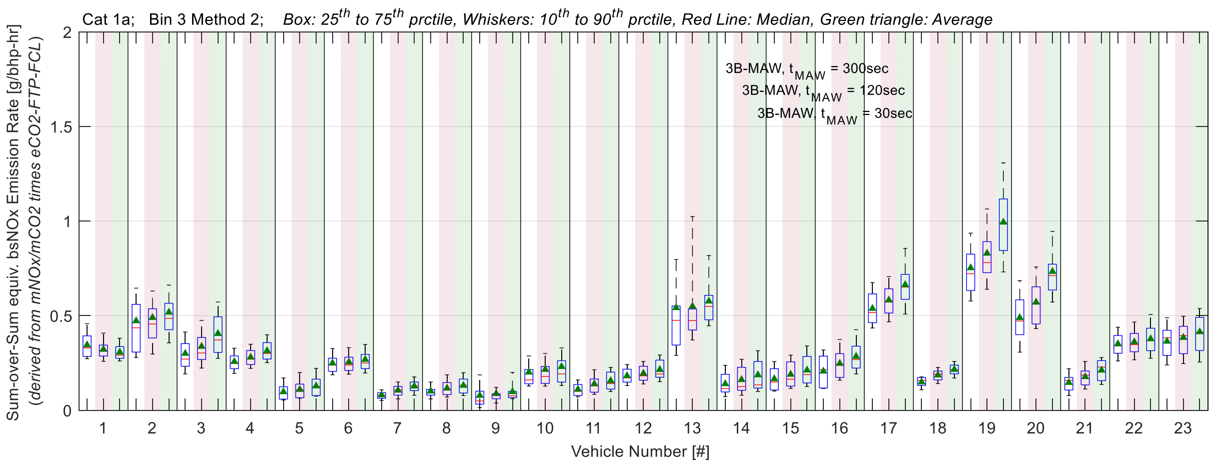


Figure 18: Comparison of Bin 3 emissions results for vehicles in category 1a (i.e., long haul); Analysis methods: t_{MAW} =300sec (white), t_{MAW} = 120sec (red), t_{MAW} = 30sec (green).

Figure 19 through Figure 21 show a comparison of bin-specific sum-over-sum NO_x emissions rates for category 1b short-haul vehicles for three different MAW durations between 300 seconds and 30 seconds.

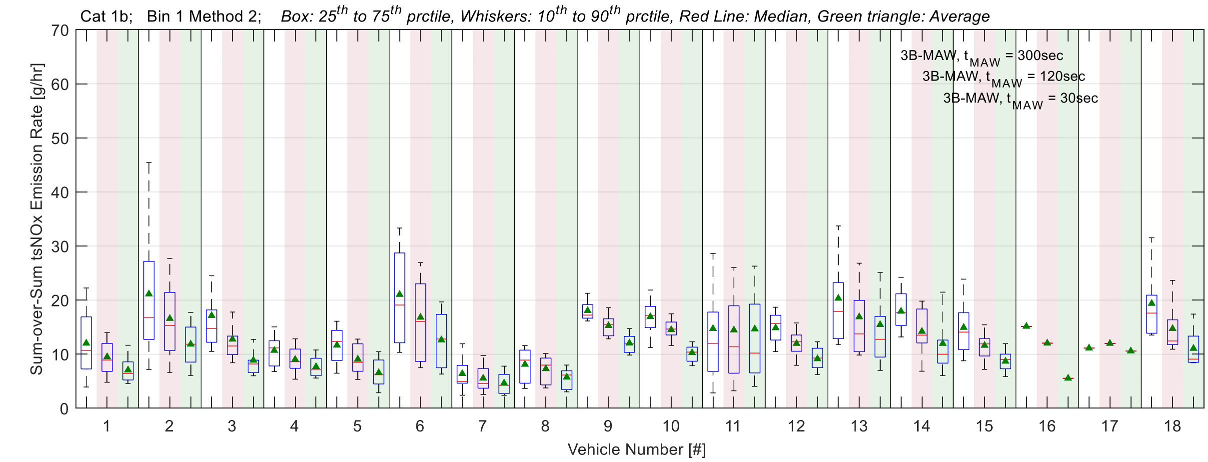


Figure 19: Comparison of Bin 1 emissions results for vehicles in category 1b (i.e., short haul); Analysis methods: t_{MAW} =300sec (white), t_{MAW} = 120sec (red), t_{MAW} = 30sec (green).

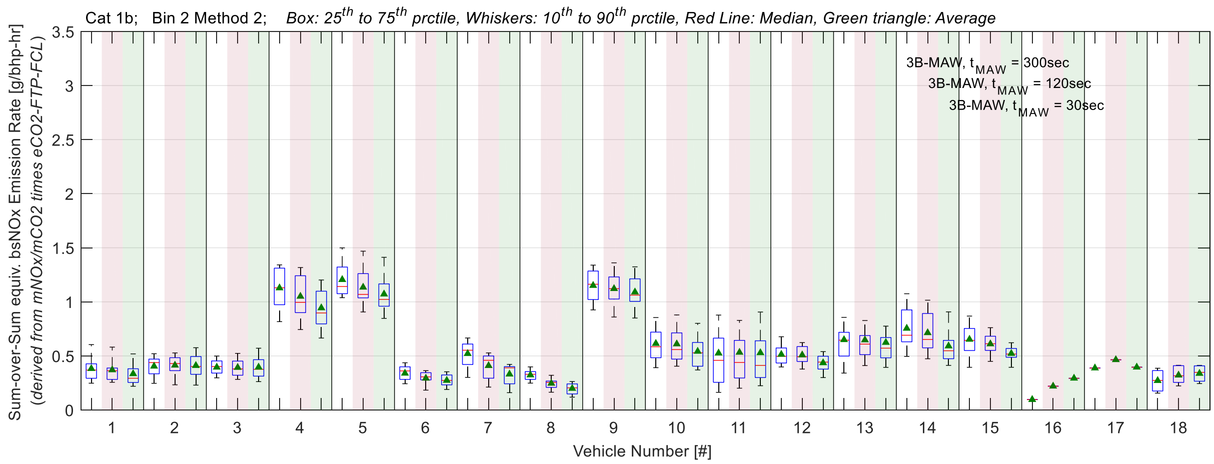


Figure 20: Comparison of Bin 2 emissions results for vehicles in category 1b (i.e., short haul); Analysis methods: t_{MAW} =300sec (white), t_{MAW} = 120sec (red), t_{MAW} = 30sec (green).

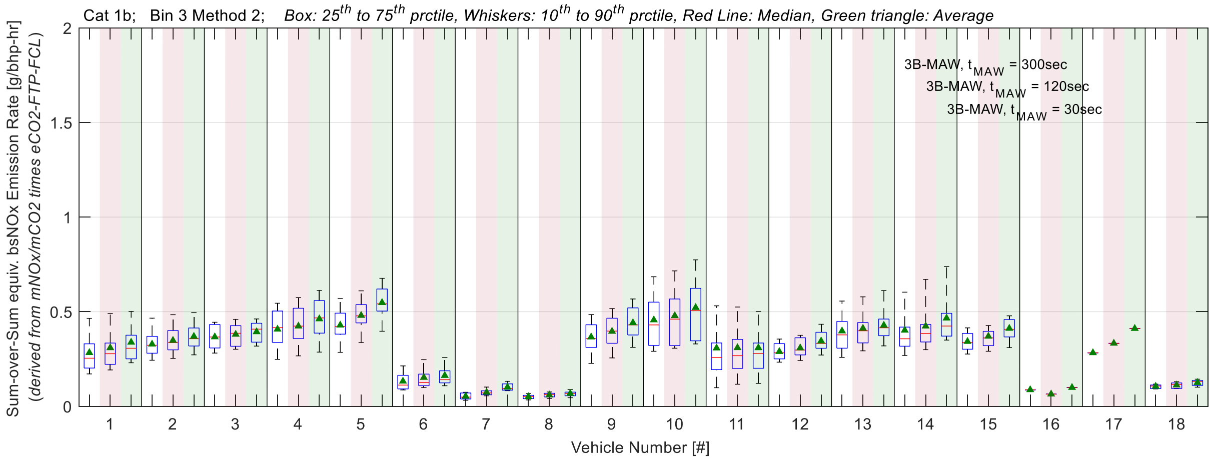


Figure 21: Comparison of Bin 3 emissions results for vehicles in category 1b (i.e., short haul); Analysis methods: t_{MAW} =300sec (white), t_{MAW} = 120sec (red), t_{MAW} = 30sec (green).

Figure 22 through Figure 24 show a comparison of bin-specific sum-over-sum NO_x emissions rates for category 6b food/beverage vehicles for three different MAW durations between 300 seconds and 30 seconds. Since typical shift-days of category 6b vehicles are characterized by relatively short and transient operation with frequent *engine-off* durations between vehicle activity for unloading of food/beverage goods at stores, the 3-hours non-idle validity criteria of the 3B-MAW method excludes many of the shift-days. Thus, for results shown in Figure 22 through Figure 24, only the 10% minimum average power of P_{max} criteria was applied, thereby allowing a larger number of shift-day NO_x emissions to be represented in this analysis.

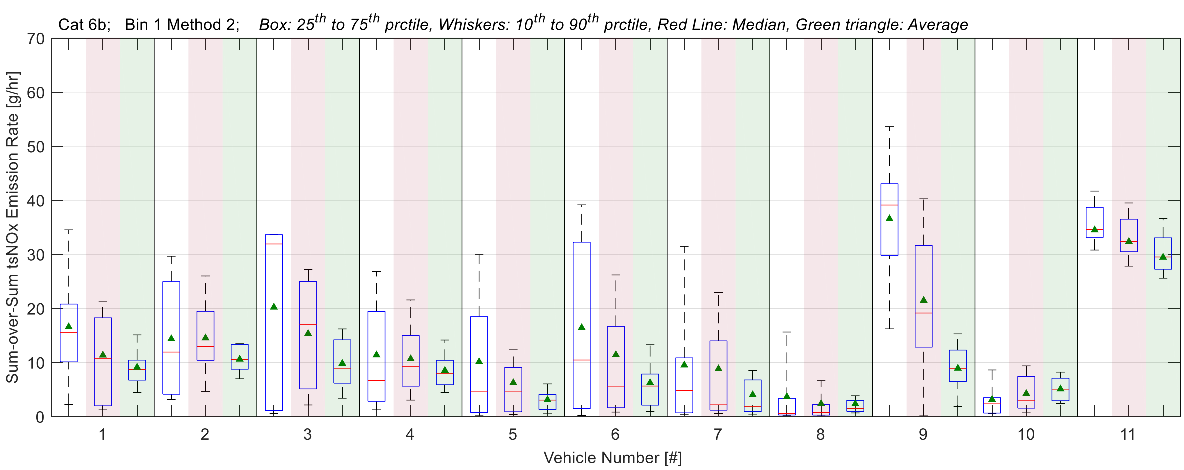


Figure 22: Comparison of Bin 1 emissions results for vehicles in category 6b (i.e., delivery truck); Analysis methods: t_{MAW} =300sec (white), t_{MAW} = 120sec (red), t_{MAW} = 30sec (green); with only 10% minimum power validity criteria applied.

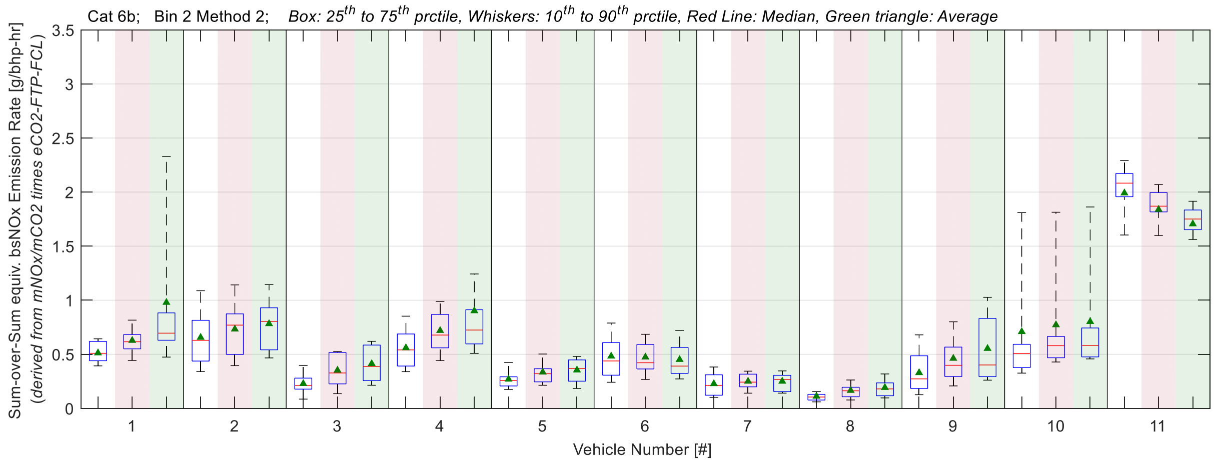


Figure 23: Comparison of Bin 2 emissions results for vehicles in category 6b (i.e., delivery truck); Analysis methods: t_{MAW} =300sec (white), t_{MAW} = 120sec (red), t_{MAW} = 30sec (green); with only 10% minimum power validity criteria applied.

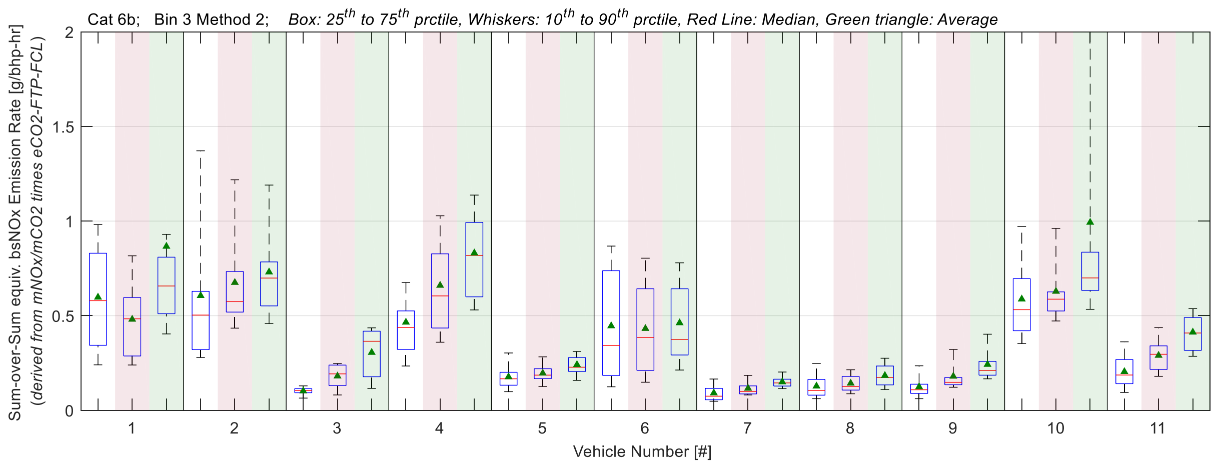


Figure 24: Comparison of Bin 3 emissions results for vehicles in category 6b (i.e., delivery truck); Analysis methods: t_{MAW} =300sec (white), t_{MAW} = 120sec (red), t_{MAW} = 30sec (green); with only 10% minimum power validity criteria applied.

In contrast, Figure 25 through Figure 27 show the comparison of the bin-specific sum-over-sum NO_x emissions rates for the same category 6b food/beverage vehicles, with both the 10% minimum average power of P_{max} and the 3-hours non-idle criteria being applied to invalidate individual shift-day datasets. As seen from Figure 25 through Figure 27, all individual shift-days for four vehicles (i.e., vehicles 1, 2, 4, and 10) out of the total population of 11 vehicles were completely invalidated due to the 3-hours non-idle criteria, and for two vehicles (i.e., vehicles 3 and 9) only a single valid shift-day remained. Also, for vehicle 9, no shift-day was actually valid for a MAW duration of 300 seconds.

Figure 25: Comparison of Bin 1 emissions results for vehicles in category 6b (i.e., delivery truck); Analysis methods: t_{MAW} =300sec (white), t_{MAW} = 120sec (red), t_{MAW} = 30sec (green); with 10% minimum power and 3hrs non-idle validity criteria applied.

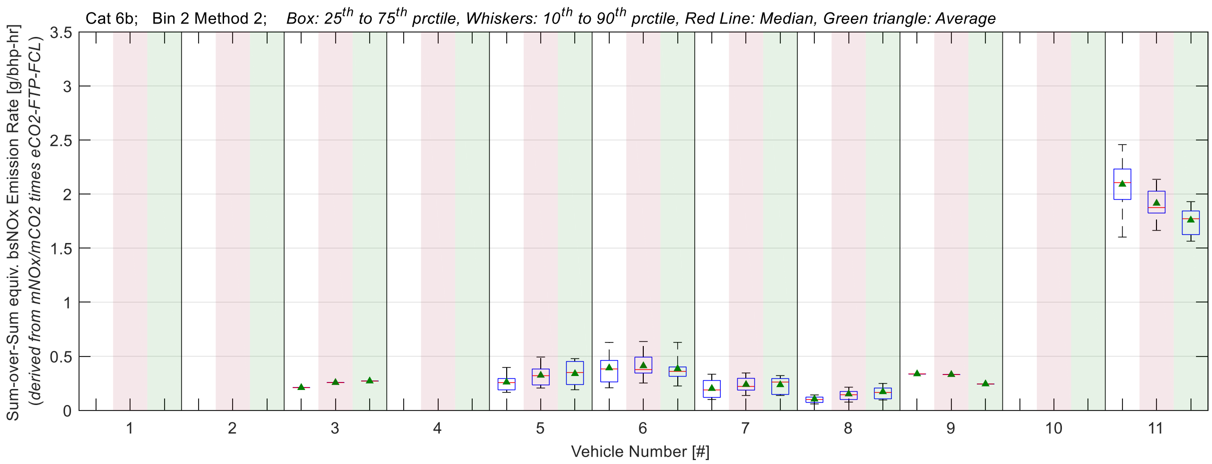


Figure 26: Comparison of Bin 2 emissions results for vehicles in category 6b (i.e., delivery truck); Analysis methods t_{MAW} =300sec (white), t_{MAW} = 120sec (red), t_{MAW} = 30sec (green); with 10% minimum power and 3hrs non-idle validity criteria applied.

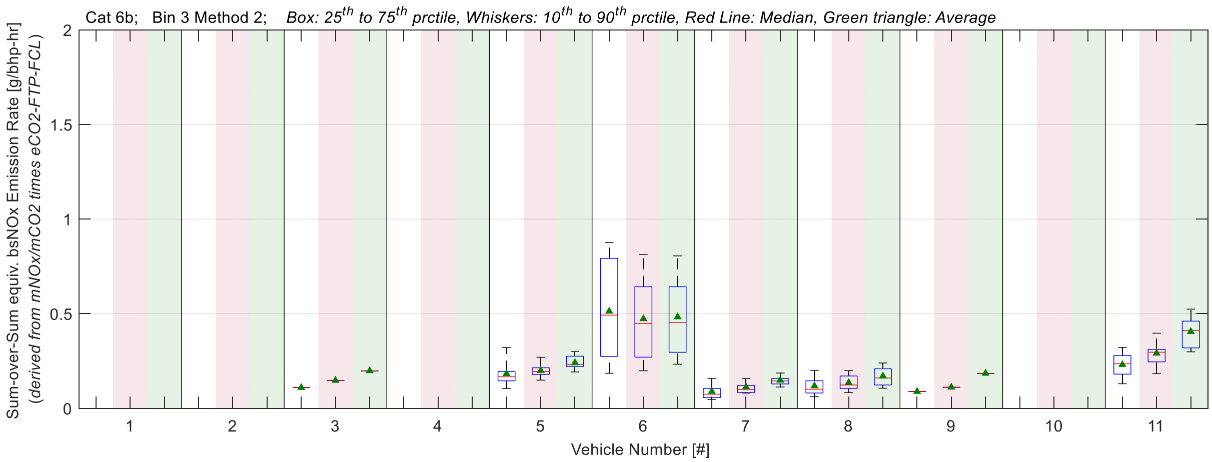


Figure 27: Comparison of Bin 3 emissions results for vehicles in category 6b (i.e., delivery truck); Analysis methods t_{MAW} =300sec (white), t_{MAW} = 120sec (red), t_{MAW} = 30sec (green); with 10% minimum power and 3hrs non-idle validity criteria applied.

2.6 Normalized CO_2 mass rate distributions for all vehicle categories

This section presents the window-averaged normalized CO_2 mass-rate distributions for each vehicle category in comparison to the 3B-MAW bin boundary limits (i.e., Bin 1 ≤ 6%, Bin 2 > 6% and ≤ 20%, Bin 3 > 20%). Each distribution is based on the number of vehicles for a given category listed in Table 1 and contains only data from shift-days that comply with the 3B-MAW validity criteria of 10% minimum average power over the entire shift-day, and 3-hours of non-idle operation, and are calculated for a MAW duration of 300 seconds. Table 9 shows the averages and percentiles of the normalized CO_2 mass rate distributions for each individual category. In addition, the Table summarizes the total percentage of valid windows falling into Bins 1, 2 or 3. The coloration of the data indicates bins with increased percentages of valid windows (shades of green), while bins with lower counts of windows are colored in shades of red. Finally, the bottom part of Table 9 shows the percentage of the total CO_2 mass attributed to one of the three bins. In contrast, Table 10 shows the distributions and total percentage of valid windows and CO_2 mass attributed to bins for the actual second-by-second data. Of particular significance, categories 2a, 3a, 6a, and 6b indicate how a significant portion of idle/creep, Bin 1-type operation is moved to the medium/high-load Bin 3 by the MAW approach.

Table 9: Window-normalized CO_2 mass rate statistics (w/ t_{MAW} = *300sec*) for each vehicle category (top table); total percentages of valid windows assigned to a given Bin 1, 2, or 3 based on norm. CO_2 mass rate (center table); percentages of total CO_2 mass assigned to a given bin; color-scale - larger percentages are shaded greener, while smaller percentages are shaded reddish.

Parameter	Vehicle Categories								
	1a	1b	2a	3a	3b	4	6a	6b	7a
Number of test vehicles	26	23	17	5	6	8	8	15	1
	Window-Averaged Normalized CO_2 [%]								
Average [%]	20.03	22.96	10.29	21.40	17.25	17.09	17.56	19.46	14.16
10th Percentile [%]	2.53	5.03	3.52	4.50	6.37	2.39	3.66	4.74	8.46
25th Percentile [%]	4.15	11.95	3.99	7.58	8.09	2.76	6.76	9.92	10.22
50th Percentile [%]	17.20	21.40	4.69	20.31	13.54	7.15	15.35	18.88	12.91
75th Percentile [%]	30.04	30.62	11.90	30.74	24.35	27.82	25.85	25.96	16.51
90th Percentile [%]	44.85	42.29	28.49	39.82	33.04	43.22	34.80	35.08	22.04
	Total percentage of valid windows falling into Bins 1, 2, and 3								
Prct. Windows in Bin 1 [%]	31.13	12.54	61.72	19.18	6.45	47.40	21.72	13.46	2.00
Prct. Windows in Bin 2 [%]	24.45	33.02	21.00	30.17	58.93	17.20	40.60	40.92	84.55
Prct. Windows in Bin 3 [%]	44.42	54.44	17.27	50.65	34.62	35.39	37.68	45.62	13.46
	Percentage of total CO_2 mass falling into Bins 1, 2, and 3								
Total CO2 mass in Bin 1 [%]	4.93	2.16	24.46	3.88	2.06	8.49	5.03	2.88	0.67
Total CO2 mass in Bin 2 [%]	15.88	19.39	22.38	17.22	37.41	12.58	28.73	28.20	74.88
Total CO2 mass in Bin 3 [%]	79.19	78.45	53.16	78.89	60.53	78.93	66.24	68.92	24.46

As can be seen, CARB's proposed bin boundaries are misaligned with the actual in-use vehicle operations shown in Table 10. This is another result showing the inadequate manner in which the 3B-MAW method reflects and segregates actual in-use operations of HDOH vehicles. The misalignment becomes very clear when the MAW-based data are plotted against the second-by-second data derived from the in-use vehicles (see Figure 28 trough Figure 36).

Table 10: Actual 1Hz CO_2 mass rate statistics for each vehicle category (top table); total percentages of valid data assigned to a given Bin 1, 2, or 3 based on norm. CO_2 mass rate (center table); percentages of total CO_2 mass assigned to a given bin; color-scale - larger percentages are shaded greener, while smaller percentages are shaded reddish.

Parameter	Vehicle Categories								
	1a	1b	2a	3a	3b	4	6a	6b	7a
Number of test vehicles	26	23	17	5	6	8	8	15	1
	Window-Averaged Normalized CO_2 [%]								
Average [%]	19.44	21.28	11.82	20.93	17.31	21.27	17.04	18.15	13.19
10th Percentile [%]	0.00	0.00	0.85	0.00	4.42	0.00	1.99	0.00	0.00
25th Percentile [%]	2.22	2.94	3.45	2.58	6.26	2.31	3.55	3.28	2.75
50th Percentile [%]	6.28	11.99	4.11	8.69	9.13	4.07	7.01	7.34	5.85
75th Percentile [%]	31.95	34.48	11.19	34.82	20.90	34.70	27.02	29.44	18.95
90th Percentile [%]	55.85	57.91	36.75	59.69	45.11	73.50	46.96	52.76	38.33
	Total percentage of valid windows falling into Bins 1, 2, and 3								
Prct. Windows in Bin 1 [%]	49.32	41.79	67.24	45.64	22.64	54.97	45.05	40.90	50.74
Prct. Windows in Bin 2 [%]	14.12	17.24	14.43	14.61	51.63	11.05	23.96	26.93	25.15
Prct. Windows in Bin 3 [%]	36.56	40.96	18.33	39.75	25.74	33.98	30.99	32.17	24.11
	Percentage of total CO_2 mass falling into Bins 1, 2, and 3								
Total CO2 mass in Bin 1 [%]	5.44	3.88	17.71	4.27	4.47	5.59	8.35	5.01	9.14
Total CO2 mass in Bin 2 [%]	8.95	10.20	13.98	8.23	29.56	6.19	14.60	14.84	20.24
Total CO2 mass in Bin 3 [%]	85.61	85.92	68.31	87.49	65.97	88.22	77.05	80.14	70.62

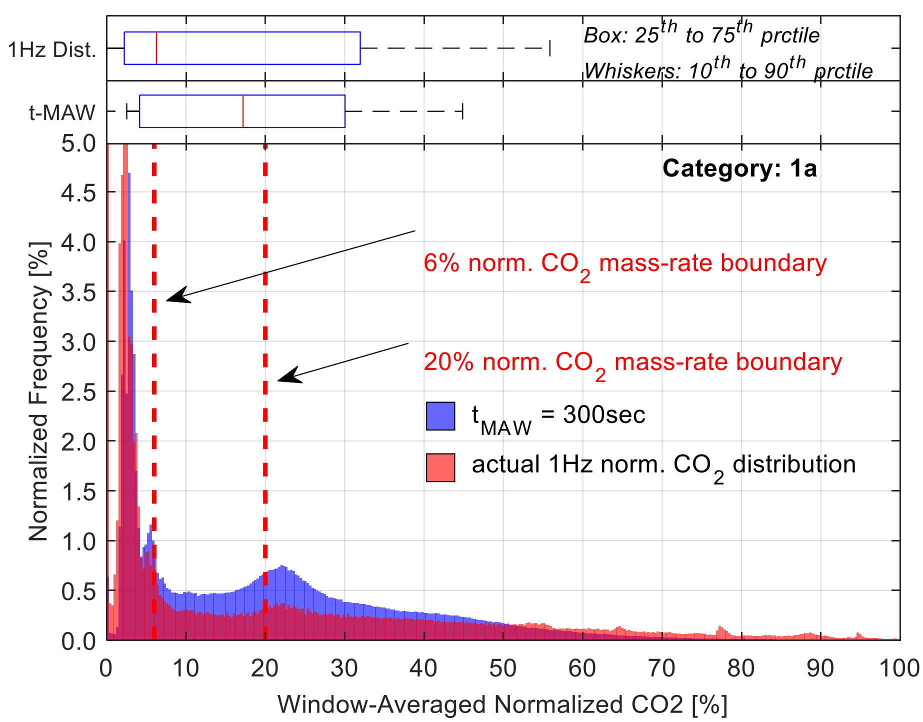


Figure 28: Window-averaged (w/ t_{MAW} = *300sec*, blue dist.) vs. actual 1Hz (red dist.) normalized CO_2 mass rate distributions for category 1a (i.e., long haul) vehicles.

Exhibit G

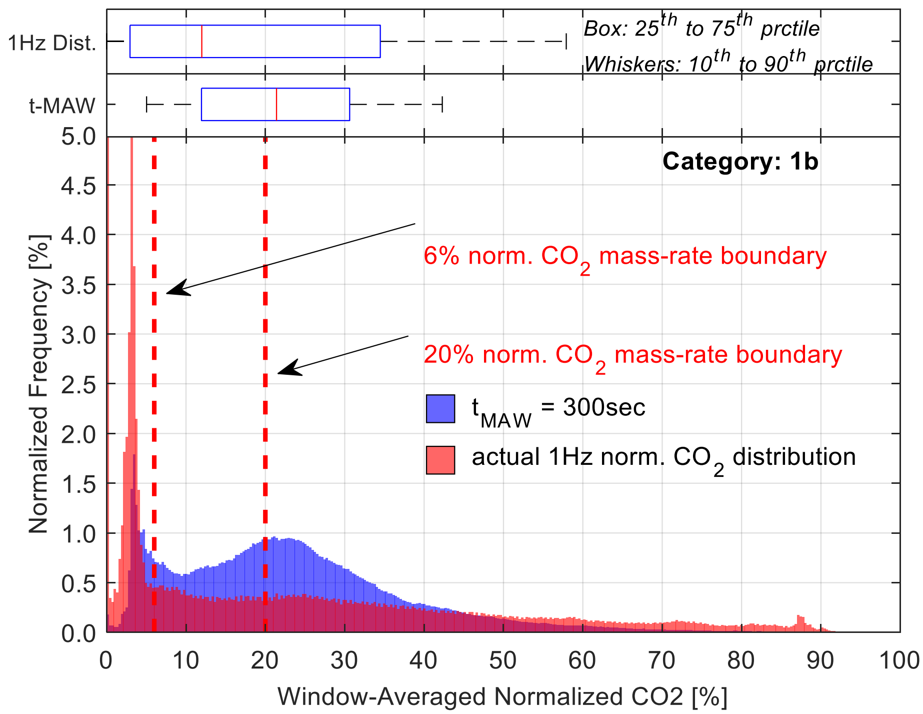


Figure 29: Window-averaged (w/ t_{MAW} = *300sec*, blue dist.) vs. actual 1Hz (red dist.) normalized CO_2 mass rate distributions for category 1b (i.e., short haul) vehicles.

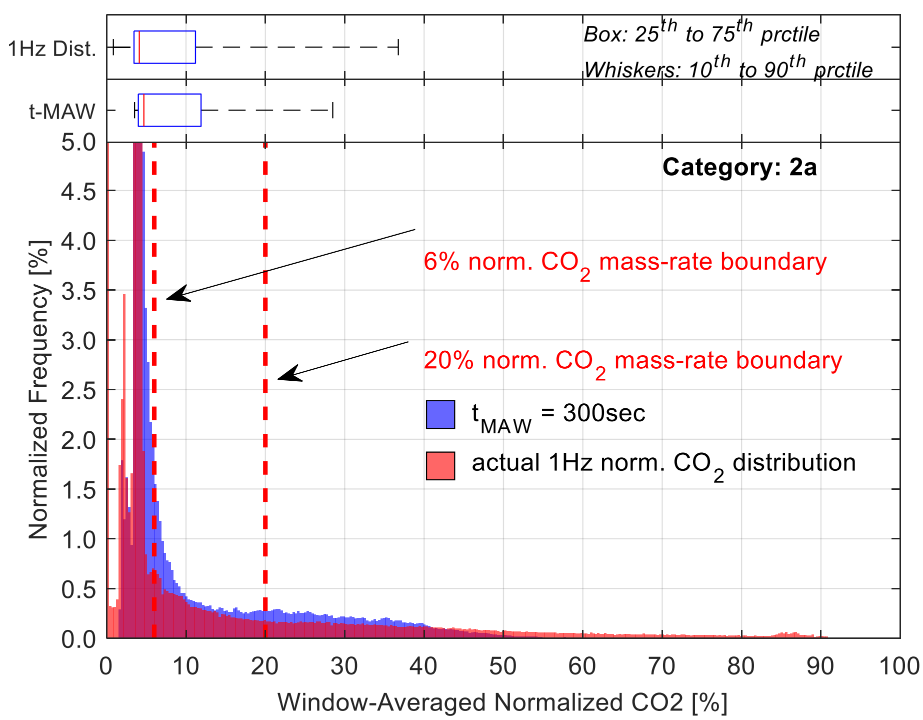


Figure 30: Window-averaged (w/ t_{MAW} = *300sec*, blue dist.) vs. actual 1Hz (red dist.) normalized CO_2 mass rate distributions for category 2a (i.e., port drayage) vehicles.

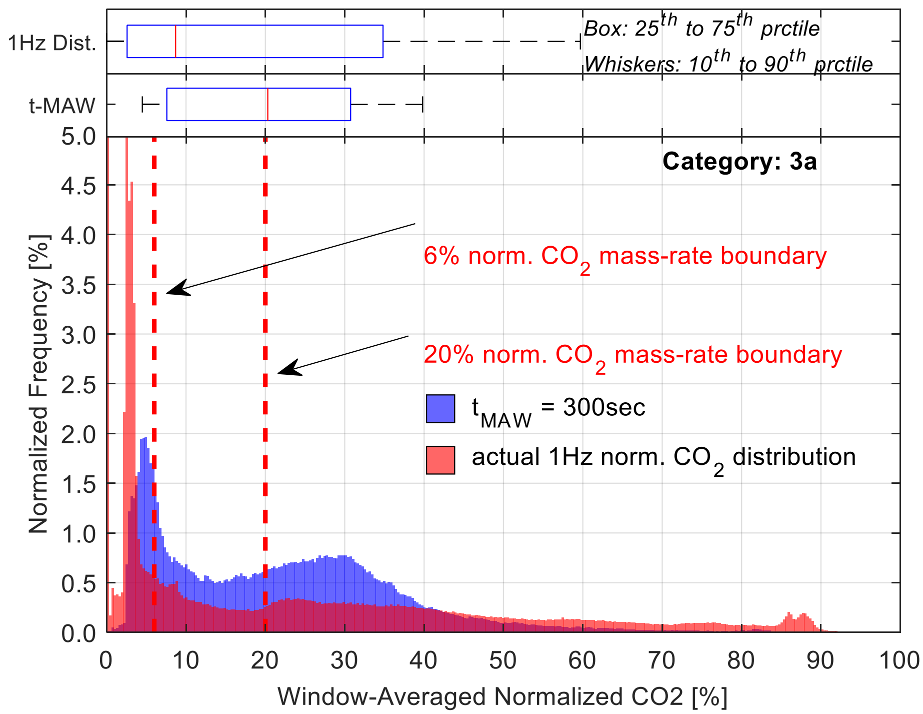


Figure 31: Window-averaged (w/ t_{MAW} = *300sec*, blue dist.) vs. actual 1Hz (red dist.) normalized CO_2 mass rate distributions for category 3a (i.e., tractor construction heavy) vehicles.

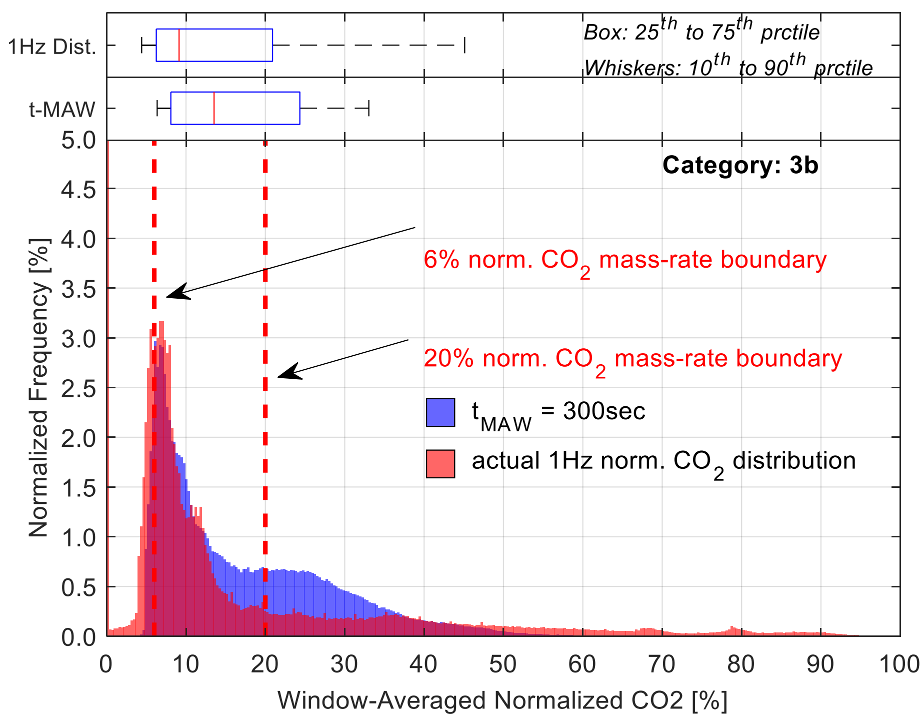


Figure 32: Window-averaged (w/ t_{MAW} = *300sec*, blue dist.) vs. actual 1Hz (red dist.) normalized CO_2 mass rate distributions for category 3b (i.e., cement mixer) vehicles.

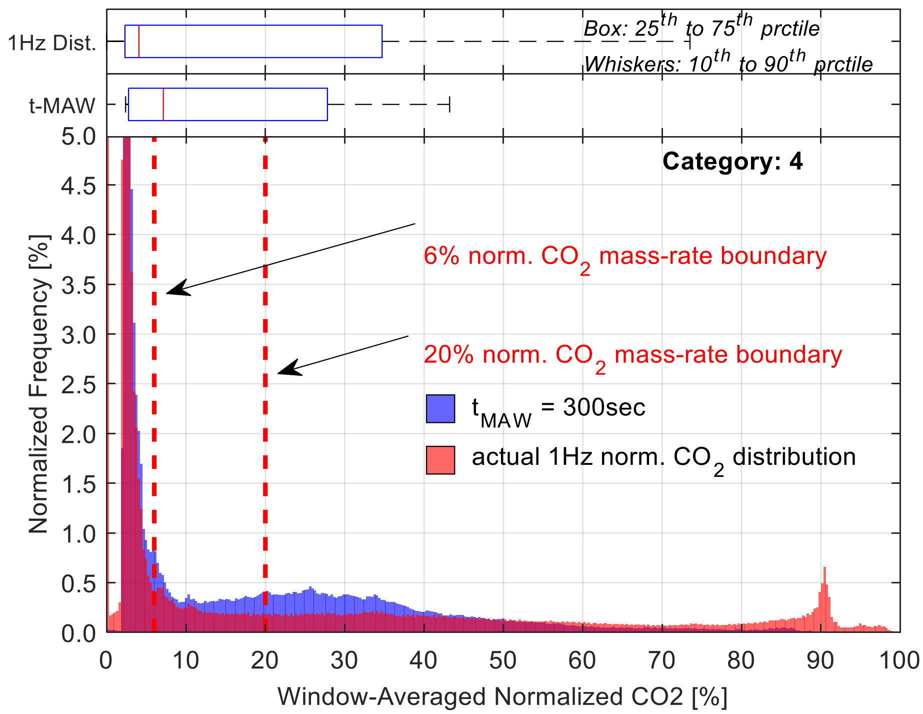


Figure 33: Window-averaged (w/ t_{MAW} = *300sec*, blue dist.) vs. actual 1Hz (red dist.) normalized CO_2 mass rate distributions for category 4 (i.e., tractor construction) vehicles.

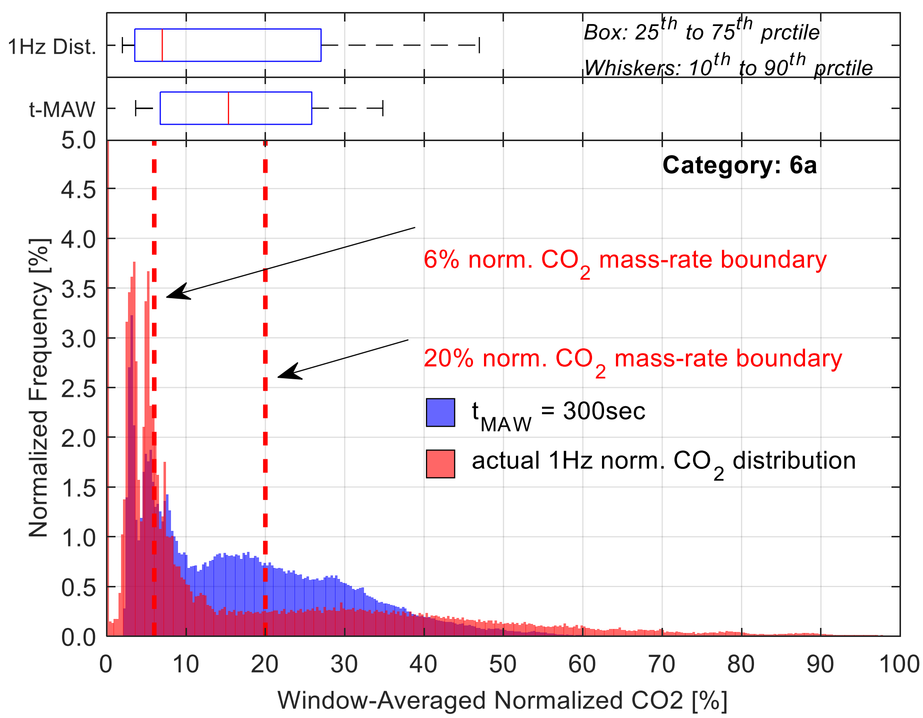


Figure 34: Window-averaged (w/ t_{MAW} = *300sec*, blue dist.) vs. actual 1Hz (red dist.) normalized CO_2 mass rate distributions for category 6a (i.e., food/beverage distribution / moving/ towing, T6 interstate small) vehicles.

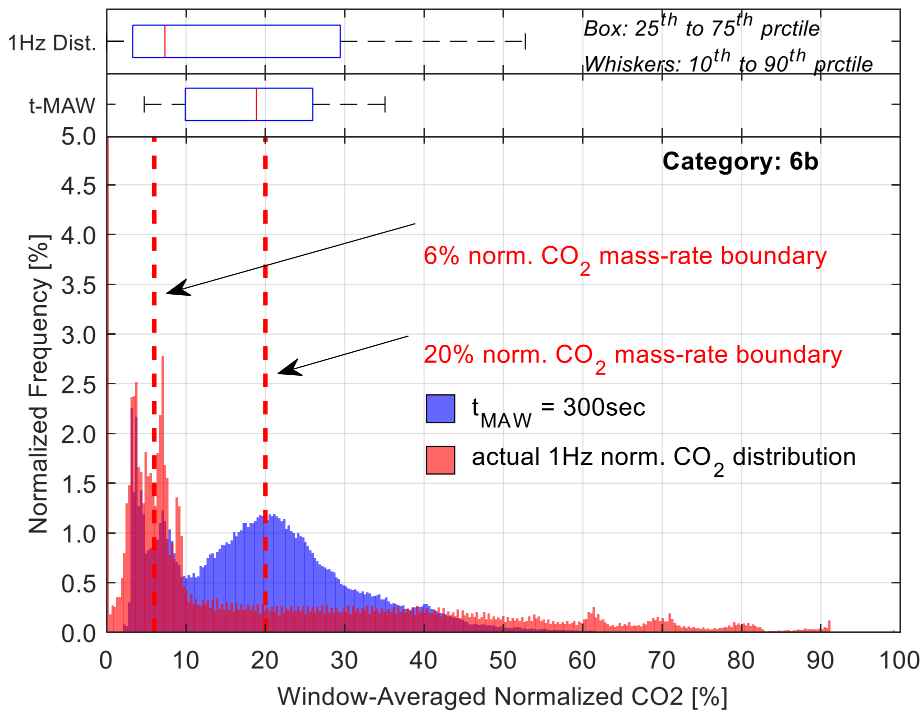


Figure 35: Window-averaged (w/ t_{MAW} = *300sec*, blue dist.) vs. actual 1Hz (red dist.) normalized CO_2 mass rate distributions for category 6b (i.e., food/beverage distribution / moving/ towing, T6 interstate heavy) vehicles.

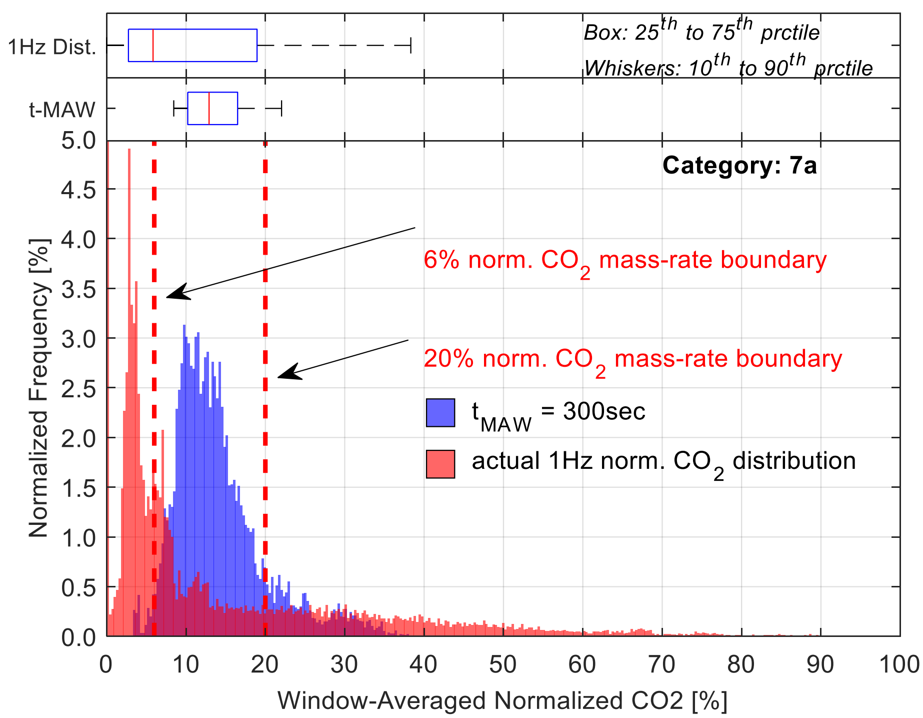


Figure 36: Window-averaged (w/ t_{MAW} = *300sec*, blue dist.) vs. actual 1Hz (red dist.) normalized CO_2 mass rate distributions for category 7a (i.e., goods distribution, T7 single) vehicles.

3 APPENDIX I

3.1 Multiple counting of datapoints for each vehicle category

Percentage count distributions of excluded, and single-, double- or triple-counted data points for each individual vehicle category are given in Figure 37 through Figure 41 along with tabulated percentiles and averages in Table 11 through Table 15.

3.1.1 Category 1a and 1b - long-haul (EMFAC: T7 NNOOS, NOOS, CAIRP) and short-haul (EMFAC: T7 tractor)

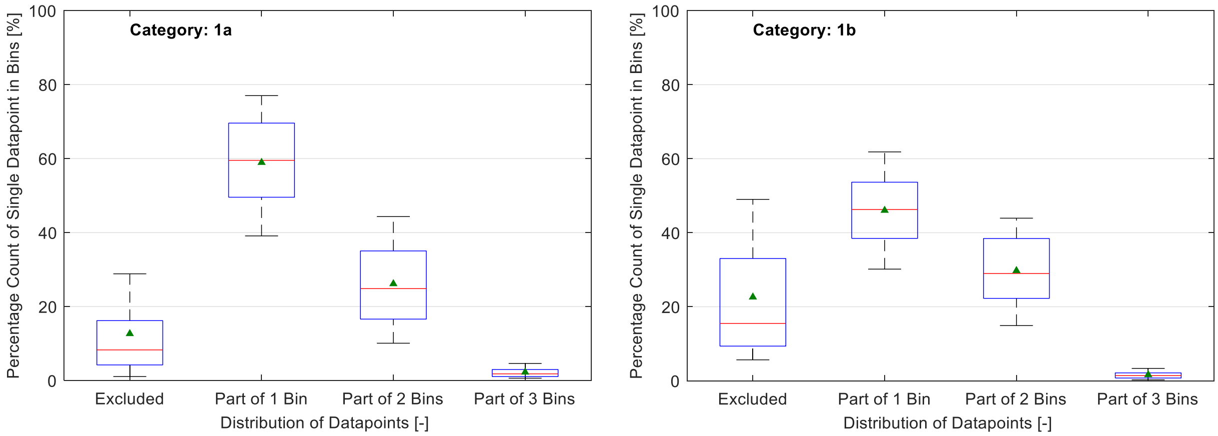
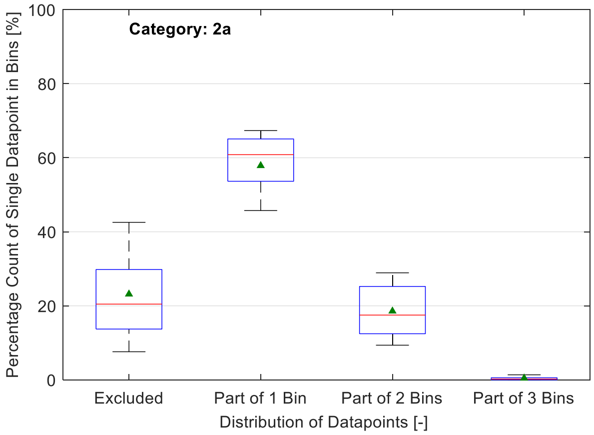


Figure 37: Percentage count of single data points appearing in either none (i.e., excluded), 1, 2 or 3 bins at the same time for vehicle category 1a (left) and 1b (right).

Table 11: Percentiles and averages of percentage count of single data points in none, one or multiple bins at the same time for vehicle category 1a (top) and 1b (bottom).

Category 1a	*Long haul (T7 NNOOS, NOOS, CAIRP)*					
Distribution of Data-points	**Percentiles**					**Mean**
[-]	**10th**	**25th**	**50th**	**75th**	**90th**	**[%]**
Excluded Data (Part of 0 Bins)	1.08	4.22	8.28	16.20	28.85	12.66
Data-points Part of 1 Bin	39.10	49.55	59.53	69.59	77.02	58.94
Data-points Part of 2 Bins	10.09	16.60	24.85	35.04	44.33	26.15
Data-points Part of 3 Bins	0.63	1.07	1.81	2.98	4.62	2.25
Category 1b	*Short haul (T7 tractor)*					
Distribution of Data-points	**Percentiles**					**Mean**
[-]	**10th**	**25th**	**50th**	**75th**	**90th**	**[%]**
Excluded Data (Part of 0 Bins)	5.66	9.36	15.49	33.03	49.00	22.61
Data-points Part of 1 Bin	30.16	38.46	46.27	53.65	61.84	46.03
Data-points Part of 2 Bins	14.91	22.27	28.97	38.43	43.93	29.75
Data-points Part of 3 Bins	0.26	0.74	1.39	2.13	3.34	1.62

3.1.2 Category 2a and 3a - port drayage (EMFAC: T7 POLA) and tractor construction heavy (EMFAC: T7 single construction)



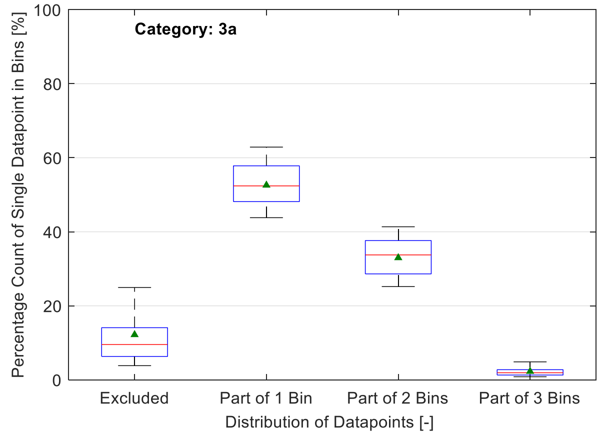


Figure 38: Percentage count of single data points appearing in either none (i.e., excluded), 1, 2 or 3 bins at the same time for vehicle category 2a (left) and 3a (right).

Table 12: Percentiles and averages of percentage count of single data points in none, one or multiple bins at the same time for vehicle category 2a (top) and 3a (bottom).

Category 2a	*Port Drayage (T7 POLA)*					
Distribution of Data-points	**Percentiles**					**Mean**
[-]	**10th**	**25th**	**50th**	**75th**	**90th**	**[%]**
Excluded Data (Part of 0 Bins)	7.59	13.72	20.45	29.81	42.55	23.15
Data-points Part of 1 Bin	45.73	53.64	60.83	65.08	67.34	57.85
Data-points Part of 2 Bins	9.35	12.45	17.49	25.23	28.91	18.56
Data-points Part of 3 Bins	0.00	0.00	0.20	0.56	1.36	0.44
Category 3a	*Tractor construction heavy (T7 single construction)*					
Distribution of Data-points	**Percentiles**					**Mean**
[-]	**10th**	**25th**	**50th**	**75th**	**90th**	**[%]**
Excluded Data (Part of 0 Bins)	3.83	6.31	9.54	14.08	24.94	12.19
Data-points Part of 1 Bin	43.81	48.14	52.40	57.82	62.87	52.57
Data-points Part of 2 Bins	25.19	28.62	33.75	37.62	41.35	32.97
Data-points Part of 3 Bins	0.83	1.29	1.91	2.74	4.86	2.27

3.1.3 Category 3b and 4 - cement mixer (EMFAC: T7 single construction) and tractor construction (EMFAC: T7 tractor construction)

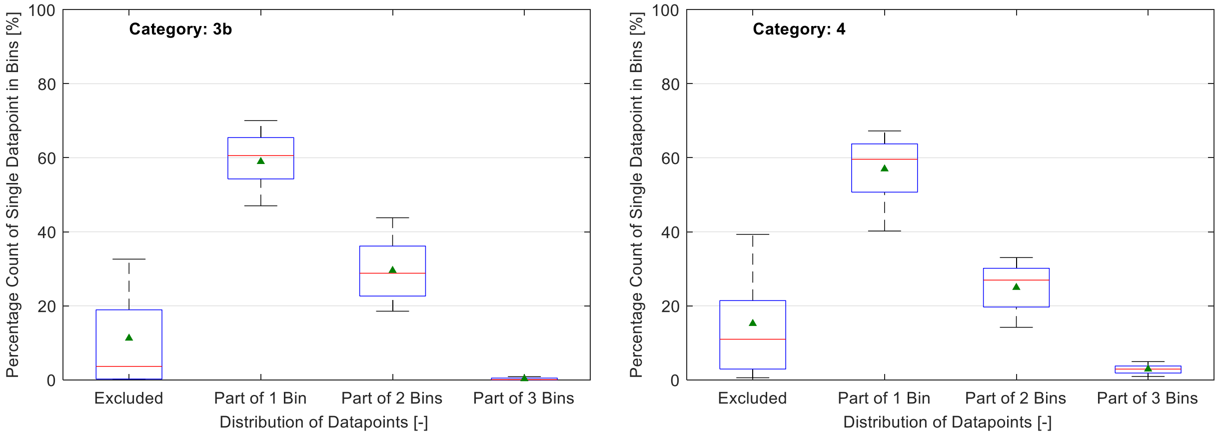


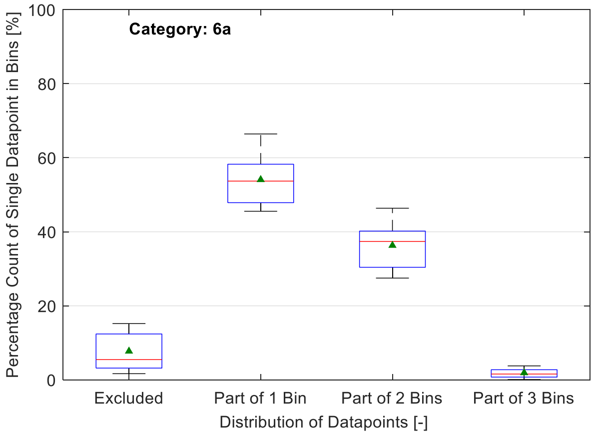
Figure 39: Percentage count of single data points appearing in either none (i.e., excluded), 1, 2 or 3 bins at the same time for vehicle category 3b (left) and 4 (right).

Table 13: Percentiles and averages of percentage count of single data points in none, one or multiple bins at the same time for vehicle category 3b (top) and 4 (bottom).

Category 3b	*Cement mixer (T7 single construction)*					
Distribution of Data-points	**Percentiles**					**Mean**
[-]	**10th**	**25th**	**50th**	**75th**	**90th**	**[%]**
Excluded Data (Part of 0 Bins)	0.15	0.22	3.63	18.91	32.60	11.23
Data-points Part of 1 Bin	47.02	54.28	60.58	65.44	70.03	58.95
Data-points Part of 2 Bins	18.54	22.63	28.80	36.14	43.79	29.54
Data-points Part of 3 Bins	0.00	0.00	0.05	0.47	0.86	0.28

Category 4	*Tractor construction (T7 tractor construction)*					
Distribution of Data-points	**Percentiles**					**Mean**
[-]	**10th**	**25th**	**50th**	**75th**	**90th**	**[%]**
Excluded Data (Part of 0 Bins)	0.56	2.91	10.96	21.41	39.30	15.20
Data-points Part of 1 Bin	40.21	50.71	59.59	63.74	67.24	56.96
Data-points Part of 2 Bins	14.18	19.67	26.94	30.13	33.03	24.94
Data-points Part of 3 Bins	0.91	1.83	2.90	3.74	4.92	2.90

3.1.4 Category 6a and 6b - food/beverage distribution / moving / towing (EMFAC: T6 instate small) and food/beverage distribution / moving / (EMFAC: T6 instate heavy)



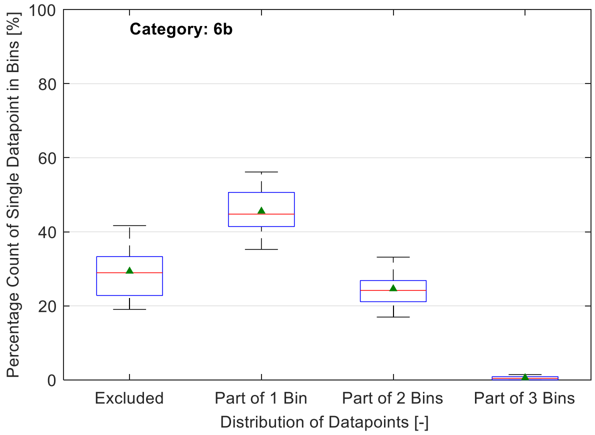


Figure 40: Percentage count of single data points appearing in either none (i.e., excluded), 1, 2 or 3 bins at the same time for vehicle category 6a (left) and 6b (right).

Table 14: Percentiles and averages of percentage count of single data points in none, one or multiple bins at the same time for vehicle category 6a (top) and 6b (bottom).

Category 6a	*Food/Beverage Distribution / Moving / Towing (T6 instate smal*					
Distribution of Data-points	**Percentiles**					**Mean**
[-]	**10th**	**25th**	**50th**	**75th**	**90th**	**[%]**
Excluded Data (Part of 0 Bins)	1.66	3.17	5.46	12.39	15.19	7.73
Data-points Part of 1 Bin	45.54	47.86	53.69	58.25	66.43	54.07
Data-points Part of 2 Bins	27.48	30.40	37.39	40.19	46.36	36.31
Data-points Part of 3 Bins	0.08	0.73	1.56	2.73	3.77	1.89
Category 6b	*Food/Beverage Distribution / Moving / Towing (T6 instate heav*					
Distribution of Data-points	**Percentiles**					**Mean**
[-]	**10th**	**25th**	**50th**	**75th**	**90th**	**[%]**
Excluded Data (Part of 0 Bins)	19.03	22.78	28.93	33.29	41.65	29.33
Data-points Part of 1 Bin	35.23	41.40	44.76	50.64	56.15	45.55
Data-points Part of 2 Bins	16.95	21.11	24.17	26.81	33.14	24.58
Data-points Part of 3 Bins	0.00	0.00	0.40	0.85	1.41	0.54

3.1.5 Category 7a - goods distribution (EMFAC: T7 Single)

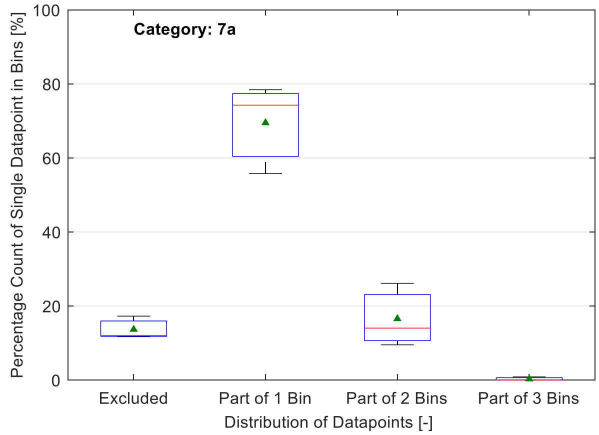


Figure 41: Percentage count of single data points appearing in either none (i.e., excluded), 1, 2 or 3 bins at the same time for vehicle category 7a.

Table 15: Percentiles and averages of percentage count of single data points in none, one or multiple bins at the same time for vehicle category 7a.

Category 7a	*Goods distribution (T7 Single)*					
Distribution of Data-points	**Percentiles**					**Mean**
[-]	**10th**	**25th**	**50th**	**75th**	**90th**	**[%]**
Excluded Data (Part of 0 Bins)	11.72	11.80	12.05	15.96	17.26	13.68
Data-points Part of 1 Bin	55.77	60.39	74.26	77.39	78.43	69.49
Data-points Part of 2 Bins	9.51	10.64	14.02	23.10	26.13	16.56
Data-points Part of 3 Bins	0.00	0.00	0.00	0.63	0.84	0.28

3.2 CO_2 emission mass rate calculation from ECU fuel rate parameter

The derivation of CO_2 mass rates ($\dot{m}CO_2$) for the 3B-MAW calculations from ECU engine fuel rate (V_{fuel}, PGN 65266, SPN, 183, Engine Fuel Rate, [L/hr]) is performed as demonstrated in the following. $\dot{m}CO_2$ in [g/s] calculated according to Eq. 1 from ECU derived fuel mass flow rate ($\dot{m}_{fuel}$) in [g/s] and reference CO_2 mass fraction per mass of fuel burnt (w_{CO2ref}) assuming complete combustion.

$$\dot{m}_{CO2} = \dot{m}_{fuel} \cdot w_{CO2ref}\ [g/s] \qquad \textbf{Eq. 1}$$

The fuel mas flow rate is calculated from Eq. 2, where V_{fuel} is the ECU broadcast fuel flow rate in [L/hr] and $\rho_{fuelref}$ the reference density of the fuel in [kg/L].

$$\dot{m}_{fuel} = \frac{\dot{V}_{fuel}}{3600} \cdot \left(\rho_{fuelref} \cdot 1000\right)\ [g/s] \qquad \textbf{Eq. 2}$$

The fuel density is calculated based on the API gravity of the test fuel. An API gravity of 34.5 is the midpoint of the 32 - 37 range of permissible certification diesel fuel, according to Table 1 of 40 CFR §1065.703. Conversion from API gravity to density, per ASTM D1250 (2008), is shown in Eq. 3. For an API gravity of 34.5, Eq. 3 yields a fuel density of $\boldsymbol{\rho_{fuelref}}$ **= 0.85157087 kg/L**.

$$\rho_{fuelref} = \frac{141.5}{\left(API_{gravity} + 131.5\right)} \cdot \frac{999.016}{1000}\ [kg/L] \qquad \textbf{Eq. 3}$$

Finally, the reference CO_2 mass fraction per mass of fuel is calculated from the fuel-specific hydrogen to carbon ratio (α_{fuel}) and the molecular weight of CO_2 (M_{CO2}), hydrogen (M_H) and carbon (M_C) as shown in Eq. 4. From 40 CFR §1036.530 Table 1, w_{Cref} is 0.874 for reference Diesel fuel, which would be equivalent to α_{fuel} = 1.717879715 (and β_{fuel}, γ_{fuel}, δ_{fuel} equal to zero), with M_{CO2} = 44.0095 g/mol, M_C = 12.0107 g/mol, and M_H = 1.0079 g/mol. The resulting value for $\boldsymbol{w_{CO2ref}}$ **is 3.2025**.

$$w_{CO2ref} = w_{Cref} \cdot \frac{M_{CO2}}{M_C} = \frac{M_C}{(\alpha \cdot M_H + M_C)} \cdot \frac{M_{CO2}}{M_C} = \frac{M_{CO2}}{(\alpha \cdot M_H + M_C)} \qquad \textbf{Eq. 4}$$