NRDC comments on Proposed Rules for Reprocessable Construction/Demolition Material Facilities

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[Warning: External email]

Please see attached comments from the Natural Resources Defense Council regarding CDPH's Proposed Rules for Reprocessable Construction/Demolition Material Facilities. As indicated in the comments, they are supported by the Chicago EJ Network, and NRDC in turn supports comments submitted by CEJN and its individual members.

I will also be submitting exhibits as attached pdf files to a series of emails. Please acknowledge receipt of the 19 exhibits. This email contains exhibits 1-9a and 9b (just over 5 MB by my count).

Thanks,
Meleah

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Re: Comments on Proposed Rules for Reprocessable Construction/Demolition Material Facilities

To Whom It May Concern,

We submit these comments on the Proposed Rules for Reprocessable Construction/Demolition Material Facilities ("Proposed Rules"), noticed by the Chicago Department of Public Health ("CDPH"), on behalf of the Natural Resources Defense Council ("NRDC") and our millions of members and activists, including thousands of members and activists in the City of Chicago, some of whom live, work and/or play along the I-55 corridor and on the Southeast Side within a short distance of facilities that would be regulated by the Proposed Rules. The Chicago Environmental Justice Network ("CEJN") and several of its individual members support these comments as well. NRDC also supports the comments of CEJN and its individual members submitted in this rulemaking. At the outset, we note the continuing and ever-more pressing need for regulations and enforcement that address the many sources of pollution from facilities impacting environmental justice communities in Chicago. More generally, the COVID-19 pandemic has resulted in gross disparities in health outcomes, including deaths, for those most vulnerable in our society. Emerging studies indicate that living in areas with polluted air is linked to greater mortality rates from COVID-19. In short, these rules, and making good on the City's commitment to protect its residents and address environmental injustices, are needed now more than ever.

We reiterate that while we welcome CDPH's regulations for reprocessable construction and demolition material facilities, environmental regulations are only one piece of the needed reforms. Regulating individual industries on the back end without addressing distributive siting issues and cumulative impacts, and more broadly upholding civil rights, is not enough. We look forward to further working with CDPH and other committed city staff and stakeholders at this higher level to ensure a clean, safe, productive and equitable Chicago for all residents.

I. Impacts of Reprocessing and Handling Construction and Demolition Materials

While a full review of the impacts of this type of industrial facilities on the environment and public health is beyond the scope of these comments, we highlight a few impacts of concern as follows:

- Reprocessing and handling of construction and demolition materials is widely recognized as a dusty business. Crystalline silica in particular can be found in a range of construction materials, including brick, concrete, and drywall. Exposure to mobilized crystalline silica from reprocessing and/or handling of construction and demolition materials can lead to silicosis, lung cancer, and other health impacts. Indeed, air quality and public health agencies have identified a specific fraction of PM for crystalline silica, PM4, and set distinct health-based thresholds for this pollutant, as discussed in more detail below.
- Facilities that take in C&D material for reprocessing will likely be handling an array of toxic substances on the front end – before, during and after the sorting process. Also, as discussed in these comments, reprocessing construction and demolition materials can result in byproducts and waste products that themselves contain toxic substances like lead and other heavy metals. Such substances can contaminate soil and water in and around the facilities handling them, as well as become airborne.
- Facilities that would be covered by the Proposed Rule are served by a wide array of trucks, including ones that remain onsite and ones that transport materials to/from facilities, which themselves produce both fugitive dust and diesel exhaust emissions.
- All of the facilities that CDPH anticipates will be covered by the Proposed Rules are located in environmental justice communities that are already overburdened by pollution. With few if any exceptions, all existing and proposed facilities furthermore are located in close proximity to homes, parks, and/or commercial areas frequented by members of the public, in many cases separated only by a single street and/or distances of less than 300 feet. We are aware of community complaints against such facilities as well, though an accounting of these complaints is beyond the scope of these comments.

For these reasons, addressing these facilities' many impacts is long overdue and sorely needed. CDPH should strengthen the Proposed Rules in the ways set forth below and in the comments of CEJN and its individual members.[1]

II. Strengthening Air Quality Impact Assessment and Air Monitoring Provisions

The Proposed Rules contain two primary provisions regarding air quality, Section 3.8.21 requiring that a permit application include an air quality impact assessment consisting of an emissions and dispersion modeling study along with dust monitoring and calibration plans, and

[1] We note that the given the time available for comment and the number and scope of closely related City regulations/guidance and state requirements, we were not able to do a complete comparison of the Proposed Rules to existing regulations like the bulk material rules adopted initially in response to petcoke threats, as amended, and Illinois State Implementation Plan requirements for visible emissions and fugitive dust. We did attempt to flag instances in which it appears the Proposed Rule would require or allow less of facilities than these existing parallel sources and comment that CDPH should take the more stringent approach. However, we may have missed some instances, and thus generally set forth that where analogous City and/or state sources require more stringent regulation of parallel operations and industries, CDPH should amend the Proposed Rules to strengthen them consistent with these existing sources.

Section 5.8 containing air quality standards and monitoring requirements. CDPH should make the following improvements to these sections.

a. Air Quality Impact Assessment – emissions and air dispersion modeling study

i. Assessing unacceptable air quality impacts

CDPH should make explicit that exceedances of individual National Ambient Air Quality Standards (NAAQS) and other federal metrics like minimum risk levels for individual metals (MRLs) are not the sole measures by which the agency will determine whether a facility poses unacceptable impacts to air quality for purposes of the Proposed Rules (and other environmental permitting). There are numerous justifications for such a position.

First, the NAAQS are not bright-line thresholds under which no health impacts will occur. U.S. EPA and other regulatory bodies have recognized that substantial impacts to health can and do occur from ambient levels of pollution below the NAAQS, including due to air exposures and to exposures via other pathways.[2] Indeed, the authors of the Clean Air Act recognized this fact, as set forth in the Prevention of Significant Deterioration provisions. The purpose of that program is "to protect public health and welfare from any actual or potential adverse effect which … may reasonably be anticipate [sic] to occur from air pollution or from exposures to pollutants in other media, which pollutants originate as emissions to the ambient air) [sic], *notwithstanding attainment and maintenance of all national ambient air quality standards*…".[3] As discussed elsewhere in these comments, PM is one primary example of a criteria pollutant for which science has not identified a no-effects threshold; lead is another clear example. Nor are individual MRLs all bright line thresholds under which no harm will occur, as set forth in our June 2018 comments to CDPH on the manganese dust rule revisions, which we incorporate here by reference. Generally speaking, more air pollution equals more harm, creating disparities even where an area or areas are "meeting" existing ambient air thresholds.

That significant health impacts can and do occur at levels below the NAAQS and MRLs is a critical concept in recognizing and addressing ongoing disparities in pollution burden across Chicago, which in turn CDPH is obligated to do (as set forth in our partners' comments and our past joint comments on the Large Recycling Facility Rules and in the General III permit proceeding). To put it another way, CDPH has the authority and the obligation to examine whether some communities bear greater pollution burdens than others and to address these disparities – even where, based on available monitoring and modeling, all such communities are and will continue to be in "attainment" with the NAAQS.

[2] *See infra* regarding PM2.5.
[3] 42 U.S.C. § 7470(1) (emphasis added).

In addition, individual thresholds like the NAAQS and MRLs do not account for the fact that communities facing proposed facilities are exposed to all or many of those pollutants together, along with other pathways of exposure. Nor do they account for the impacts of historic exposure to high pollution levels. Properly accounting for public health impacts requires assessing the combined and potentially synergistic effects of multiple pollutant exposures, looking at past as well as present and proposed pollution levels. Moreover and of critical importance for environmental justice communities, these individual thresholds do not adequately take into account the social determinants of health, i.e., fully recognize that some people are more vulnerable to pollution than others based on their socioeconomic characteristics.

Determining a proper "background" level by which to evaluate pollution from a proposed facility to the NAAQS and/or MRL also is not an exact science, due in large part to the dearth of monitoring data for many areas within Chicago – including even the industrial areas. It is well established that "fenceline" communities often experience localized pollution levels far above those captured by the small number of regulatory monitors typically used in such NAAQS and/or MRL evaluations.[4] Here in Chicago, for instance, fenceline regulatory-grade monitors placed at KCBX and S.H. Bell, facilities engaged in material handling with residential areas just across the street, have identified localized levels of PM10 and metals far above those recorded at the nearest Illinois network monitor at Washington High School during the same time period. In addition, self-reported, paper calculation emissions estimates from state inventories typically added to "background" monitoring data to account for nearby sources in modeling air quality impacts of a proposed facility are inaccurate at best and tend to underestimate actual air quality in an area as a whole. As we've raised in the past, inventories often completely omit substantial sources of air pollution, in particular alleged minor and/or de minimis sources; this was the case for the S.H. Bell manganese-handler, which was absent from the state's emissions inventory prior to community advocacy efforts. Calculated emissions for fugitive emissions are an especially imprecise exercise.

These shortcomings in available air quality and emissions data are an especially important point for evaluation of reprocessable construction and demolition material facilities, which as noted above are located very close to residential areas, parks and commercial areas in EJ communities: there is a single regulatory monitor along the I-55 corridor in Chicago, located at Perez Elementary School, which monitors only Total Suspended Particulate and metals. At the same time, members of the Chicago EJ Network have conducted or are conducting air quality monitoring studies that demonstrate elevated levels of air pollution in their communities. CDPH can and should seek out this and other available data when characterizing the air quality in communities.

[4] *See, e.g.*, Ex. 1, McLaughlin, T. et al., Special report: U.S. air monitors routinely miss air pollution – even refinery explosions, Dec. 1, 2020, Reuters, available at https://www.reuters.com/article/usa-pollution-airmonitors-specialreport/special-report-u-s-air-monitors-routinely-miss-pollution-even-refinery-explosions-idUSKBN28B4RT.

There are available ways for CDPH to take into account these issues in implementing its rules. For instance, CDPH could implement its own, local version of a prevention of significant deterioration and/or disparities program and significant impact levels. Public health methodologies also exist for assessing combined exposures to multiple pollutants, as well as to account for heightened vulnerabilities of some individuals and subpopulations. CDPH can and should require onsite (and near-site), baseline monitoring for use in modeling of proposed new or expanded facilities. For all of these reasons, CDPH should make clear that the individual NAAQS and individual MRLs or equivalent thresholds are not bright line tests by which the agency will evaluate air impacts from proposed facilities.

ii. PM2.5

The Proposed Rules require, at minimum, estimation and modeling of PM10.[5] While we appreciate that the Proposed Rules set the minimum requirements for the air quality impact assessment and that CDPH may require additional analysis in the individual case, we strongly recommend that CDPH require PM2.5 estimation and modeling as part of the minimum requirements for this class of facilities.

This request is based on several factors:

- Rock crushing activities can and do generate significant, localized amounts of fine particles, though the majority of particles from such mechanical activity will be in the larger fraction.
- While air monitoring data is limited, available data for Chicago supports that the environmental justice areas in which these facilities are or are proposed to be located likely have elevated levels of PM2.5 overall and in relation to other Chicago communities, particularly along the roadways to/from the industrial areas in which they are sited. For instance, the stationary monitor at Washington High School near the proposed site for the Chicago Port and Rail facility has the highest 24-hour design value for PM2.5 (2018-2020 data) among PM2.5 state monitoring sites in Chicagoland. In addition, hand-held monitoring efforts have identified relatively elevated levels of PM2.5 in areas near existing and proposed facilities that would be covered by the rules, including between Ashland and Loomis and Cermak and the Canal; at California and the Canal; along Loomis south of the Canal to 47th; and at 106th and Avenue O.[6]
- Reprocessable construction and demolition material facilities are typically served by a number of heavy-duty vehicles that move materials around the site, as well as to and from

[5] Proposed Rules at 3.8.21 and 3.8.21.1.1

[6] *See* Air Quality Chicago, Prevalence of High PM2.5 Levels, interactive website available at https://airqualitychicago.org/results/.

the site. As taken up elsewhere in these comments and our partners' comments, such vehicles can have significant impacts on local air quality, from both their mechanical emissions and combustion emissions (with the latter mainly resulting in increased PM2.5, though see elsewhere in these comments regarding other pollutants associated with diesel combustion).

- CDPH recently released its Air Quality Impact Evaluation Interim Guidance, to be used for the air quality impact evaluation required by the March Air-Zoning ordinance. This Interim Guidance calls for modeling of PM2.5 (and NO2, see below) along with PM10, demonstrating that such modeling of fine particles for facilities covered by the Proposed Rule is feasible now. Since none of the existing or two proposed reprocessable construction/demolition material facilities will be subject to the new site plan review's air quality impact evaluation requirements, omitting PM2.5 from the required minimum air quality demonstration under the Proposed Rules would result in an omission of a critical impact and perpetuate discrepancies between the two air evaluation processes.
- U.S. EPA has recognized that to date, science has not identified a "no-effects" threshold for particulate matter – i.e., more PM equates to more harm.[7] In addition, U.S. EPA has committed to revisiting the National Ambient Air Quality Standard for PM, "because available scientific evidence and technical information indicate that the current standards may not be adequate to protect public health and welfare."[8] Groups like the American Lung Association have in the past set forth bases for revising the PM2.5 NAAQS to an annual standard of 8 ug/m3 and a 24-hour standard of 25 ug/m3.[9] For comparison, according to U.S. EPA, the 24-hour design value for PM2.5 at the Washington High School monitor (2018-2020) is 25 ug/3.

In sum, facilities that would be regulated by the Proposed Rules may pose significant additional PM2.5 pollution in parts of Chicago that are already bearing a disproportionate PM2.5 burden. CDPH thus should require PM2.5 modeling as a baseline requirement of the Proposed Rules, to ensure such facilities do not pose unacceptable impacts to public health and well-being.

iii. PM4

CDPH should in the Proposed Rules define and regulate PM4 crystalline silica, a specific fraction and type of particles for which air quality and public health agencies have set health-based thresholds. For instance, California has an inhalation reference exposure level of 3 ug/m3

[7] Including under the Trump Administration, *see, e.g.*, 85 Fed. Reg. 82684 at 82696, 82697, 82698, and 82704 (Dec. 20, 2020).
[8] Ex. 2, U.S. EPA, EPA to Reexamine Health Standards for Harmful Soot that Previous Administration Left Unchanged, June 10, 2021, available at https://www.epa.gov/newsreleases/epa-reexamine-health-standards-harmful-soot-previous-administration-left-unchanged.
[9] *See* Ex. 3, American Lung Association, Comments on Docket ID No. EPA-HQ-OAR-2015-0072 (Nov. 12, 2019), at 5, available at https://www.lung.org/getmedia/84f50d4f-9d3e-4599-ae01-9896b98bc062/lung-association-comments-to-4.pdf.pdf.

to be applied to respirable particles of crystalline silica, which occur in the PM4 fraction.[10] While efforts to define and regulate PM4 crystalline silica have taken place primarily in the occupational setting, they can and should be used for assessing community/environmental exposures as well, especially given the proximity of facilities in this category to homes, parks, and other places where members of the general public congregate.[11] For application to community settings, we recommend application of additional safety factors to the California REL or other occupational thresholds for crystalline silica to account for exposure of the general population, including people who are unhealthy, have pre-existing lung diseases, asthmas, etc., and pregnant women and children, as well as continuous exposures.[12]

iv. Other pollutants

In addition to requiring an assessment of impacts on PM2.5 levels and regulating PM4 crystalline silica, CDPH should require evaluation of air pollutants more closely associated with localized creation of hot spots by diesel trucks, such as ultrafine particles (PM0.1) and nitrogen oxides (NOx, of which nitrogen dioxide or NO2 is the main indicator).

Ultrafine particles are a product of combustion associated with diesel vehicles. They have negative health effects distinct from (though overlapping with) PM2.5, given how they penetrate the bronchioles and lungs.[13] Ultrafine particles also can travel up the olfactory pathway to the brain, causing cerebral and autonomic dysfunction; *in utero* exposure to ultrafine particles is associated with low birthweight.[14] Moreover, while to date the rule of thumb is that localized concentrations of PM associated with mobile source corridors drop off to background levels by roughly 500 meters, research demonstrates that ultrafine particle plumes from roadways can travel in the downwind direction for roughly 1.5 to 2 km, in particular overnight and in the early morning.[15]

Nitrogen oxides impact health not only through their contribution to ozone/PM/smog formation, but also via direct exposure to breathing air with high levels of NOx. U.S. EPA provides this summary of the health impacts of NO2:

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[10] *See* Ex. 4, OEHHA, Chronic Toxicity Summary, Silica (Crystalline, Respirable), February 2005, available at https://oehha.ca.gov/media/downloads/air/document/silicacrelfinal.pdf.

[11] *See, e.g.*, Ex. 5, ATSDR, Air Exposures to Particulate Matter and Silica Dust, Valley, WA, July 2019 (describing an investigation of PM and crystalline silica at an elementary school located across the street from a large producer of silica sand products).

[12] We recommend use of at least an additional 3X for UFintraspecies differences such as health status and 10X for children.

[13] *See* Ex. 6, Schraufnagel, D. The Health Effects of Ultrafine Particles, Experimental & Molecular Medicine (2020) 52:311–317, available at https://www.nature.com/articles/s12276-020-0403-3.pdf.

[14] *Id.*

[15] *See, e.g.*, Ex. 7, Choi, W. et al., Prevalence of wide area impacts downwind of freeways under pre-sunrise stable atmospheric conditions, Atmospheric Environment (2012) 62:318-327, abstract available at https://www.sciencedirect.com/science/article/abs/pii/S1352231012007753?via%3Dihub.

> … exposures [to NO2] over short periods can aggravate respiratory diseases, particularly asthma, leading to respiratory symptoms (such as coughing, wheezing or difficulty breathing), hospital admissions and visits to emergency rooms. Longer exposures to elevated concentrations of NO_2 may contribute to the development of asthma and potentially increase susceptibility to respiratory infections. People with asthma, as well as children and the elderly are generally at greater risk for the health effects of NO_2.[16]

Not only is NO2 associated with diesel vehicles, but diesel vehicles driven at slow, "urban" speeds are associated with exceptionally high levels of NO2 emissions. An in-use study of heavy-duty diesel vehicles in the U.S. by the International Council on Clean Transportation found that "[a] disproportionate amount of NOx emissions from heavy-duty vehicles is emitted during the low-speed operation characteristic of urban driving" and that "[v]ehicle operation at speeds of less than 25 mph results in NOx emissions of more than five times the certification limit for the average heavy-duty vehicle in the study."[17] The report notes that "a single line-haul truck emits the NOx equivalent of 100 cars for each mile driven in urban driving."[18] A recent study using mobile air sensors identified significantly elevated markers of primary combustion, including NOx, near truck-intensive industrial facilities, consistent with levels found within 200 meters of highways.[19]

For these reasons, and because reprocessable construction and demolition material facilities are vehicle-intensive uses, CDPH should require evaluation of ultrafine particles and NO2 under the Proposed Rules. Moreover, as discussed elsewhere in these comments, CDPH recently released interim guidance on air quality modeling. This guidance requires 1-hour

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[16] Ex. 8, U.S. EPA, Basic Information about NO2, available at https://www.epa.gov/no2-pollution/basic-information-about-no2.

[17] *See* Ex. 9, ICCT, Current State of NOx Emissions from In-Use Heavy-Duty Diesel Vehicles in the United States, available at https://theicct.org/publications/nox-emissions-us-hdv-diesel-vehicles (Badshah. H et al., white paper available at https://theicct.org/sites/default/files/publications/NOx_Emissions_In_Use_HDV_US_20191125.pdf). We note that while data on localized NO2 levels is relatively thin for Chicago, U.S. EPA has confirmed to us that preliminary analyses of ozone data that take into account summer 2021 show the South Water Treatment Plant as having the highest ozone value among Chicagoland ozone monitors. This is a significant change from historic high values, which have been at the far north monitors further back and at the Northbrook monitor (along with Alsip and Evanston) in the more recent past. Such a shift may be explained at least in part by high volumes of truck traffic on interstates and local roadways on the South/Southeast Sides relative to reduced vehicle traffic on Northside highways during the pandemic. While ozone concentrations are not a direct measure of localized NO2, this data suggests that NO2 levels are relatively high in these areas as well.

[18] *Id*. This finding is consistent with research looking at the reduced effectiveness of selective catalytic reduction on diesel vehicles at lower temperatures associated with off cycle idling. *See* Ex. 10, Mark Janssen, Mobile Source Modeling: Strengths, Weaknesses, and Opportunities for Improvement, presented at LADCO Regional Air Quality Meeting, April 2019, at slide 12, available at https://www.ladco.org/wp-content/uploads/Meetings/2019-04_Regional_AQ/6_Janssen-Onroad-Spring-Meeting2019_V5_04172019.pdf.

[19] *See* Ex. 11, Miller, D. et al., Characterizing Elevated Urban Air Pollutant Spatial Patterns with Mobile Monitoring in Houston, Texas, *Environmental Science & Technology* (2020) *54* (4), 2133-2142, available https://pubs.acs.org/doi/pdf/10.1021/acs.est.9b05523.

NO2 modeling, including onroad mobile source diesel emissions (and implicitly nonroad mobile diesel emissions, such as from forklifts, dozers and other mobile sources that remain at facility yards), demonstrating that NO2 modeling is feasible and should be required in the Proposed Rules as well. To the extent that CDPH finds it is not currently feasible to conduct similar modeling for ultrafine particles, CDPH should develop alternative methodologies for assessing a project's impact on ambient levels of ultrafine particles.

v. On-road mobile source diesel

Relatedly, CDPH should include on-road mobile diesel engine emissions in the minimum required air quality impact assessment. The Proposed Rules, like the Large Recycling Facility Rules, state that while "diesel engines" are generally to be included in the minimum air quality impact assessment, "[d]iesel emissions from on-road mobile sources are not required to be included in the modeling study."[20] It appears that this language in the Large Recycling Facility Rules came about due CDPH's agreement to abide by an industry request to exclude diesel emissions, citing (without further analysis) this company's comment that it "believes" the City's idling reduction plan is sufficient to protect air quality from truck emissions.[21]

Moving forward, CDPH should not adhere to this past decision (and should update its Large Recycling Facility Rules as well). As set forth in an October 2020 letter to CDPH regarding this aspect of the Large Recycling Facility Rules, which we incorporate here by reference with respect to the Proposed Rules, on-road mobile diesel engines can cause significant "hot spots" of pollution around the facilities they serve. Since that letter, additional studies have come out pointing to systemic racial and ethnic disparities in hyperlocal air pollution attributable to truck traffic, with communities of color generally bearing disproportionately greater burdens.[22]

Also since that letter, CDPH has issued its Interim Guidance on air quality studies under the March 2021 Air Quality Zoning Ordinance, which expressly requires inclusion of on-road mobile sources. Failing to require on-road mobile sources here would create inconsistent

[20] Proposed Rules at 3.8.21.1.1. We note that this language can and should be read to *require* inclusion of diesel emissions of *nonroad* mobile sources in all air quality impact assessments. Furthermore, given that Section 3.8.21 itself states that the requirements are "not necessarily limited to" the more detailed directions that follow, we also read this language as recognizing that CDPH can require inclusion of diesel emissions from on-road mobile sources on a case-by-case basis. It should do so where, as is the case with many if not all of these facilities, the facilities are located in close proximity to homes, parks and other places where people congregate; are in areas having relatively poor air quality; and are served by many diesel trucks.

[21] *See* CDPH, Official Response to Public Comments on Proposed Rules for Large Recycling Facilities, June 5, 2020, at 35 and 37 (discussing comments by SIMS and stating that "in response to SIMS's in response to SIMS's comments, CDPH clarified that engine emissions from onroad mobile sources do not have to be addressed in the modelling in response to SIMS's comments, CDPH clarified that engine emissions from onroad mobile sources do not have to be addressed in the modelling [sic].").

[22] *See, e.g.*, Ex.12, Chambliss, S. et al., Local- and regional-scale racial and ethnic disparities in air pollution determined by long-term mobile monitoring, Proceedings of the National Academy of Sciences (2021), available at https://www.pnas.org/content/pnas/118/37/e2109249118.full.pdf.

approaches and confusing results, with analysis under the Proposed Rules falling short of accounting for facilities' full impacts on the environment and public health. Conversely, including on-road mobile sources in the minimum modeling required by the Proposed Rules would ensure that CDPH can assess a more complete picture of a facility's impact and take measures to address the impact. Such measures may include, but are not limited to, requiring a facility to have vehicle charging infrastructure for both nonroad and on-road vehicles and/or to further reduce its PM emissions from stationary emissions sources to reduce the facility's overall impact, such as through additional controls (like enclosure), limits on capacity/throughput, and/or other design features. As noted elsewhere in these comments, Existing Facilities that would be subject to the Proposed Rules will not undergo site plan review unless and until they modify or expand, and thus (all else constant) will not have to evaluate their on-road mobile source diesel emissions unless such evaluation is required under the Proposed Rules.

vi. Emissions estimates and modeling

While the Proposed Rules themselves do not include detailed guidelines or procedures for estimating emissions and conducting air quality modeling, CDPH recently released an Air Quality Impact Evaluation Interim Guidance that the agency intends to use in the "air quality impact evaluation" required under the March Air-Zoning ordinance.[23] Assuming CDPH will employ the same approaches in the Proposed Rules as are set forth in this Interim Guidance, we note that we plan to provide CDPH additional comments on the interim guidance, and thus request that CDPH take such comments into account for purposes of these Proposed Rules.[24]

b. Air Quality Impact Assessment – dust monitoring plan

The Proposed Rules lay out an independent requirement that applicants submit a dust monitoring plan that describes various aspects of the required monitors and weather station, along with a schedule and plan for quarterly testing to ensure compliance with prohibitions on visible dust beyond the fenceline and opacity requirements.[25] They also contain a separate provision requiring applicants to submit state operating programs for fugitive dust.[26] Given differences between state and local requirements and to ensure that applicants comply with both of these separate requirements in the Proposed Rule, CDPH should make explicit that applicants must submit a separate dust monitoring plan that explicitly addresses CDPH's Rules and not simply rely on a state operating program for purposes of satisfying both provisions. Such a requirement

[23] *See* CDPH, Air Quality Impact Evaluation Interim Guidance, September 2021, available at https://www.chicago.gov/content/dam/city/depts/cdph/InspectionsandPermitting/Air%20Quality%20Impact%20Evaluation%20Interim%20Guidance.pdf; *see also* CDPH, Air Quality Zoning Ordinance, available at https://www.chicago.gov/city/en/depts/cdph/supp_info/healthy-communities/chicago-air-quality-zoning-ordinance.html.

[24] On October 25, we submitted a request to CDPH for clarification on the process for submitting comments on the Interim Guidance.

[25] Proposed Rules at 3.8.21.1.2.

[26] Proposed Rules at 5.8.6.

will help avoiding a situation where, for example, an applicant submits a state operating plan that lacks the quarterly testing plan mandated by CDPH's regulations. Alternatively, an applicant can rely on a single plan submitted for both purposes, if that plan in fact clearly addresses all applicable state and local requirements.

c. Air Monitoring, Section 5.8

i. State Operating Program for Fugitive Dust.

See above comment regarding differences between state and local requirements, and the need to ensure that applicants clearly demonstrate in their dust plan submitted pursuant to the Proposed Rules how they will comply with CDPH's Rules.

ii. Air monitoring requirements

We reiterate and incorporate by reference our comments on the air monitoring provisions of the Large Recycling Facility Rules. In sum, CDPH should ensure that the types of air monitors required under the Proposed Rules provide sufficiently high quality and reliable data to be used for assessing compliance with the Proposed Rules and other related local and/or state requirements. As a general matter, we support CDPH requiring federal regulatory reference monitors and strongly encourage CDPH to continue to require such monitors, and to allow use of alternative monitoring methods only where such methods are for all intents and purposes functional equivalents of such reference monitors or where used to supplement data collected from reference monitors.

In addition, CDPH should establish air monitoring requirements for the other pollutants we raise in these comments, specifically PM2.5, PM4 crystalline silica, ultrafine particles, and NO2.

iii. Additional RALs for pollutants in addition to PM10

CDPH should establish in this rulemaking additional RALs for the pollutants other than PM10 that we raise in these comments, specifically PM2.5, PM4 crystalline silica, ultrafine particles, and NO2.

III. Strengthening Various Definitions, Control Requirements and Performance Standards

a. *Enclosure of processing and handling operations*

CDPH should strengthen the Proposed Rules by requiring enclosure of processing areas and storage piles to the greatest degree feasible. Enclosure is an available best practice that will better ensure continuous prevention of harmful fugitive dust over a wide range of operating conditions than other work-practice measures that depend on constant adherence by facility staff and that

are impacted by daily conditions such as temperature and other weather conditions. Because enclosure is relatively self-enforcing, it can also reduce or eliminate the need for various forms of monitoring that a facility would otherwise need to staff and undertake on an ongoing basis, like opacity monitoring, and/or disruptions in operations like high wind cessations as raised below. Also, enclosure can reduce the need for additional stormwater and/or noise measures, in addition to these air benefits. CDPH's rules for petcoke and coal handling recognize that enclosure is an available control for dusty operations such as those posed by reprocessable construction and demolition material facilities.[27] Despite this availability and these benefits, the only reference to enclosure of some form in the Proposed Rules is in Section 5.4.3., which requires "incidental debris" to be placed inside a three-sided enclosure.[28]

b. *Cessation of operations during high wind conditions*

Alternatively, if CDPH does not mandate enclosure of processing and handling operations, it should at minimum require cessation of outdoor processing and handling operations during periods of high winds. Such an approach is already required by Section E of the general bulk material rules as a prerequisite for facilities handling bulk materials other than coal and coke maintaining outdoor piles. These rules would otherwise apply to reprocessable construction and demolition material activities absent the exemption for construction and demolition materials in the definition of "bulk solid material" (see below). The general bulk material rules contain a baseline/default requirement that outdoor bulk material storage is allowed if "[d]isturbance of outdoor Bulk Solid Material piles, including but not limited to outdoor loading, unloading, and any other Processing, [is] suspended during High Wind Conditions, as detected by the wind monitor."[29] CDPH should adopt a similar requirement here.

c. *Zero Emissions vehicles and other diesel mitigation measures*

As noted elsewhere in these comments and those of our partners, diesel emissions are a serious concern for communities faced with truck-intensive uses like reprocessable construction and

[27] *See id*. (discussing use of wetting where "enclosing the material is impractical").
[28] We were unable to find a definition of "incidental debris" in the Code or the Proposed Rule. CDPH should define this term for purposes of the Proposed Rules, to provide clarity on the nature of this material as opposed to "reprocessable" or reprocessed material, as well as "waste" material and any other material that is or may be handled by facilities subject to the Propose Rules. We note that to the extent that "incidental debris" here or any other material that is or may be handled by facilities subject to the Proposed Rules is analogous or similar to a recognized byproduct of C&D reprocessing known as "recovered screen material," and/or if any such material or debris otherwise has the potential to contain toxic or hazardous substances such as polycyclic aromatic hydrocarbons, lead and other heavy metals, CDPH should adopt provisions addressing this material that take into account its composition and the material's potential impact on soil and water, along with air. *See* Ex. 13, Stormwater, New Life for C&D, Aug. 23, 2013 ("Stormwater C&D"), available at https://www.stormh2o.com/bmps/article/13008348/new-life-for-cd.
[29] *See* Rules for Control of Emissions from Handling and Storing Bulk Materials (January 2019), Part E, Outdoor Storage of Bulk Solid Materials Other than Coke or Coal, Section 7.0(4), High Wind Events and Section 2.0(12), definition of "high wind conditions."

demolition material facilities. Diesel emissions are caused not only by trucks coming and going from the site, but also from diesel engines that remain onsite, such as nonroad vehicles (construction equipment) and mobile energy sources/generators. Indeed, nonroad engines – in particular older ones that operate on a non-continuous basis, and so emit air pollutants at a relatively high amount – can be a substantial source of air pollution. According to air quality experts with whom we have consulted, such old and dirty engines may account for a large NO2 signal on Chicago's Southside that is not explained or accounted for by current mobile source emission inventories. Given these impacts and available control measures, CDPH can and should in this rulemaking acknowledge and address these vehicles and other diesel engines as part of its duty to protect public health and to address health disparities and civil rights.

The Proposed Rules contain several provisions related to vehicles:

- Section 3.8.13 requires a traffic assessment, which in turn requires various plans of operation, including an idling reduction plan that "[d]emonstrates compliance with Section 9-80-095 of the Code *and that minimizes unnecessary idling of vehicles and equipment in order to avoid contributions to poor air quality and noise*" (emphasis added);
- Section 3.9.9 requires that the source's Operating Plan include a description of vehicles "to be used at the Facility," including their types and EPA Tier rating of each;
- Section 5.7 generally pertains to the availability of sufficient vehicles to accept and process permitted volumes or weights of material, and further states that *"[s]uch vehicles and equipment shall be operated in a manner that minimizes emissions*, including but not limited to [express dust control measures]" (emphasis added); and
- Section 5.8.14 prohibits back-ups and idling of vehicles serving regulated facilities on public streets and rights-of-way.

In sum, the Proposed Rules recognize the potential impacts to public health and the environment from vehicles, and generally include duties to minimize idling and emissions. CDPH should build on this base by requiring use of Zero Emission technologies and other similar diesel mitigation measures wherever feasible.

Various sources are available for assessing the current availability of Zero Emission technologies for a range of diesel sources associated with reprocessable construction and demolition material facilities. In particular, agencies in California have informational materials that can and should inform CDPH's requirements for diesel sources here.[30] Even where Zero Emission vehicles themselves are not yet commercially available for purchase for a certain segment, CDPH can and should require all sites relying on diesel vehicles to be "electrification-ready" on the front end,

[30] *See, e.g.*, Ex. 14, CARB, Zero-Emission Off-Road Strategies, November 2020, available at https://ww2.arb.ca.gov/sites/default/files/2020-11/ZEV_EO_Off-Road_Fact_Sheet_111820.pdf.

including for both future electric vehicles and for purposes of providing zero-emission onsite power sources where in the past facilities have relied on diesel generators and idling truck engines.

Electrification of sites not only reduces emissions, it can also reduce noise impacts.[31] In addition, electrification of construction/nonroad vehicles can have other benefits for operators, including improved workplace air quality for workers themselves, along with reduced maintenance costs (especially brake expenses) over the life of the vehicle.[32]

For these reasons, CDPH can and should require Zero Emission technologies in place of diesel sources in the Proposed Rules. At the very least, CDPH should expressly reference Zero Emission technologies as alternatives to diesel sources and incorporate Zero Emission technologies in its case-by-case consideration of facilities under the Proposed Rules, as part of the overall charge to minimize idling by and emissions from diesel sources. Lastly, as discussed above, CDPH can and should make express that required diesel mitigation measures in general and in the individual case may entail changing the design and/or capacity of facilities to reduce their reliance on diesel engines, requiring the cleanest available diesel engines instead of allowing older and dirtier engines, and/or requiring other "offsetting" reductions from other, comparable sources of air pollutants associated with the proposed use. CDPH should also commit in these rules to reassessing the availability/feasibility of Zero Emission technologies to replace diesel in permit renewals.

We also note here that while electrification of vehicles offers near-term localized air quality improvements from reduced diesel emissions, the electric grid continues to rely heavily on fossil fuel generation, with disproportionate impacts on environmental justice communities. We strongly urge the City to pursue aggressive policies and targets for cutting our dependence on fossil fuels and reducing or eliminating their disproportionate impacts on environmental justice communities.

d. *Definition of "facility" to prevent improper segmenting*

CDPH should revise the definition of "facility" to protect against improper segmenting of nearby and closely related operations. In the past, we have raised concerns about adjacent and closely related facilities claiming to be separate facilities and thereby trying to avoid CDPH's regulatory

[31] *See, e.g.*, Ex.15, Daily Commercial News, Inside Innovation: Electrification makes worksites both "green" and not heard, Construction Connect, available at https://canada.constructconnect.com/dcn/news/technology/2021/03/inside-innovation-electrification-makes-worksites-both-green-and-not-heard.

[32] We also note that vehicle operators themselves report appreciating the cleaner operation of electric vehicles. For instance, at the recent Illinois Clean Truck Fleet Forum on September 30, a participant conveyed that one worker testing an electric nonroad vehicle noted that he appreciated not spilling diesel fuel on his expensive work boots and that he no longer had to shower every day after work, because his wife could not smell work on him anymore.

thresholds triggering the agency's rules, specifically the Large Recycling Facility Rules and their application to the proposed General III operation. We incorporate those comments by reference here.

One specific concern with the definition of "facility" in the Proposed Rules is that it discusses the "Site," which in turn is separately and circularly defined as "all areas of Property that are available for use or are used in the operations of the Facility." CDPH should revise these definitions to ensure that plot/site boundaries are not the defining element of a "facility," and conversely to ensure that all adjacent or nearby operations that meet various ownership/control and interrelatedness criteria are assessed and permitted together as a single "facility" under these rules.

e. *Pavement*

CDPH should strengthen Section 3.8.6 regarding Pavement to include that any such pavement must be sufficiently strong and robust to withstand the degree of heavy-duty vehicle operations to be undertaken at the facility, as well as to prevent contamination of soil, water, and air. The current language states that pavement must be "designed, constructed, and maintained to accommodate the vehicle flow rates and type of traffic loading expected at the Facility." This passage should be strengthened to explicitly reference withstanding over time the amount of force involved with continuous operation of heavy-duty vehicles, given the history of pavement issues at similar facilities.[33]

We also note particular concern that it appears that at least several of the existing facilities are largely if not entirely on unpaved sites, posing a high risk of soil contamination along with water and air impacts given the potential constituents in reprocessable construction and demolition materials. While it is difficult to assess pavement at facilities via available Google aerial images because of the light gray color of the material handled, it appears that the Vulcan facility on Racine may be unpaved, along with Reliable at Pulaski, Lindahl at California, Reliable at Grand, the proposed Stockyard Materials site at Ashland, and the proposed Chicago Port and Rail site at 106th. CDPH should require immediate measures to address potential soil, water and air contamination due to the unpaved nature of these sites.

f. *Vibration*

CDPH should include vibration requirements in the Proposed Rules. Processing of construction and demolition waste can entail use of heavy oscillating equipment, as well as a significant

[33] We incorporate by reference our past comments to CDPH regarding paving problems at recycling facilities here, including those on the chronic paving issues at the RMG recycling operations on S. Burley, as evidenced by CDPH's inspection reports.

amount of rotating equipment.[34] This equipment creates steady-state and impact vibration, which can be transmitted to adjacent properties, creating impacts to neighbors, including potential nuisances. Buffers and other measures can and should be required to ensure that vibration from these facilities does not negatively impact the surrounding community. To the extent that Chicago already has general-purpose vibration requirements, CDPH should evaluate whether those are sufficient for this industry and incorporate them by reference/amend the Proposed Rules accordingly.

IV. Strengthening Public Participation

CDPH should revise the Proposed Rules to include public participation requirements for its permitting of reprocessable construction and demolition material facilities. Such requirements are important to ensure that the environmental justice communities in which these facilities are currently located or proposed have an opportunity to be heard. In addition, it is important to provide an opportunity for public participation in the local permit proceeding because, as set forth elsewhere in these comments, it does not appear that any of them – including the two proposed facilities along I-55 and on the Southeast Side – will be subject to the public participation measures in the March Air Quality Zoning Ordinance unless and until they seek to modify or expand their facilities or entirely new facilities are proposed moving forward.

While the Code includes a 90-day timeframe for the Commissioner to take action upon an application, this is sufficient time for incorporating a public comment period.[35] Moreover, CDPH can provide a public comment period to aid in determining the sufficiency of the application as it has for the Large Recycling Facility Rules, with a subsequent determination that the application is complete triggering the Code's 90-day decision timeframe. And to the extent that 90 days is insufficient time to consider the materials and make a determination, especially with regards to civil rights considerations, City Council should take action to amend this provision and/or CDPH should determine other available procedural measures to ensure community participation and protection of public health and civil rights.

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[34] *See supra*, Stormwater C&D.

[35] *See* Chicago Municipal Code 11-4-1930(B). While the Proposed Rules reference both the aldermanic letter expressly required under 11-4-1930 and the public notification and community meeting requirements under the March 2021 Air Quality Zoning Ordinance, neither is a sufficient substitute for public participation in CDPH's permitting process. First, aldermanic approval is not equivalent to the viewpoints of community residents. Second, as discussed in these comments, none of the existing or proposed facilities will go through the site plan review process unless and until they seek to modify or expand their operations, though they will be required to comply with the Proposed Rules. Third, even if/when facilities must go through the enhanced site plan review process, that process is not a substantive equivalent of CDPH's Proposed Rules, and thus the scope of comments considered is likely to be more limited.

V. Removing the Loophole for Facilities Processing and/or Handling Aggregates, Concrete and Other "New" or "Frontend" Construction Materials

CDPH must close the gap in its environmental regulations for facilities that handle new construction materials or materials used in processes that create new construction materials, a gap that is perpetuated by Proposed Rules that would apply only to facilities handling and processing reprocessable construction and demolition material. Given that processing and handling of "frontend" or new construction materials also creates significant dust and other environmental impacts very similar to those from reprocessable construction and demolition materials facilities, CDPH should take immediate action to close this loophole. It can and should do so in this action by revising the Proposed Rules to apply to all qualifying facilities that handle construction and demolition materials, including those that handle construction material "used in" construction (not simply reprocessable construction and demolition material).[36]

At the very least, CDPH must ensure immediate and stringent regulation of slag grinding operations, in particular that proposed by an Ozinga-affiliated company at 11701 S. Torrence and so potentially imminently facing the Southeast Side community. If CDPH determines not to apply the Proposed Rules to construction and demolition materials more broadly, it could still address slag grinding here by revising the Proposed Rules to cover reprocessing of materials from other industrial processes, along with construction and demolition materials. As discussed in comments on CDPH's legal authorities and obligations submitted by our partners, CDPH has ample authority and a duty to address slag grinding now.

a. *Gap in CDPH regulations for facilities handling and/or processing "new" or frontend construction materials*

In its 2014 bulk material rules, CDPH created an exemption for "Construction and Demolition Materials," as well as "materials used in manufacturing cement at a facility that has obtained [permits and approvals from the Illinois EPA]" from the definition of "Bulk Solid Material."[37] Those 2014 rules further defined "Construction or [sic] Demolition Material" as "material used in or resulting from the construction … of utilities, structures, buildings, and roads, including but not limited to stockpiles of crushed stone, sand and gravel, hot mix asphalt plants or ready mix concrete plants." CDPH explained the exclusion by claiming that such facilities are "already subject to numerous local regulatory and permitting restrictions."[38] At the same time, CDPH only stated with respect to new construction materials that "[*c]onstruction site stockpiles* are

[36] Alternatively, CDPH could do so by amending the definition of "Bulk Solid Material" in the January 2019 Rules for Control of Emissions from Handling and Storing Bulk Materials to eliminate the prior, inadequately supported exclusion for construction and demolition materials.
[37] *See* CDPH, Rules and Regulations for Control of Emissions from Handling and Storage of Bulk Material Piles [sic], March 2014, at Section 2.0(3) and same regulations, January 2019 version.
[38] *See* CDPH, Official Response to Public Comments on the Proposed Rules and Regulations for the Handling and Storage of Bulk Material Piles, March 13, 2014, at 3 and 7.

subject to numerous limitations and requirements to prevent dust and other nuisances" (emphasis added)[39]; it did NOT discuss existing local regulatory and permitting restrictions that apply to new construction materials at facilities that handle these materials but that are not themselves "construction sites." Such facilities include, but are not limited to, concrete and aggregates companies.

Thus, the 2014 Dust Rules and subsequent revisions may be read as excluding facilities engaged in handling and processing of new construction materials from CDPH's dust requirements that otherwise apply to a wide range of facilities and materials. And because the Proposed Rules apply only to facilities that handle reprocessable construction and demolition materials, it appears that they, too, omit such facilities from needed local protections.

b. *Impacts from facilities handling "new" construction materials on environmental justice communities*

This omission and ongoing gap/exclusion is particularly disturbing in light of the history of problems with concrete, aggregates and similar companies in Chicago EJ communities, problems that have not be adequately addressed by state and federal requirements. Some examples include:

- MAT Asphalt, which has generated numerous community complaints, to which agencies are finally responding[40]
- Ozinga, against which U.S. EPA has brought an enforcement action over fugitive dust[41]
- Chicago Rail and Port, LLC, against which U.S. EPA has brought an enforcement action over fugitive dust[42]

We are also aware of numerous other community complaints against other similar facilities located in environmental justice communities, a full accounting of which is beyond the scope of these comments. Moreover, the continued gap/exclusion is deeply troubling given the apparent systematic increase in such activities in Chicago EJ communities, including but not limited to:

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[39] *Id*. at 3.

[40] *See, e.g*., Ex. 16. Brett Chase, Feds jump into McKinley Park asphalt plant fracas, Chicago Sun-Times, July 28, 2021, available at https://chicago.suntimes.com/2021/7/28/22598956/mat-asphalt-mckinley-park-epa-environmental-protection-dick-durbin-tammy-duckworth-michael-tadin.

[41] *See* Ex. 17, U.S. EPA, EPA in Illinois, Ozinga Ready Mix, Inc., available at https://www.epa.gov/il/ozinga-ready-mix-inc.

[42] See Ex. 18, U.S. EPA, Notice of Violation, EPA-5-18-IL-10. April 20, 2018, available at https://www.epa.gov/sites/default/files/2018-06/documents/chicago_rail_and_port_llc_nov.pdf. We cite this example to the extent that Chicago Rai and Port was handling, is handling, or is proposing to handle new construction material or reprocessed construction material from sources other than construction and demolition, recognizing that Chicago Rail and Port is one of the proposed reprocessable construction and demolition material facilities identified by CDPH.

- Increasing activity over the past several years at three Ozinga sites on the Southeast Side, including those at 95th, 103rd [43], and S. Torrence, including a recent proposal to combine waste or other materials with slag
- The proposed Invert facility that would be located between the RMG and KCBX sites; that would process and handle limestone mined at the site; and whose recipients/customers for the limestone have been left largely unaddressed by the company's public statements to date[44]

In sum, these facilities have disproportionately impacted environmental justice communities and will continue to do so unless and until the City takes affirmative action to address the full range of construction and demolition material facilities.

c. *Basis and need for adopting regulations for facilities processing and/or handling new construction materials*

As set forth in our partners' comments, CDPH has extensive authority to regulate impacts from facilities handling and/or processing new construction materials.[45] Given the impacts of these facilities in general and on environmental justice communities in particular, CDPH has an obligation to adopt such regulations.

Nor can the City rely solely on the enhanced site plan process under the March Air Quality Zoning Ordinance to address the impacts of such facilities. First, this site plan review process does not appear to apply to existing facilities that do not modify or expand their businesses. CDPH regulations, including the Proposed Rules, generally do require such "Existing" facilities to submit detailed applications, meet various performance requirements, and overall demonstrate that they will not pose unacceptable impacts to public health or the environment.[46] Second, the site plan review is a new and untested process that has no demonstrated track record of

[43] Regarding the 103rd St. facility, we note that according to a CDPH inspection report, in February 2021 Inspector Sampra went to 2255 E 103rd in response to a diesel odor complaint. At that time, the inspector noted constant truck traffic going back and forth, saying there was "no relief" from the diesel odor. We note that this location is a short distance to the East from the Ozinga facility at 1818 E 103rd, though it is unclear to us whether the truck traffic is associated with that Ozinga facility or with the adjacent NS Thoroughbred Bulk Transfer Terminal. Aerial satellite images available through Google Earth show the Thoroughbred facility north of 103rd with significant container storage in a May 2021 image; other available historic satellite images do not show such containers onsite. Regardless of which facility the trucks are serving, the trucks are creating unrelenting diesel exhaust in the residential neighborhood on 103rd, which the City should take into account in considering any new air emissions in this area and with respect to abating existing disparate burdens.

[44] *See, e.g.*, Ex. 19, The Invert, FAQs, available at https://theinvertchicago.com/faq/ (response to "Do you really just want to sell the material that would be removed from the site?").

[45] *See also* CDPH, Rules and Regulations for Control of Emissions from the Handling and Storage of Bulk Material Piles, 2014, Preamble (laying out the authority for adopting the 2014 rules).

[46] *See, e.g.,* Proposed Rules at 9 ("Existing Facilities must apply to renew their permit on an annual basis prior to the expiration of their current permit" and "Existing facilities are required to submit all the information required by Section 11-4-1930 of the Code and outline in Section 3.0 of these rules [with the exception of facilities qualifying for a Short Application Form]") and Section 2.0, Definitions (definitions of "Existing Facility," "Expand" or "Expansion," and "Modify or Modification").

producing community-protecting, enforceable requirements. Third, the site plan review process does not appear to require the same depth and breadth of inquiry or requirements as CDPH has outlined in the Proposed Rules and other similar regulations.

VI. Conclusion

In closing, the Proposed Rules recognize the need to address the many impacts of reprocessable construction and demolition material facilities on public health and the environment. We strongly urge CDPH to ensure that it addresses the full slate of impacts from these and closely analogous facilities – especially their disproportionate impact on environmental justice communities – in final rules, consistent with these comments and those of our partners.

Sincerely yours,

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ENVIRONMENT

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Special Report: U.S. air monitors routinely miss pollution - even refinery explosions

By Tim McLaughlin, Laila Kearney, Laura Sanicola

(Reuters) - When explosions ripped through a Philadelphia oil refinery last year, the shock waves knocked Felicia Menna's front door frame out of place. Then came the black smoke.

FILE PHOTO: A plume of smoke emits from a fire that broke out at a Chevron refinery in Richmond, California August 06, 2012. REUTERS/Staff/File Photo

“My throat was closing shut,” recalled Menna, who lives about a mile away. “My nostrils felt like they were on fire.”

She went to an emergency room, where doctors put her on a vaporizer device to ease her breathing and treated her with intravenous Benadryl for allergic reactions, according to medical records she provided to Reuters. She was among several dozen people who sought treatment after the blast, according to a neighborhood group that tracked affected residents.

One of the explosions was so large that a National Weather Service satellite captured images of the fireball from space. Refinery owner Philadelphia Energy Solutions later told regulators that the blasts released nearly 700,000 pounds of hazardous chemicals, including butane, and about 3,200 pounds of hydrofluoric acid, which can cause fatal lung injury in high concentrations. The incident remains under investigation by the U.S. Chemical Safety Board.

Yet the federal air quality index (AQI) score for south Philadelphia showed that day as one of the year's cleanest, according to data from the U.S. Environmental Protection Agency (EPA). The score was based on readings from part of the federal network of air quality monitoring devices, which are operated by the city of Philadelphia with oversight from state regulators and the EPA. None recorded any significant pollution.

"To say there was no impact to air quality was crazy," said Peter DeCarlo, an environmental engineering professor at Johns Hopkins University who lived in Philadelphia at the time and studied the city's monitoring system.

The episode illustrates a much broader failure of the U.S. air-pollution monitoring system, according to a Reuters examination of data from the EPA and independent monitoring organizations, along with interviews with scientists and environmental researchers. The government network of 3,900 monitoring devices nationwide has routinely missed major toxic releases and day-to-day pollution dangers, the data show.

The network, for example, identified no risks from 10 of the biggest refinery explosions over the past decade, the Reuters review of EPA data shows, even as thousands of people were hospitalized and the refineries reported toxic emissions to regulators.

Reuters also reviewed data from 10 community-based air monitoring projects by residents worried that government air-quality assessments are inaccurate. Those efforts often revealed pollution spikes and hot spots the EPA network never captured.

About 120 million Americans live in counties that have no EPA pollution monitors at all for small particle pollution, according to agency data. That was the case when an oil refinery in Superior, Wisconsin exploded in 2018, causing a leak of 17,000 barrels of asphalt and blanketing Superior and neighboring Duluth, Minnesota in clouds of black smoke. Though Superior has Wisconsin's only refinery, the city of 27,000 people isn't big enough to require permanent government air-pollution monitors nearby, said a spokeswoman for the Wisconsin Department of Natural Resources, citing EPA guidelines.

Fine particles - measuring less than 2.5 microns - are far smaller than a grain of sand and are considered the most dangerous form of pollution because they penetrate the bloodstream and

cause lung and heart disease. Major sources include power plant and industrial smoke stack emissions, as well as vehicle exhaust.

The system's failures pose a public health risk, independent scientists say. The monitors underpin the Air Quality Index that many Americans, including those with respiratory disease, rely on to determine whether the outdoor air is safe. Pollution detected - or missed - by the monitors also guides regulatory decisions on whether new or expanded industrial projects can be permitted under the National Ambient Air Quality Standard. If pollution in the area is below regulatory thresholds, the projects generally go forward.

The data also inform and justify environmental policy decisions - and have often been used by President Donald Trump to tout his environmental record. Trump has cut back on policies aimed at addressing climate change by limiting carbon emissions. In his losing re-election campaign, he referred to the AQI this year when he asserted that America has the world's cleanest air. A leading Yale University study, produced annually, ranks the nation 16th for air quality globally.

President-elect Joe Biden, a Democrat, has said he would step up prosecutions for illegal polluting; push for a worldwide ban on government subsidies for fossil fuels; tighten fuel economy standards for vehicles; and put limits on methane pollution from oil and gas facilities.

The EPA declined to comment on the monitors' performance during specific pollution events, including the refinery explosions examined by Reuters, but said the network was generally accurate and reliable. "We are confident that the monitoring network provides data that allows decision-makers - states, public health officials, etc. - to make informed decisions on public health" and the permitting of plants in polluting industries, the EPA said in a statement.

The EPA oversees the network of pollution-monitoring devices, which are maintained and operated by state and local environmental agencies, who also share the financial burden. With probes that suck in air, the devices use filters, light pulses and beta rays to detect gas and particle pollution so tiny that concentrations are measured in parts per billion.

Academics, along with current and former regulators, say the network's problems are many and varied: Monitors are sparsely and poorly placed; the program is underfunded; and the network is not equipped to meet current pollution threats. The monitoring program emerged piecemeal after the 1970 Clean Air Act, mainly to track acid rain, smog and ozone pollution. Those hazards have largely subsided, replaced by more localized threats including toxic compounds and particulate matter from a wide range of industry and natural hazards, such as wildfires.

Individual monitors have also proven inaccurate, often recording pollution levels that can vary wildly from audit monitors placed beside them, according to government quality-assurance audits. Nearly half of the country's monitors meant to capture fine particulate matter did not meet federal accuracy standards, an EPA audit released in 2015 found.

When explosions rocked the Philadelphia refinery, the closest monitor for hazardous chemicals was programmed to operate only one of every six days - and therefore missed the incident entirely, according to EPA data reviewed by Reuters. Other Philadelphia monitors were either upwind or too far away to detect the explosion's pollution, according to the EPA data, which shows wind direction and speed. The refinery owner, Philadelphia Energy Solutions, filed for bankruptcy after the explosion and sold the property this year to a Chicago developer that plans to convert it to a mixed-use industrial park.

It wasn't the first time monitors programmed to operate sporadically missed pollution from a major explosion. When Chevron Corp's refinery in Richmond, California, caught fire in 2012, clouds of particulate matter forced 15,000 people to seek treatment, according to the U.S. Chemical Safety Board.

But the closest government monitor of hazardous chemicals recorded no problems because it was turned off. It had been programmed to work one of every 12 days, according to EPA data. The EPA and local regulators told Reuters that certain types of monitors are designed to operate only occasionally to reduce costs and labor. In 2013, Chevron agreed to pay $2 million in fines and restitution after pleading no contest to six misdemeanor criminal charges in connection to the fire.

Monitors are also sometimes programmed to limit the level of pollution recorded. A government monitor in Imperial County, California, operated by local and state regulators, recorded much lower readings of day-to-day air pollution in 2017 than were actually occurring because it had been programmed to max out at a lower level. The EPA acknowledged the issue to community organizations after the groups discovered higher readings with their own monitors.

“It’s almost unbelievable this can happen in the United States,” said Michael Jerrett, chair of the environmental health science department at the University of California, Los Angeles, and an adviser on the community monitoring project.

Researchers from the University of California San Francisco did a post-mortem on the Chevron refinery fire as part of a community health study. They concluded many of the people who suffered initial health problems continued to have worsening health in the years after, including chronic respiratory issues such as asthma.

Chevron said in a statement that it has worked since the 2012 fire to improve safety, reduce pollution and provide the community real-time data on air quality around its refinery. “Chevron recognizes the value of complete and accurate air quality data,” the company said.

In south Philadelphia, Menna said her initial symptoms from the blast’s fallout wore off in about a week, but she continued to cough for six months.

“I still don’t know if I have long-term effects,” she said.

UNDERFUNDED SYSTEM

A study conducted in 2013 during the administration of President Barack Obama, a Democrat, detailed a number of problems with the U.S. air monitoring network. The report proposed improvements including boosting monitoring near major polluting infrastructure, sampling for more pollutants, and doing more urban field studies to better understand block-to-block variability in air quality. But the weaknesses largely remain today because neither the Obama nor the Trump administration invested more in the monitoring network.

Slideshow (4 images)

Over the past five years, the number of government monitors nationally has declined by 4% as state and local environmental agencies cut spending, according to EPA figures. Federal grants to state and local air-quality agencies have not increased in 15 years, according to testimony earlier this year by the National Association of Clean Air Agencies, a nonpartisan group based in Arlington, Virginia.

“The public’s desire for pollution data is exploding, but the government has less resources,” said Lyle Chinkin, chief scientist at environmental research firm Sonoma Technology, who has testified for the EPA in lawsuits accusing coal plant operators of Clean Air Act violations.

The EPA said it has improved the system despite what it acknowledged was flat funding for the past decade. The agency said it has replaced some labor-intensive, manual monitors with automatic monitors that provide round-the-clock, real-time data. The continuous monitors cost less to operate, but can also be less reliable than manual monitors in measuring particulate matter, according to EPA quality control audits.

Local groups worried about air quality have been trying to fill the gaps.

A community project in New York City, for example, has deployed up to 150 air monitors over the past decade. It found small particle pollution from traffic has been 50% higher in low-income neighborhoods than wealthier ones because they tend to be closer to major thoroughfares. By contrast, the EPA network run by state regulators in New York City has less than 30 monitors, preventing the EPA from providing city neighborhoods with a granular view of air quality, said Holger Eisl, director of the community project.

In Imperial County, California, the predominantly Latino community had long suspected government monitors were not giving a true reading of local pollution from agricultural burning and factories across the border in Mexico. An organization called Comite Civico del Valle installed 40 of its own monitors in 2015 to compare against the handful of government monitors. The devices detected sky-high levels of coarse particle pollution, at times exceeding the worst days in Beijing, among the world's most polluted cities. Course particle pollution, produced by activities including wildfires and farming operations, can increase risk of heart and lung diseases.

The 24-hour maximum level of coarse particulate matter recorded by the community monitors surged as high as 2,430 micrograms per cubic meter in 2017, according to project organizers. That's 40 times greater than the World Health Organization's recommended level. The nearest government monitor, however, showed concentrations of only 985 micrograms per cubic meter, according to EPA data. Researchers discovered, after consulting with the EPA, that the government monitor had been programmed to record nothing higher than 985 micrograms.

"We exposed them many times by finding things the government monitors were not finding," said Luis Olmeda, executive director of Comite Civico del Valle.

The EPA acknowledged the monitors' default setting was capped. It said the manufacturer warned that using higher settings can impair readings of lower pollution levels. After learning of the high readings on the community monitors in Imperial County, state and county environmental officials adjusted the area's monitors to capture pollution levels up to 10,000 micrograms. The EPA detailed the change of settings in September when it ruled that the

county's air had improved enough to comply with federal regulations on coarse particle pollution.

Overall, between October 2016 and February 2017, the community monitors detected 1,426 episodes of elevated levels of particulate matter, or 12 times what government monitors recorded. The EPA ruled in October that Imperial County meets clean air standards. The agency excluded nearly 100 days of excessive pollution between 2014 and 2018, saying sand and dust storms in the desert area were "exceptional events."

Community groups in Baltimore; Albany, New York; and East Oakland, California, have also independently found pollution missed by the EPA system. In Baltimore's Curtis Bay neighborhood, community monitors revealed 24% higher fine particle pollution than government monitors, according to 2015 results published by the nonprofit Environmental Integrity Project.

Even small increases in exposure to particle pollution within a city can significantly increase the progression of heart disease, even if the levels remain below federal standards, according to Joel Kaufman, a physician-epidemiologist at the University of Washington and editor-in-chief of Environmental Health Perspectives.

The EPA acknowledged that community monitoring programs had been useful in identifying hot spots. But the EPA added that the low-cost monitors sometimes used by community groups have cheaper components and can have higher error rates than government monitors, and may not operate as well in harsh climates.

Government monitors also have problems. EPA performance evaluations have identified a long-running trend of imprecision and a bias toward undercounting pollution levels, according to the agency's 2015 audit report. The audit covered about 1,000 government fine particulate matter monitoring sites, operated by nearly 100 environmental agencies. It found that 46% of the agencies had monitors that failed to meet the EPA's standard for precision and 44% of agencies had devices that failed the bias standard.

In a statement, the EPA said the network's accuracy has since improved, and that 21% of agencies had monitors that failed to meet its precision standard between 2017 and 2019 and

39% had monitors that failed its bias goal.

TOSSING RESULTS TO HELP INDUSTRY

When EPA monitors capture pollution that exceeds regulatory limits, the EPA sometimes throws out those results for the purposes of its air-quality assessments - clearing the way for industrial development.

Trump's economic agenda has included fast-tracking the re-designation of areas of the country that are out of compliance with pollution standards, sometimes redrawing maps to exclude certain air monitors. Nationally, the administration has re-designated 54 out-of-compliance areas since 2017. Some of its decisions have been thrown out by the courts as arbitrary.

In Sheboygan, Wisconsin, for example, a court rejected the EPA exclusion this year of a monitor recording high ozone levels near the shoreline of Lake Michigan so that part of the surrounding county could be reclassified as complying with federal clean air standards. The EPA justified the move by arguing the monitor was unduly influenced by pollution coming from elsewhere via "lake breezes."

The EPA said the re-designations reflect greater progress toward cleaner air.

Industry can also benefit from the placement of monitors - a process that polluting companies can influence, said Corbett Grainger, a University of Madison-Wisconsin environmental economics professor who led a study of monitor site selection.

The EPA provides guidance on where monitors are placed, but state regulators have wide discretion. The Wisconsin researchers found that state regulators in counties that are close to exceeding pollution standards often place monitors in cleaner areas when they have the option, a conclusion based on a study of years of EPA monitor data and pollution estimates from satellites.

"We found that, on average, newly sited monitors are placed in relatively clean areas," said Grainger, the Wisconsin environmental economist. The positioning, he said, suggests that local regulators strategically avoid pollution hot spots.

The EPA declined to comment on the study.

In 2015 and 2016, Missouri regulators allowed St. Louis-based utility Ameren Corp to select sites to install four sulfur dioxide (SO2) monitors around its Labadie coal plant. The plant is ranked by the EPA as the second largest SO2 polluter in the country. The EPA and state regulators signed off on the monitoring sites as accurately capturing the plant's pollution - over the objections of environmental groups that argued the locations would prevent monitors from picking up the coal plant's peak SO2 concentrations.

Ameren told state regulators it followed EPA guidelines in locating the monitors. The company declined to comment for this story.

The Missouri Department of Natural Resources said choosing the locations was a collaborative effort with the company and the EPA, and that regulators reviewed and verified Ameren's analysis of the sites. "It's not unusual for facilities to submit their own analysis," the department said.

In August, the EPA told Missouri's governor that it plans to move ahead with redesignating the area around Labadie as in compliance with pollution standards.

But pollution from the plant travels far beyond the surrounding area, said Chinkin, the atmospheric scientist. Based on a computer simulation, he said in court testimony in 2019 that Labadie's SO2 output converts to fine particulate matter because of the heat and humidity during summer in St. Louis.

The result, Chinkin testified, is particulate pollution that extends across the entire eastern half of the United States. The worst impacts, he said in a phone interview, can be seen "hundreds of miles beyond Missouri."

Reporting by Tim McLaughlin, Laila Kearney and Laura Sanicola; Editing by Richard Valdmanis and Brian Thevenot

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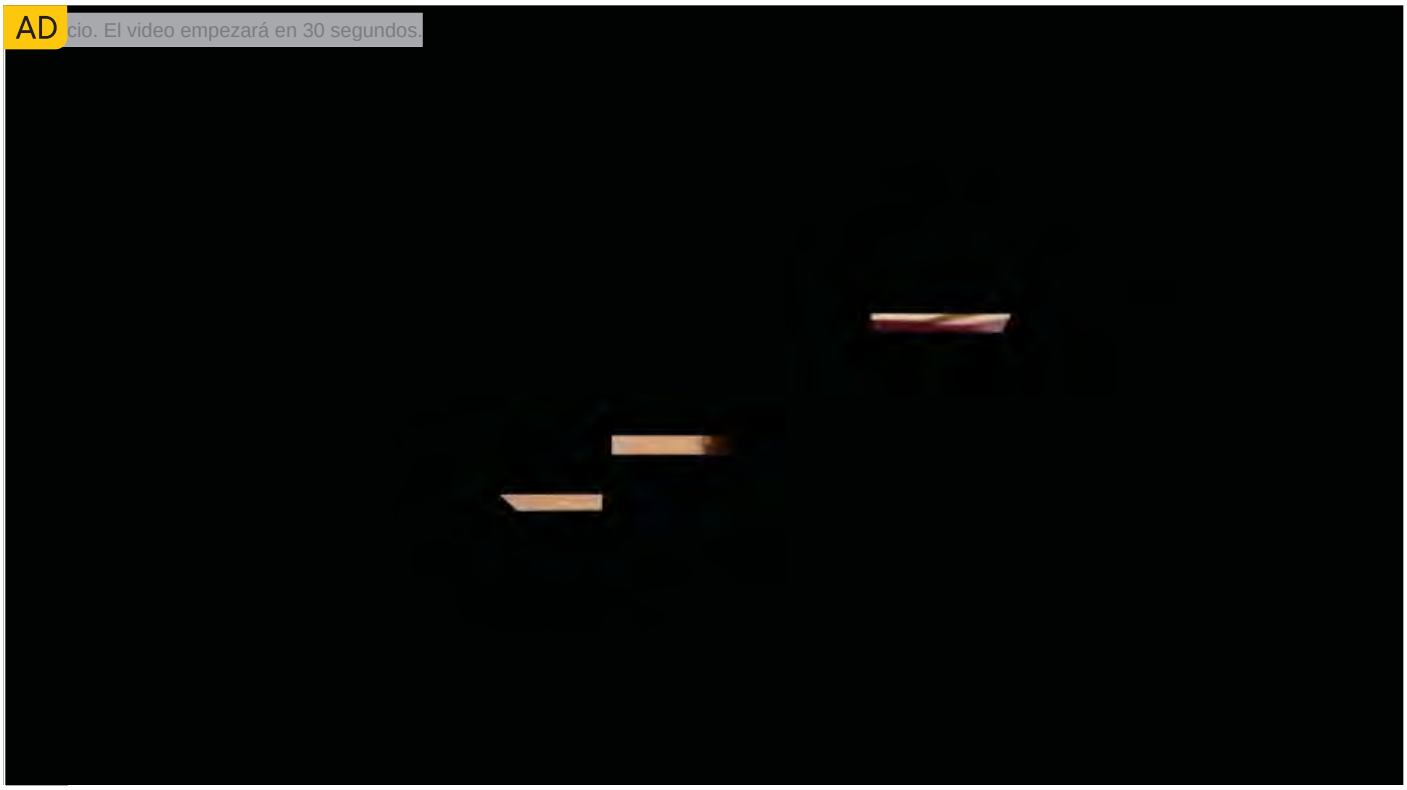

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EPA to Reexamine Health Standards for Harmful Soot that Previous Administration Left Unchanged

June 10, 2021

Contact Information
EPA Press Office (press@epa.gov)

WASHINGTON (June 10, 2021) — Today, EPA announced that it will reconsider the previous administration's decision to retain the particulate matter (PM) National Ambient Air Quality Standards (NAAQS), which were last strengthened in 2012. EPA is reconsidering the December 2020 decision because available scientific evidence and technical information indicate that the current standards may not be adequate to protect public health and welfare, as required by the Clean Air Act.

"The most vulnerable among us are most at risk from exposure to particulate matter, and that's why it's so important we take a hard look at these standards that haven't been updated in nine years," **said EPA Administrator Michael S. Regan.** "EPA is committed to ensuring this review, and other upcoming NAAQS reviews, reflect the latest science and public health data."

The strong body of scientific evidence shows that long- and short-term exposures to fine particles (PM2.5) can harm people's health, leading to heart attacks, asthma

attacks, and premature death. Large segments of the U.S. population, including children, people with heart or lung conditions, and people of color, are at risk of health effects from PM2.5. In addition, a number of recent studies have examined relationships between COVID and air pollutants, including PM, and potential health implications. While some PM is emitted directly from sources such as construction sites, unpaved roads, fields, smokestacks or fires, most particles form in the atmosphere as a result of complex reactions of chemicals such as sulfur dioxide and nitrogen oxides, which are pollutants emitted from power plants, industrial facilities and vehicles.

EPA's 2020 Policy Assessment concluded that the scientific evidence and information support revising the level of the annual standard for the PM NAAQS to below the current level of 12 micrograms per cubic meter while retaining the 24-hour standard. The agency also received numerous petitions for reconsideration as well as lawsuits challenging the December 2020 final action.

EPA will move expeditiously to reconsider the decision to retain the particulate matter NAAQS, in a manner that adheres to rigorous standards of scientific integrity and provides ample opportunities for public input and engagement. As part of this process, the agency will develop a supplement to the 2019 Final Integrated Science Assessment (ISA) that will take into account the most up-to-date science, including new studies in the emerging area of COVID-related research.

This supplement will be reviewed at a public meeting by the chartered Clean Air Scientific Advisory Committee (CASAC), supported by a particulate matter review panel of scientific experts on the health and welfare impacts of PM. The CASAC and the PM panel will also review a revised policy assessment and formulate advice to the Administrator. As with all reviews, the public will have opportunities to comment on these documents during the CASAC review process, as well as to provide input during the rulemaking through the public comment process and public hearings on any proposed decision.

EPA expects to issue a proposed rulemaking in Summer 2022 and a final rule in Spring 2023, following an open, transparent process with opportunities for public review and comment. In accordance with Executive Orders and guidance, the agency will be considering environmental justice during the rulemaking process.

For more information on the NAAQS review process and documents related to prior PM NAAQS reviews, visit https://www.epa.gov/naaqs/particulate-matter-pm-air-quality-standards <https://epa.gov/naaqs/particulate-matter-pm-air-quality-standards>

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LAST UPDATED ON JUNE 10, 2021

November 12, 2019

Administrator Andrew Wheeler
U.S. Environmental Protection Agency
1200 Pennsylvania Avenue NW
Washington D.C. 20460

Sent via Regulations.gov.

Re: Comments on Docket ID No. EPA-HQ-OAR-2015-0072

Dear Administrator Wheeler:

The American Lung Association appreciates the opportunity to provide comments on the Policy Assessment for Review of the National Ambient Air Quality Standards for Particulate Matter, External Review Draft, September 2019 (Draft PA).

EPA's revised review process undermines the protection of public health

The Lung Association continues to express our objections to the changes to the process that EPA has adopted in this review. EPA's changes restrict the full discussion and review of the information, undermining the core purpose of this process: to set standards that "protect health with an adequate margin of safety." While the Lung Association has long supported and, indeed, taken legal action to ensure the completion of the reviews in a timely manner, the Lung Association opposes the current process because it undermines the ability of CASAC and EPA to arrive at appropriate and adequate decisions on these standards. The revised process threatens that the decisions you make as Administrator would not be based on a thorough review of the evidence.

The review process adopted in 2006 followed an open, deliberative discussion led by CASAC of the changes needed to improve the process. Based on the desire to provide an informed and robust assessment of the information, EPA established a protocol that included separate reviews of two separate drafts of critical documents, including a separate Risk and Exposure Assessment (REA). That process ensured that EPA would have reached conclusions on the scientific evidence about health and welfare impacts before beginning work on the policy implications. CASAC had agreed to such a plan for the review of the PM NAAQS in 2016, at the

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beginning of this process. EPA upended that agreement and reversed the long-established process with no public or CASAC review of the proposed changes.

EPA has set up an unprecedented, flawed process to truncate the review of the particulate matter and the ozone NAAQS. With these changes, critical information that forms the basis of the decisions is absent or unresolved. The Draft PA for particulate matter should have been fully informed by the Integrated Science Assessment (ISA). EPA should not have released the Draft PA until EPA finalized the ISA. Until the ISA is final, no final, reliable determination of the air quality criteria exists; that is, there is no full conclusion on the information which "accurately reflect[s] the latest scientific knowledge useful in indicating the kind and extent of all identifiable effects on public health or welfare which may be expected from presence to such pollutant in the ambient air." 42 U.S.C 7408(a)(2).

Without first finalizing the ISA, EPA impairs the determination of the relevant policy decisions in accessing the Draft PA. EPA's process unacceptably handicaps the review.

Further, EPA dismissed the established independent advisory panel for particulate matter, a step that cost the CASAC and EPA essential expertise in the complex avenues that the documents explore. Such panels had served multiple CASAC reviews for decades. EPA offered flawed arguments for eliminating the in-place, working panel: that under the Clean Air Act, only CASAC can advise EPA, and that such elimination would expedite the review. Both arguments are specious. The independent panels have always provided expert assistance only to CASAC, which CASAC then used to advise EPA. The panels have worked closely with CASAC to assist in an accurate and thorough scientific review following the adopted schedule.

Not until CASAC itself acknowledged its limitations and requested assistance in a letter to you did EPA take limited steps to provide additional assistance.[1] EPA appointed a new pool of advisors who lacked experience in the NAAQS review process and PM, and then mangled the process again by limiting advisory actions to one single panelist's opinion, by letter, in response to written questions. This restricted process eliminated the traditional approach that provided a more complete and open discussion with multiple, experienced panelists who contributed independent perspectives and deliberated their consensus recommendations on topics throughout the reviews of each document.

The shining light in this damaged process is the diligence and thoroughness of the EPA staff in preparing this assessment. In general, there is much to support in their assessment. They have attempted to provide a full, extensive review, albeit based on the draft ISA, which should provide added evidence to EPA that more protective standards are needed for particulate matter.

The Lung Association strongly urges EPA to issue a second draft PA, reappoint the independent CASAC advisory panel and restore the process that the Agency had previously followed to complete this review to protect public health. The Lung Association also urges that EPA publish the final ISA before releasing a second draft PA.

Millions of Americans face increased risk from particulate matter

The Lung Association agrees with the finding in the draft PA, that "a substantial portion of the U.S. population" face increased risk from breathing particulate matter. The Lung Association supports recognition of these groups as at risk, as mentioned in the draft PA: children and teenagers; older adults; people with chronic lung diseases or cardiovascular diseases; people who are overweight

or obese; people with specific genetic variants; Hispanics; non-Hispanic blacks; and people who have low incomes. Evidence also indicates that healthy adults who work and exercise outdoors also face higher risk. The Lung Association particularly calls attention to the new evidence showing that African Americans face a three-time higher risk from PM than the entire population.[2] This adds to the evidence that African Americans, Hispanics and low-income communities face higher risk because social and environmental disparities.

The Lung Association is acutely concerned about the impacts to millions of people with lung diseases. In 2017, estimates show that 15.3 million adults had chronic obstructive pulmonary disease (COPD) and 25.2 million Americans had asthma, including 6.2 million children.[3] In 2019, more than 228,000 Americans will be diagnosed with lung cancer.[4]

All Americans deserve to have their health protected by stronger PM standards.

The current fine particulate standards fail to protect public health

One of the key findings in this Draft PA is overwhelming evidence that the current annual fine particulate ($PM_{2.5}$) standard fails to protect public health. The Lung Association strongly supports that conclusion, as discussed below. However, strong evidence exists that the 24-hour $PM_{2.5}$ standard also fails to provide that required protection to public health. The Lung Association urges EPA to strengthen both the annual and the 24-hour $PM_{2.5}$ standards.

Today, more robust evidence than ever before supports the increased risk of premature death from levels below the current standards. No evidence exists of a threshold to that risk, as EPA has acknowledged. Recent U.S. studies that restrict the analysis to long-term exposures below 10 $\mu g/m^3$ [5] and Canadian studies that find evidence down to and below 8 $\mu g/m^3$ [6] all found premature deaths at those lower levels. A large study looking at short-term exposures also added evidence of harm below the current annual standard.[7] These studies offer new information that answered questions in previous reviews. The Canadian studies, in particular, offer data on low levels of exposures over long periods.

The Draft PA estimates that the current standards allow "a substantial number" of deaths from $PM_{2.5}$ exposure in the U.S. today. The analysis in Table 3-5 estimates that, using 2015 air quality data adjusted to just meet the 2015 annual standard, the median estimated annual mortality ranges from 13,500 to 52,100 premature deaths.[8] The evidence extends to the 24-hour standard, where even with the current standard, the median estimated mortality from short-term exposures ranges from 1,200 to 3,870 premature deaths annually.[9] These studies show consistent evidence that the current standards allow significant and unacceptable increased risk to health, unquestionably failing to provide the legally required protection. In addition, EPA's estimates do not explore the evidence of harm from multiple other health effects of PM exposure, including COPD and asthma exacerbations. The Draft PA therefore does not consider the full impact of these inadequate standards.

The vast evidence that $PM_{2.5}$ shortens lives remains consistent as it has from the landmark studies in the 1990s. The follow-up studies that further tracked those cohorts mirror the findings in newer studies looking at lower levels of pollution[10] including some with one of the largest data sets in history, the 61 million people in the Medicare cohort.[11] The research included studies that examined the evidence using different ways of accessing exposure,[12] diverse regions of the country and diverse populations,[13] and different statistical models.[14]

We strongly disagree with the specious arguments provided in the Draft PA to create some justification for retaining the current standard. The powerful evidence from these epidemiological studies alone undercut these rationalizations of uncertainty. The evidence from the toxicological studies and clinical trials support these conclusions. The lack of studies examining exposures down to zero create no valid uncertainty over whether reducing PM levels further would add to the health benefits. We urge EPA to remove those arguments and that option from consideration.

We also disagree with EPA's evaluation that the 24-hour $PM_{2.5}$ standard provides sufficient protection for public health. EPA argues that its primary use is to supplement the annual standard. Even with that role, that combination fails to provide protection for many communities across the nation where the annual level is quite low. For communities in Alaska, parts of the Northwest and parts of New England, shorter-term exposures pose the primary risk because of the emissions from woodstoves and other sources that create elevated levels of $PM_{2.5}$. Nearly all these areas have year-round concentrations that are well under the annual standard. The risk assessment in the Draft PA did not include areas in these parts of the nation, limiting the assessment of exposures to these sources.

Further, while wildfires do pose a significant source of the 24-hour exposures at high levels, they should not be dismissed as not caused by human action. Droughts exacerbated by climate change and fires from flawed electrical infrastructure as seen in California this year provide two examples of the human actions that contribute to the expanding burden of wildfires in the United States. Research supports this. In a study examining wildfires nationwide, researchers estimated that human activities caused 84 percent of wildfires between 2009 and 2012.[15]

The Lung Association continues to support changes to the form of the short-term standard, recommending at 99th percentile rather than at the 98th. The Draft PA continues to argue that the 98th percentile offers more stability to the standard than would at 99th percentile standard. While that is true, stability fails public health protection when the 98th percentile allows as many as 21 days to be exempted before meeting the standard.

Further, the Lung Association urges the consideration of a rolling 24-hour standard, rather than one that covers the 24 hours of a single calendar day. As one of the main sources of 24-hour PM spikes, woodstove smoke often peaks during shorter, overnight periods that may not be appropriately captured in the split that occurs at midnight. The Lung Association also urges EPA to require $PM_{2.5}$ monitoring to be a continuous monitoring network. The continued use of monitoring limited to data capture only every three or six days adds to the gaps in protection that Americans deserve from this deadly pollutant.

The coarse particulate standard should be strengthened

The PM_{10} standard has stood in place, unaltered, since its adoption in 1987. That comes despite the long recognition that, while intended to provide protection against the coarse particles ($PM_{10-2.5}$), inadequate monitoring and research limit the standard's ability to protect health from these particles. In 2010, CASAC advised EPA to consider a stronger PM_{10} standard with a different form (98th percentile) and a more protective level, down to 65 µg/m^3.[16] This Draft PA did not examine that recommendation.

EPA should reconsider the CASAC 2010 recommendation on PM_{10} in a second Draft PA. The Lung Association supports strengthening the standard, based on a more complete review of the 2010 CASAC recommendations. In addition, EPA should expand the monitoring data for the coarse

fraction ($PM_{10-2.5}$), to provide sufficient information to assess exposure and health risks that may be different from those solely of the fine particle fraction.

Recommendations for proposed $PM_{2.5}$ standards

Real world studies demonstrate that the current standards fail to protect health. EPA must adopt stronger standards based on the best available evidence that would protect health with an adequate margin of safety

Based on the information in the Draft ISA and Draft PA, the Lung Association urges EPA to strengthen the annual $PM_{2.5}$ standard to 8 $\mu g/m^3$ and the 24-hour standard to 25 $\mu g/m^3$. The Draft PA considers annual standard levels down to 8 $\mu g/m^3$ based on the current evidence. These studies—including the Medicare cohort study[17] that found mortality associated with levels as low as 7 $\mu g/m^3$—provide sufficient, robust evidence that the standard should be no higher than 8 $\mu g/m^3$ to protect public health with an adequate margin of safety.

The Lung Association recommends adopting a stronger standard of 25 $\mu g/m^3$ for the 24-hour standard, as well as changing the form of the standard to the 99th percentile.

EPA should support additional research

Research forms the basis of our understanding of the complex sources, composition, size, transmission, and health risks from particulate matter. The Lung Association urges EPA to increase research on particulate matter health impacts, including improved monitoring and health effect research on ultrafines, $PM_{10-2.5}$ and speciation, especially on respiratory health.

EPA should return to the previous review process with an experienced CASAC and independent advisory panel

The flawed process imposed on this review by EPA poses serious limits to the Agency's ability to fulfill its requirements under the Clean Air Act to protect human health. The inadequate review is now accompanied by an even more egregiously reduced review of the ozone NAAQS in an even shorter time. These reviews create an overwhelmingly impossible task for CASAC and EPA to complete in any reasonably appropriate way.

The Lung Association strongly urges EPA to restore the review process previously adopted and to restore the prior CASAC members and the prior independent panel. Only by returning to the full process that EPA abandoned will EPA be able to fulfill its duties required under the Clean Air Act.

Sincerely,

Deborah Brown
Chief Mission Officer

[1] Cox L.A. 2019. Clean Air Scientific Advisory Committee Letter to A. Wheeler, U.S. Environmental Protection Agency. CASAC Review of the EPA's Integrated Science Assessment for Particulate Matter (External Review Draft – October 2018). EPA-CASAC-19-002, April 11, 2019.

[2] Di Q. Wang Y, Zanobetti A, Wang Y, Koutrakis P, Choirat C, Dominici F and Schwartz JD. 2017. Air pollution mortality in the Medicare population. *NEJM*. 376(26): 2513-2522.

[3] Centers for Disease Control and Prevention. National Health Interview Survey, 2017. Analysis by the American Lung Association Epidemiology and Statistics Unit Using SPSS Software.

[4] Siegel Rl, Miller KD, Jemal A. Cancer Statistics, 2019. CA: A Cancer Journal for Clinicians. 2019; 69 (1):7-34.

[5] Shi I., et al. 2016. Low Concentration PM2.5 and mortality; estimating acute and chronic effects in population-based study. *Environmental Health Perspectives*, 124(1)46-52.

[6] Szyszkowicz M. 2009. Air pollution and ED visits for chest pain, *American Journal of Emergency Medicine*. 27(2): 165-168; Steib DM, et al. 2009 Air pollution and emergency department visits for cardiac and respiratory conditions: A Multi-city time series analysis. *Environmental Health: A Global Science Access Source*. 8(25):25; Weichenthal S. et al. 2016 Ambient PM2.5 and risk of emergency room visits from myocardial infarction: Impact of regional PM2.5 oxidative potential: a case-crossover study. *Environmental Health*. 15:46.; Weichenthal et al., 2016. "$PM_{2.5}$ and emergency room visits for respiratory illness: effect modification by oxidative potential." *AJRCCM*. 194(5): 577-586.

[7] Di Q, Dai L, Wang Y, Zanobetti A, Choirat C, Schwartz JD and Dominici F. 2017. Association of short-term exposure to air pollution with mortality in older adults. *JAMA* 318(24): 2446-2456.

[8] U.S. EPA, Policy Assessment for Review of the National Ambient Air Quality Standards for Particulate Matter, External Review Draft, September 2019, Table 3-5; Thurston GD, Ahn J, Cromar KR, et al. 2016. Ambient particulate matter air pollution exposure and mortality in the NOH-AARP Diet and Health Cohort. *Environmental Health Perspectives* 124(4): 484-490; Pope CA, Turner MC, Burnett R, et al. 2015.. Relationships between fine particulate air pollution, cardiometabolic disorders and cardiovascular mortality. *Circulation Research* 116(1): 108-U258.

[9] Ito K, Ross Z, Zhou J, Nádas A, Lippmann M and Thurston GD. 2013. National Particle Component Toxicity (NPACT) initiative: Study 3. Time-series analysis of mortality, hospitalizations, and ambient PM2.5 and its components. Boston, MA, Health Effects Institute**:** 95-125; Zanobetti A, Dominici F, Wang, Y and Schwartz, JD. 2014. A national case-crossover analysis of the short-term effect of PM2.5 on hospitalizations and mortality in subjects with diabetes and neurological disorders. *Environmental Health: A Global Access Science Source* 13(1): 38.

[10] U.S. EPA. Draft Integrated Science Assessment for Particulate Matter. 2018 sections 11.2.2.2, 11.2.5

[11] Di Q et al., 2017.

[12] EPA 2018, section 11.2.5.1

[13] EPA 2018, section 11.2.5.3

[14] EPA 2018, section 11.2.5.2

[15] Balch JK, Bradley BA, Abatzoglou JT, Nagy RC, Fusco EJ, and Mahood AL. 2017.Human-started wildfires expand the fire niche across the United States. *Proceedings of the National Academy of Sciences.* 114, 2946-2951.

[16] Samet J. CASAC Letter to the Administrator in Review of the Policy Assessment for the Review of the PM NAAQS – Second External Review Draft (June 2010). EPA – CASAC-10-015. September 10, 2010.

[17]Di Q et al. 2017.

CHRONIC TOXICITY SUMMARY

SILICA (CRYSTALLINE, RESPIRABLE)

(silicon dioxide, quartz, tridymite, cristobalite)

CAS Registry Number: 7631-86-9

I. Chronic Toxicity Summary

Inhalation Reference Exposure Level	**3 µg/m^3** [respirable, as defined occupationally by ACGIH (2004)/ISO (1995)
Critical effect(s)	Silicosis in miners and other workers
Hazard index target(s)	Respiratory system

II. Physical and Chemical Properties (HSDB, 2001)

Description	Transparent crystals
Molecular formula	SiO_2
Molecular weight	60.09 g/mol
Density	2.65 g/cm^3 @ 0 °C (quartz)
Melting point	1610 °C
Boiling point	2230 °C (2503.20 °K)
Vapor pressure	10 torr @ 1732 °C
Solubility	Practically insoluble in water or acids, except hydrofluoric acid; very slightly sol. in alkali.
Conversion factor	Not applicable

In crystalline silica, the silicon and oxygen atoms are arranged in a definite regular pattern throughout the crystal. The characteristic crystal faces of a crystalline form of silica are the outward expression of this regular arrangement of the atoms (HSDB, 2001). This REL is meant to be applied only to particles of crystalline silica (quartz, cristobalite, tridymite), of respirable size, as defined by the occupational hygiene methods described by ACGIH (2004)/ISO (1995) which has a 50% cut-point at 4 µm particle aerodynamic diameter. This occupational definition of respirable differs from the environmental definition of respirable, which is PM_{10}. (The occupational particle category "thoracic" has a 50% cut-point at 10 µm particle diameter (ACGIH, 2004) and the category "inhalable" has a 50% cut-point at 100 µm particle diameter (ACGIH, 2004).)

III. Major Uses and Sources

At least 11 chemically identical forms (polymorphs) have been described for crystalline silica. Alpha-quartz is the most abundant polymorph and constitutes 12% of the earth's crust (Elzea, 1997). Silica is also found in the amorphous (non-crystalline) state. The amorphous silica in diatomaceous earth (composed mainly of the cell walls of diatoms) can be converted to the crystalline form cristobalite by heating to 1000-1100 ºC (calcining). Silica is often associated

with silicates, which, in addition to silicon and oxygen, contain other metals such as iron, magnesium, aluminum, calcium, potassium, and sodium.

The major uses of silica are in the manufacture of glass, abrasives, ceramics, and enamels, in scouring and grinding compounds, and in molds for castings. Silica is also used in decolorizing and purifying oils and petroleum products; as a clarifying agent; in filtering liquids; and in the manufacture of heat insulators, firebrick, and fire- and acid-proof packing materials. As diatomite (naturally occurring diatomaceous earth), silica is used as a filtration agent, as an abrasive, and as an industrial filler. Sources of ambient respirable crystalline silica in California include mines, quarries, diatomaceous earth calcining plants, sand blasting, and entrained fines (e.g., PM_{10}) from surface soil. The annual statewide industrial emissions from facilities reporting under the Air Toxics Hot Spots Act in California based on the most recent inventory were estimated to be 2,514,981 pounds of crystalline silica (CARB, 2001. The fraction, which is respirable as defined either occupationally or environmentally, is not known.

Measurement of crystalline silica has evolved. Instrumentation has varied by country. In South Africa since the 1930s, dust was collected with a konimeter (Le Roux, 1970; Cherrie and Aitken, 1999). A small volume of air (e.g., 5 cm^3 captured in less than a second) was collected (impacted) onto a small area of a glass slide coated with adhesive. Total dust particles were counted and expressed as dust particles per cubic centimeter. Later, slides were heated to 500-550 ºC (ignition) to remove carbonaceous materials and immersed in hot 50% hydrochloric acid followed by a second ignition to remove acid-soluble materials. The remainder was mostly silica particles, which could be counted. The konimeter was superseded by the thermal precipitator, which also deposited particles onto glass but could sample larger air volumes at high flow rates (> 1 L/minute) for several hours. With time, particle counting was replaced by estimation of a particle's surface area, initially by examining slides but more recently by an automated method (Kitto, 1960; 1970).

In the United States the impinger method was used from 1922 until 1984 (Lippmann, 2001). Air was drawn into a trap containing fluid, particles in an aliquot of the fluid were counted under magnification, and concentrations were expressed as million particles per cubic foot of air sampled. Later, gravimetric analysis was introduced. Gravimetric analysis is dominated by the larger particles in any given size range.

When it was realized that only a fraction of the dust was responsible for silicosis, respirable dust was collected onto filters using size-specific dust collectors, such as horizontal plate elutriators in South Africa and cyclones in the United States. The sizes of particles collected on the filter were a function of the apparatus used and the rate of airflow through the apparatus. Quartz dust was quantified by examining filters in an electron microscope with a specific X-ray diffraction beam absorbed by crystalline silica. The National Institute of Occupational Sciences and Health (NIOSH, 2003) has approved Method 7500, which uses one of three approved cyclones and a 5 μm PVC membrane filter to sample, and X-ray diffraction to measure crystalline silica. The ARB has used Method 7500 in research projects.

In order to harmonize respirable particulate sampling methodology in workers, an international agreement has been reached to use dust samplers that have a 50% cut point for particles of 4 μm aerodynamic diameter (ISO, 1995; ACGIH, 2004).

Various attempts have been made to estimate the changes in silica levels in workplaces over time (e.g., Seixas *et al*., 1997 for diatomaceous earth facilities in California; Verma *et al*., 1989 for Ontario hard rock miners). However, although some conversion factors have been proposed, correlation between dust particle number in earlier studies, when dust concentrations were higher, and dust particle weight in the later studies, when the dust concentrations have been lowered, is imprecise so it is difficult to compare the earlier silica measurements with the more recent ones.

IV. Effects of Human Exposures

Inhalation of crystalline silica initially causes respiratory irritation and an inflammatory reaction in the lungs (e.g., Vallyathan *et al*., 1995). Acute exposures to high concentrations cause cough, shortness of breath, and pulmonary alveolar lipoproteinosis (acute silicosis). After chronic but lower workplace exposures to silica for six to sixteen years, the small airways become obstructed as measured by pulmonary function tests (e.g., decreased FEV_1) in granite quarry workers (no measurement of silica levels reported; Chia *et al*., 1992). In a report on the hazards of exposure to crystalline silica, the American Thoracic Society (1997) stated: "Studies from many different work environments suggest that exposure to working environments contaminated by silica at dust levels that appear not to cause roentgenographically visible simple silicosis can cause chronic airflow limitation and/or mucus hypersecretion and/or pathologic emphysema." Hnizdo and Vallyathan (2003) also concluded that "chronic levels of silica dust that do not cause disabling silicosis may cause the development of chronic bronchitis, emphysema, and/or small airways disease that can lead to airflow obstruction, even in the absence of radiological silicosis." Fibrotic lesions associated with crystalline silica have also been found at autopsy in the lungs of granite workers who lacked radiological evidence of silicosis (Craighead and Vallyathan, 1980).

Silicosis results from chronic exposure; it is characterized by the presence of histologically unique silicotic nodules and by fibrotic scarring of the lung. The histological progression of silicosis has been described as: (1) granuloma composed of histiocytic cells, collagen, and lymphocytes; (2) cellular fibrotic nodule with irregular collagen at the center and circular collagen at the periphery; (3) more mature nodule with acellular and avascular center; and (4) late mature nodule composed of dust and collagen including a calcified center (Green and Vallyathan, 1996). Lung diseases other than cancer associated with silica exposure include silicosis, tuberculosis/silicotuberculosis, chronic bronchitis, small airways disease, and emphysema (Oxman *et al*., 1993; Park *et al*., 2002; Hnizdo and Vallyathan, 2003; Balmes *et al*., 2003). Silica exposure has been implicated in autoimmune diseases (rheumatoid arthritis, scleroderma, systemic lupus erythematosus) in gold miners and granite workers (Steenland and Goldsmith, 1995; Parks *et al*., 1999) and in the causation of kidney disease in some occupations (Goldsmith and Goldsmith, 1993; Stratta *et al*., 2001), possibly by an immune mechanism.

At the cellular level, silica particles are engulfed in the lung by alveolar macrophages (AM). According to the generally assumed pathological model, the AM subsequently release various growth factors and reactive oxygen species (ROS; superoxide anion, hydrogen peroxide, hydroxyl radical) (Lapp and Castranova, 1993; Mossman and Churg, 1998; Ding *et al*., 2002). ROS and some growth factors (e.g., activator protein-1, platelet activating factor) are inflammatory and attract neutrophils to the site of inflammation, while other factors (fibronectin,

alveolar macrophage-derived growth factor) stimulate fibroblasts to proliferate and to make collagen. Since silica particles cannot be digested by the macrophage, the inflammatory process becomes chronic (frustrated phagocytosis). An increased silica burden leads to more foci of inflammation, nodule formation, and fibrosis. The internal process can continue after external exposure ends. Silica particles also enter into alveolar Type I epithelial cells (Churg, 1996), which can lead to cell death of Type I cells and to hypertrophy and proliferation of Type II epithelial cells to replace the Type I cells. The epithelial repair process is associated with a subsequent increase in collagen formation.

The initial diagnosis of silicosis is often based on chest radiographs. Recent papers have used the 1980 classification by the International Labor Organization (ILO, 1980) to identify and classify silicosis into categories and subcategories of seriousness by comparison of patient radiographs with ILO-supplied reference radiographs taken at various stages of silicosis (Table 1):

Table 1. International Labor Organization categorization of silicosis (ILO, 1980).

ILO Category	***Qualitative Description***
0/0	No small (up to 1 cm) silicotic opacities (nodules) are present
0/1	Probably no nodules, but some areas of radiograph are suspect [possible silicosis]
1/0	Small silicotic nodules are most likely present, but not certainly [probable silicosis]
1/1	Small silicotic nodules are definitely present
1/2	Small silicotic nodules are definitely present; other areas of the radiograph may indicate more advanced lesions including large opacities (> 1 cm), pleural thickening, etc.
2/1, 2/2, 2/3, 3/2, 3/3	More advanced stages of silicosis/increasing certainty of the presence of lung abnormalities

Some reports (e.g., Kreiss and Zhen, 1996; Hughes *et al*., 1998) use 1/0 (probable) as the basis of classification of silicosis, since many cases of silicosis are not detected by chest radiographs, yet silicotic nodules and other lesions are found at autopsy (Craighead and Vallayathan, 1980; Hnizdo *et al.*, 1993). Other reports (e.g., Hnizdo and Sluis-Cremer, 1993) use the definite 1/1 as the lowest category indicating silicosis. Some disease is missed by radiography and is determined only by autopsy (Hnizdo *et al*., 1993). The ILO criteria are intended as an epidemiologic classification and comparison tool, not as a diagnostic classification on an individual basis. In occupational medicine practice, a group of tests is used to clinically diagnose silica-related lung disease including physical examination, X-rays, and high resolution computed tomography (CT) scans of the lung (e.g., Begin *et al*., 1991; Olivetti *et al*., 1993).

A. Environmental silicosis

Several studies have reported "environmental silicosis", cases where the silicosis occurs in the absence of an industry usually associated with the disease (reviewed by USEPA, 1996). In one of the stronger examples, Saiyed *et al*. (1991) investigated non-occupational pneumoconiosis in Ladakh, India, high in the western Himalayas where there are no mines or industries. Among

449 randomly selected inhabitants of three villages, there were many cases of pneumoconiosis associated with progressive massive fibrosis (nodules > 1 cm) and "egg shell" calcification of hilar glands. The prevalence of pneumoconiosis was 2.0% (3/150) in the village of Saboo, 20.1% (31/149) in Shey, and 45.3% (68/150) in Chushot, and corresponded with the severity of dust storms and the presence or absence of chimneys in the kitchens (i.e., ventilated cooking). Without chimneys (Chushot), dust concentrations in kitchens averaged 7.5 mg/m^3 during cooking periods. The free silica content of the dust storms was 60-70%. The authors suggested that exposure to free silica from dust storms and to soot from cooking with domestic fuels caused the pneumoconiosis. Perhaps the interaction of silica and soot led to the disease. Such exposures in this and other studies, such as Bar-Ziv and Goldberg (1974), might be considered to be non-industrial but occupational, since the subjects studied by Saiyed *et al*. (1991) were involved in the domestic work of cleaning and cooking (USEPA, 1996). In any case, the exposures were very high and thus similar to some occupational exposures.

B. Occupational silicosis

Several relatively recent reports have presented data that allow a quantitative relationship between occupational dust exposure and the development of silicosis in workers to be calculated.

Hard rock miners in Ontario, Canada (Muir *et al*., 1989)

Muir *et al*. (1989) examined the relationship between cumulative exposure to silica (free crystalline silica, specifically alpha-quartz) and the development of silicosis in 2109 male hard rock (uranium, gold, mixed metals) miners in Ontario, Canada. The miners began work between 1940 and 1959 and were followed either until they ended their dust exposure or until December 31, 1982 (whichever came first). Five X-ray readers examined chest radiographs; one or more readers identified 32 cases of silicosis, defined as ILO category 1/1 or greater with round opacities. All five readers agreed on only six cases, while 12 cases were identified by only one reader (Table 2). A Weibull model of the form

$$R(x) = 1 - \exp[-(\alpha x)^{\beta}] \quad (x \geq 0, \beta > 0)$$

gave the best fit to the data for cumulative risk R of silicosis as a function of cumulative exposure in units of (mg/m^3)-yr. In this model x is the cumulative exposure (lagged five years), α is the Weibull scale parameter, and β is the Weibull shape parameter (Table 2). Estimates of α and β for each reader are given in Table II of Muir *et al*. (1989).

Table 2. Silicosis Risk *vs*. Cumulative Respirable Silica in (mg/m^3)-y (Table IV of Muir et al.)

Reader	***Cases (n)***	***1% risk*** [a]	***2% risk***	***5% risk***	***10% risk***
1	14	3.5 (2.4-5.1)	5.7 (3.9-8.4)	11.2 (6.8-18.2)	18.6 (9.9-35.0)
2	24	2.7 (2.0-3.6)	4.1 (3.2-5.3)	7.1 (5.5-9.1)	10.9 (8.1-14.8)
3	24	3.0 (2.3-3.9)	4.3 (3.4-5.3)	6.9 (5.6-8.5)	9.9 (7.8-12.7)
4	14	3.7 (2.6-5.2)	5.6 (4.1-7.7)	9.8 (6.7-14.3)	15.1 (9.3-24.4)
5	7	5.7 (4.0-8.0)	7.8 (5.5-11.0)	11.9 (7.8-18.3)	16.5 (9.7-28.2)
Any reader	32	2.1 (1.6-2.9)	3.3 (2.6-4.2)	6.0 (4.8-7.5)	9.6 (7.3-12.5)
At least 3	15	3.5 (2.5-4.9)	5.4 (4.0-7.3)	9.5 (6.6-13.6)	14.6 (9.3-23.2)
All readers	6	6.1 (4.1-8.9)	8.5 (5.6-12.8)	13.2 (7.8-22.5)	18.7 (9.7-36.1)

[a] In parentheses is the 95% confidence interval (CI) for each risk estimate.

The Ontario cohort gives the shallowest dose-response relationship for silicosis of the several cohorts examined (see Summary Table 15 below) due in part to the lack of follow-up of members who left the mines (either for another type of work or for retirement). Silicosis often develops after leaving employment (Hnizdo and Sluis-Cremer, 1993; Chen *et al*., 2001). In Hnizdo and Sluis-Cremer (1993), for more than half the cases of silicosis radiographic signs developed at an average of 7.4 years after mining exposure ended. In addition, some of the Ontario miners in the Muir *et al*. study may have changed to a less dusty job if their physician told them that their (annual) radiograph showed abnormalities. The lack of follow-up, leading to under-ascertainment of silicosis, is a serious limitation of this study.

Gray iron foundry workers (Rosenman *et al*., 1996)

Rosenman *et al*. (1996) evaluated 1,072 (96.8% males) current and retired workers in a Mid-western gray iron foundry, which produces engine blocks for the automotive industry. Medical records and silica exposure data were analyzed for those with at least 5 years of employment as of June 1991. Nearly half had worked at the foundry for 20 years. Sixty had radiographic evidence of pneumoconiosis (ILO categories 1/0 and greater). Twenty-eight workers had radiographs consistent with silicosis; of these 25 had simple silicosis and three had progressive massive fibrosis. The prevalence of radiographic changes consistent with silicosis increased with years at the foundry, work area, quantitative silica exposure, and cigarette smoking. In regard to quantitative silica exposure, the authors stated that 0.3-2.7% of workers at the OSHA standard (90-100 $\mu g/m^3$) were silicotic, as were 4.9-9.9% of workers above 100 $\mu g/m^3$. After controlling for confounders, Rosenman *et al*. (1996) used a logistic regression analysis based on cumulative silica exposure to determine an odds ratio of 1.45 for developing a radiograph consistent with silicosis after 20 years of work at 100 $\mu g/m^3$ and an odds ratio of 2.10 after 40 years of work at 100 $\mu g/m^3$ (Tables 3 and 4). This study probably underestimates risk due to lack of follow-up of the current workers. Although silica is not the only toxic chemical in a foundry, the unique nature of the silicotic nodule diminishes the likelihood of confounding by other exposures.

Table 3. Silicosis risk based on Rosenman *et al*. data (Finkelstein, 2000)

Cumulative silica exposure	***Prevalence of silicosis***
< 2 (mg/m^3)-y	0.4%
2-6 (mg/m^3)-y	2.7%
> 6 (mg/m^3)-y	10%

Table 4. Odds ratios for silicosis (from Table 8 of Rosenman *et al.*)[a]

Time-weighted average silica exposure (mg/m^3)	*20-year cumulative exposure [(mg/m^3)-y]*	*Odds ratio (95% C.I.)*	*40-year cumulative exposure [(mg/m^3)-y]*	*Odds ratio (95% C.I.)*
0.010	0.2	1.04 (1.02-1.15)	0.4	1.08 (1.05-1.11)
0.025	0.5	1.10 (1.06-1.14)	1.0	1.20 (1.12-1.30)
0.050	1.0	1.20 (1.12-1.30)	2.0	1.45 (1.25-1.68)
0.075	1.5	1.32 (1.18-1.47)	3.0	1.74 (1.40-2.17)
0.100	2.0	1.45 (1.25-1.68)	4.0	2.10 (1.15-2.82)
0.150	3.0	1.74 (1.40-2.17)	6.0	3.04 (1.96-4.72)
0.200	4.0	2.10 (1.56-2.82)	8.0	4.40 (2.45-7.93)
0.300	6.0	3.04 (1.96-4.72)	12.0	9.24 (3.83-22.3)

[a] Additional mean silica exposures, their calculated odds ratios, and 95% confidence intervals (C.I.) are given in the paper.

Diatomaceous earth workers in California (Hughes *et al.*, 1998; Park *et al.*, 2002)

Hughes *et al.* (1998) investigated 1,809 Caucasian male diatomaceous earth workers in Lompoc, California, who had at least one year of exposure to cristobalite between 1942 and 1987. The crystalline silica isomorph cristobalite is formed when the amorphous silica in diatomaceous earth is calcined at 1000-1100 ºC. Quantitative estimates of dust exposure were made and published in the peer-reviewed literature by Seixas *et al*. (1997) based on 6395 air sampling records taken from 1948-1988. The average estimated respirable dust concentrations for 135 jobs were 3.55 ± 1.25 mg/m^3 prior to 1949, 1.37 ± 0.48 mg/m^3 from 1949-1953, 0.47 ± 0.16 mg/m^3 from 1954-1973, and 0.29 ± 0.10 mg/m^3 from 1974-1988. The workers had periodic chest radiographs. Based on the median of radiographic readings by three independent readers, 81 workers (4.5%) were judged to have opacities on chest radiographs (small opacities, ILO profusion ≥ 1/0, and/or large opacities). Age-adjusted relative risk of opacities increased significantly with cumulative exposure to crystalline silica. The concentration of respirable crystalline silica was an important determinant of risk after accounting for cumulative exposure. The workers were split into two categories: those exposed to < 0.50 mg/m^3 (or hired after 1950) and those exposed to > 0.50 mg/m^3 (or hired before 1950). The risk of opacities for a cumulative exposure to crystalline silica of 2.0 mg/m^3-yr is shown in Table 5.

Table 5. Silica exposure and silicosis based on data of Hughes *et al*. (1998)

Average crystalline silica exposure	*Cumulative risk of silicotic opacities*
< 0.50 mg/m^3 (or hired after 1950)	1.1%
> 0.50 mg/m^3 (or hired before 1950)	3.7%

The findings of Hughes *et al.* (1998) indicate an exposure-response relationship between cumulative exposure to crystalline silica as cristobalite and radiographic opacities. The relationship was substantially steeper among those exposed at the highest average concentrations of crystalline silica. The authors believe that the data do not support the regulatory assumption that cristobalite is more fibrogenic than quartz (i.e., prior to 2000 the occupational limit for cristobalite was half that for quartz), since at average silica levels comparable to other

epidemiologic studies quartz gave a higher incidence of silicosis than did cristobalite in this study. However, since radiography can under-diagnose silicosis, complete accounting for silicosis will require evaluation at autopsy. The ACGIH recently lowered the TLV for alpha-quartz from 100 to 50 $\mu g/m^3$, so that it has the same TLV as cristobalite (ACGIH, 2000).

Park *et al*. (2002) carried out a quantitative risk assessment, by Poisson regression methods, of the onset of silicosis among the diatomaceous earth workers in Lompoc. A linear relative risk model gave the best fit to the data. They estimated an excess lifetime risk for radiographic silicosis of 68-75 cases per thousand workers exposed to 50 $\mu g/m^3$ silica (cristobalite) for a 45 year work-life, then living to age 85. At 1 $\mu g/m^3$ silica the excess lifetime risk was estimated to be 1.6 cases of lung disease other than cancer per thousand workers exposed (Table 6).

Table 6. Excess lifetime risk of silicosis predicted by Park *et al*. (2002)

Silica concentration (mg/m^3)	***45 year cumulative exposure in mg/m^3-y***	***Radiographic silicosis - all workers***	***Radiographic silicosis in workers with < 10 mg/m^3-y***
0.001	0.045	6.2/1000*	1.6/1000
0.005	0.225	17/1000	7.8/1000
0.01	0.45	26/1000	16/1000
0.02	1.8	39/1000	31/1000
0.05	2.25	68/1000	75/1000
0.1	4.5	100/1000	140/1000
0.2	9	150/1000	260/1000

* Excess risk estimates assume that workers were exposed to a constant silica concentration for up to 45 years (ages 20-65). Annual risks are accumulated up to age 85.

White South African gold miners (Hnizdo and Sluis-Cremer, 1993)

Hnizdo and Sluis-Cremer (1993) investigated silicosis risk retrospectively in a cohort of 2,235 white male South African gold miners. Exposure estimates were made for nine separate occupational categories based on a special study of dust levels in these mines done by Beadle in the 1960s (Beadle, 1971). To compensate for the fact that the average hours working in dust ranged among the 9 categories from 4 hours for "other officials" to 8 hours for "shaft sinkers and developers," exposure was "normalized" to 8-hour shifts. The workers had a minimum of 10 years and an average of 24 years service from 1940 until the early 1970s. Dust levels were fairly constant during this period (see, e.g., Table 2 in Gibbs and DuToit (2002)). The miners had an annual chest radiograph while mining; they were followed until 1991 for radiographic signs of the onset of silicosis. An ILO category 1/1 (definite silicosis) or greater was selected to designate silicosis. Two independent readers initially read the chest films, but only the reader whose interpretations correlated better with autopsy results was used for additional analysis; the use of one reader is a limitation of the study. There were 313 miners (14% of the cohort) who developed radiographic signs of silicosis at an average age of 55.9 years. The latency period was largely independent of the cumulative dust exposure (CDE). In 57% of the silicotics, the radiographic signs developed at an average of 7.4 years after mining exposure ceased. The risk of silicosis determined by chest radiographs increased exponentially with cumulative dust dose. At the highest level of 15-(mg/m^3)-years CDE (approximately 37 years of gold mining at an

average respirable dust concentration of 0.4 mg/m^3), the cumulative risk for silicosis reached 77% as estimated by the accelerated failure time model using the log-logistic distribution (SAS Proc LIFEREG):

$$CR(t) = 1 - \{1/[1 + \exp(-\mu/\sigma) \times t^{(1/\sigma)}]\}$$

where CR(t) = cumulative risk at time t, and μ (2.439) is the intercept and σ (0.2199) is the scale parameter estimated by SAS's LIFEREG procedure. The authors concluded that the risk of silicosis was strongly dose-dependent, but that the latency period was largely independent of dose. The life table analysis (SAS Proc LIFETEST) below (Table 7) shows the number of miners who developed silicosis ("cases"), the number of miners considered by the authors to be at risk, and the risk per unit of CDE (also as calculated by the authors). In the table in column 1 (in parentheses) are OEHHA's determination of the mg/m^3–yr respirable silica exposure, based on Hnizdo and Sluis-Cremer's estimate of 30% silica in the dust, and in column 4 is the total number of miners actually at each midpoint level of CDE or silica. The values in column 4 of Table 7 are the number of workers in the group with the temporally integrated dust exposure in column 1.

Table 7. Life table results - Risk of silicosis per unit Cumulative Dust Exposure (CDE)
(from Table IV of Hnizdo and Sluis-Cremer, 1993)

Midpoint in (mg/m^3)-y of CDE (silica)	***Cases of silicosis***	***Number of workers at risk based on life table***	***Number of workers remaining at this CDE midpoint***	***"Risk/ unit CDE"***	***Mean years in dust***	***Mean dust conc. (mg/m^3)***
1 (0.3)	0	2218	204			
3 (0.9)	9	2014	474	0.002	20.5	0.17
5 (1.5)	48	1540	556	0.016	23.5	0.24
7 (2.1)	85	984	469	0.045	27.2	0.30
9 (2.7)	93	515	318	0.099	28.0	0.33
11 (3.3)	53	197	142	0.156	29.4	0.38
13 (3.9)	20	55	44	0.222	31.5	0.41
15 (4.5)	5	11	11	0.227	37.0	0.42

[a] CDE = Σ number of dusty shifts x mean mass respirable dust conc. x average number of hours spent underground / (270 shifts/year x 8 h/shift)

A plot of risk of silicosis per unit of Cumulative Dust Exposure (CDE) versus the mid-point unit CDE, as given in Figure 1 of the Hnizdo and Sluis-Cremer report, and a plot of % silicosis among the workers actually exposed to a given level of silica (Figure 2), as determined by OEHHA staff, respectively, are given below.

Figure 1. Risk of silicosis per unit CDE vs. CDE mid-point

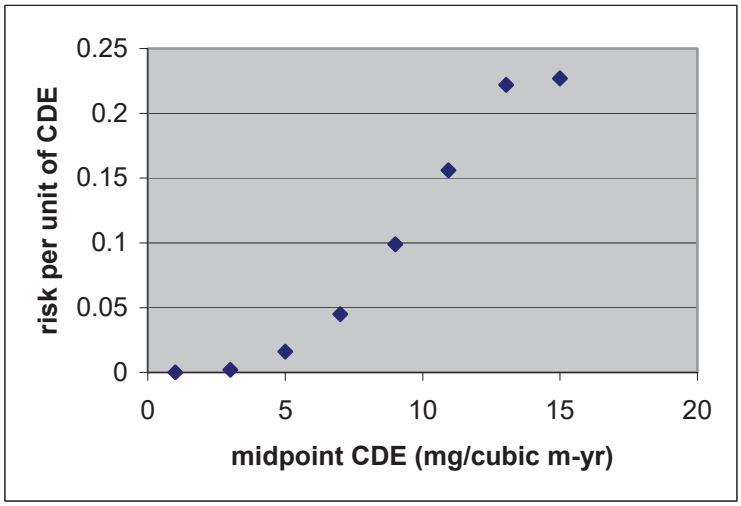
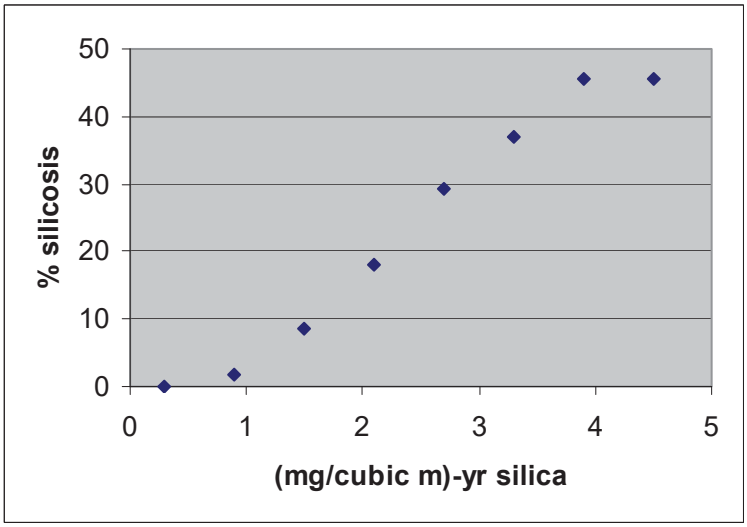


Figure 2. Percent silicosis among workers at each silica level



Black South African gold miners (Churchyard *et al.*, 2004; Murray *et al.*, 1996)

Black migrant contract workers constitute a large majority (85 - 90%) of South African gold miners. In a cross sectional study, Churchyard *et al*. (2004) interviewed and took chest radiographs of 520 black gold miners (mean age = 46.7 years, range = 37.1 – 59.9) who were still mining (average service = 21.8 years, range 6.3-34.5). Two readers examined the radiographs. As in the Hnizdo and Sluis-Cremer study, silicosis was defined as an ILO (1980) profusion of ≥ 1/1. The mean respirable dust concentration was 0.37 mg/m^3 (0 - 0.70); the mean

quartz concentration was 0.053 mg/m^3 (0 - 0.095). The prevalence of silicosis was determined to be 18.3% by one reader and 19.9% by the other (mean 19.1%) (Table 8). This included several workers with more serious silicosis as indicated by ILO profusions ≥ 2/1 (see Table 1). Significant trends were found between the prevalence of silicosis and: (1) length of service (OR = 1.69 per 5 years), (2) mean intensity of exposure (OR = 1.18 per 0.01 mg/m^3), and (3) cumulative exposure to quartz (OR = 3.2). The study confirms the large burden of silicosis among older black workers in this industry (see next paragraph). The burden is likely to worsen with continuous employment in dusty jobs. For this cohort the prevalence of silicosis will increase even if the miners stop mining immediately. If, as assumed by the authors, the dust levels during the working life of these black miners were constant, silicosis developed while they were exposed to a quartz level below the workplace limit of 0.100 mg/m^3.

Table 8. Silicosis in black gold miners (Churchyard *et al*., 2003; 2004)

Cumulative quartz exposure in mg/m^3-yr	***Mid-point of cumulative quartz exposure***	***Number in quintile****	***Cases of silicosis***	***Percent silicosis***
0 – 0.80	0.4	103	11	10.7
0.80 – 0.99	0.9	97	8	8.2
0.99 – 1.24	1.12	103	18	17.5
1.24 – 1.48	1.36	104	23	22.1
1.48 –3.08	2.28	103	33	32.0
(Total)		(510)**	(93)	(18.2)

* Personal communication from Dr. J. teWaterNaude, December 2, 2004.
** Ten of the 520 films were unreadable.

Murray *et al*. (1996) analyzed data from 16,454 black South African gold miners dying from unnatural causes between 1975 and 1991 in order to study change in prevalence in silicosis and pulmonary tuberculosis (TB). TB prevalence increased from 0.9% in 1975 to 3.9% in 1991, while that for silicosis increased from 9.3% to 12.8%. The prevalence of both increased with age and duration of service. Silicosis was the most significant predictor of TB (OR = 1.78, CI = 1.27 - 2.30, p = 0.0001). A highly significant trend for TB, for year of autopsy, remained after adjustment for other variables, such as age and duration of service (OR = 1.04, CI = 1.01 – 1.06, p = 0.0046). (Another 21,202 black gold miners died of natural causes during the study period.)

Hong Kong granite workers (Ng and Chan, 1994)

Ng and Chan (1994) investigated silicosis among 338 male workers, who had worked at least one year between 1967 and 1985 in two granite quarries in Hong Kong. Three readers examined the chest radiographs. Silicosis was defined as an ILO classification of at least 1/1 (for small rounded opacities) or greater, assigned by at least two of the three readers. Exposure was estimated for each worker based on job category and particle counts. Thirty-six workers (10.6%) were designated silicotic. Both a logistic and a linear model fit the data well. The study suffered because only about half of the previously employed granite workers were studied, which

probably led to an underestimate of silicosis risk in at least the highest exposure category and maybe in others. The data are summarized in Table 9.

Table 9. Silica exposure and silicosis in Ng and Chan (Finkelstein, 2000)

Mean cumulative exposure (mg/m^3)-y	***Prevalence of silicosis*** [a]
< 1	0%
3.1	13%
7.1	25%
22	22%

[a] rounded opacities determined by at least 2 of 3 readers (Table 3 of Ng and Chan)

Gold miners in South Dakota (Steenland and Brown, 1995)

Steenland and Brown (1995) studied a very large cohort (3330) of white male gold miners in South Dakota, who had worked at least 1 year underground between 1940 and 1965 (average = 9 years underground). The mine dust contained on average 13% silica (range = 1-48%). A job-exposure matrix was created for full-time underground workers grouped into five categories. The authors estimated that most miners were exposed to a median silica level of 0.05 mg/m^3, but that those hired before 1930 were exposed to a median level of 0.15 mg/m^3. A total of 170 cases of silicosis (5.1% of the cohort) was determined from death certificates only (n = 128 cases), from two cross-sectional radiographic surveys in 1960 and 1976 (n = 29 cases; ILO category 1/1 or greater), or from both (n = 13 cases). Unfortunately, only 25% of living cohort members were surveyed radiographically. The life-time risk of silicosis was less than 1% with a cumulative exposure under 0.5 mg/m^3-years and increased to 68% to 84% for the highest cumulative exposure category (more than 4 (mg/m^3)-years) (Table 10).

Table 10. Risk of silicosis for cohort by cumulative exposure (Table 3, Steenland and Brown)

Silica exposure in (mg/m^3)-yrs: range (midpoint)	***Miners with silicosis***	***Number entering exposure category (from life table)***	***Number remaining at this exposure level***	***Cumulative [a] Risk***	***Mean years of exposure***	***Mean year first exposed***
0-0.2 (0.10)	5	3330	1530	0.002	2.9	1953
0.2-0.5 (0.35)	5	1800	740	0.005	9.7	1948
0.5-1.0 (0.75)	15	1060	376	0.017-0.022[b]	15.4	1942
1.0-2.0 (1.50)	33	684	353	0.060-0.084[b]	13.2	1931
2.0-3.0 (2.50)	44	331	206	0.167-0.245[b]	18.8	1926
3.0-4.0 (3.50)	42	125	73	0.403-0.534[b]	25.5	1921
>4.0	26	52	52	0.678-0.844[b]	30.6	1914

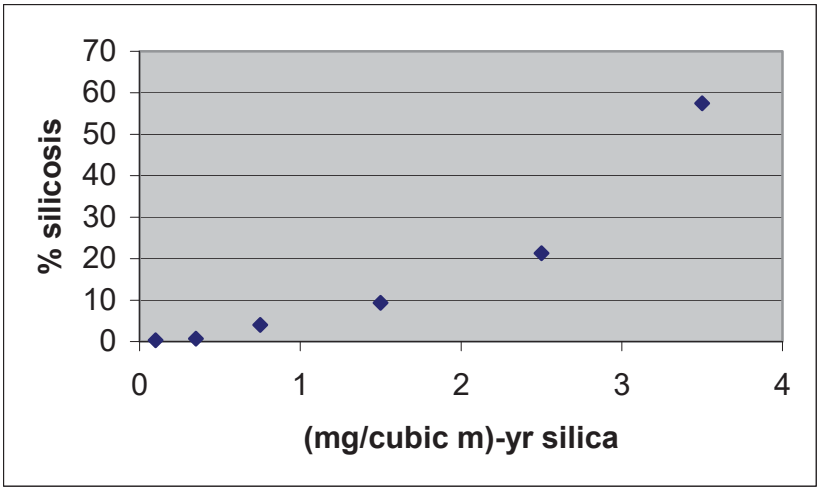
[a] Cumulative risk = 1−exp[−sum of (hazards * interval width)], where the hazards for each category of cumulative exposure are:
no. cases/(width*(no. entering category – 0.5*no. cases – 0.5*no. withdrawals))

[b] Cumulative risk adjusted for age and calendar time (Steenland and Brown, 1995)

The best predictor of disease was cumulative exposure ((mg/m^3) – years), followed by duration of exposure (years), and then by average exposure (mg/m^3). Figure 1 of Steenland and Brown indicates that a plot of their data for silicosis risk versus cumulative silica exposure was similar to a plot of the data of Hnizdo and Sluis-Cremer (1993). After adjustment for competing risks of death, Steenland and Brown estimate that a 45-year exposure to 90 - 100 μg/m^3 silica would lead to a lifetime risk of silicosis for gold miners of 35% to 47%. A limitation of this study is the reliance on death certificates rather than on ILO interpretation of radiographs. In addition, no mention was made of validating the data on the death certificates. It was also not clear what, if any, autopsy data were available. A plot of silicosis incidence among the workers (as determined by OEHHA staff) actually exposed to the estimated level of silica is given in Figure 3 below. An accompanying editorial (Wagner, 1995) commended the article for estimating both

the risk of silicosis while working and the lifetime risk of silicosis resulting from exposure during work.

Figure 3. % Silicosis vs. silica exposure in Steenland and Brown (see Table 10)



Miners in Leadville, Colorado (Kreiss and Zhen, 1996)

Kreiss and Zhen (1996) investigated the exposure-response relationships for silicosis among 134 male miners over 40 years old in Leadville, Colorado. The men had been studied three years earlier in a random sample of respiratory disease in their community (Kreiss *et al*., 1989). Of 100 dust-exposed miners, 32 had radiological profusions of small opacities of ILO category 1/0 or greater at a mean of 36.1 years since their first silica exposure. Of miners with cumulative silica, exposures of 2 (mg/m^3)-years or less, 20% had silicosis while 63% of miners accumulating greater than 2 (mg/m^3)-years had silicosis. Average silica exposure was also strongly associated with silicosis prevalence rates (Table 11).

Table 11. Miners studied by Kreiss and Zhen (1996)

Average silica exposure	***% silicotics***
0.025-0.05 mg/m^3	13% (5/38)
> 0.05-0.1 mg/m^3	34% (15/44)
> 0.1 mg/m^3	75% (9/12)
Cumulative silica exposure	*% silicotics*
≤ 2 (mg/m^3)-y	20% (14/70)
2 – 4 (mg/m^3)-y	63% (15/24)

Based on logistic regression models of the form $R(x) = [1 + \exp(-\alpha - B'x]^{-1}$, Kreiss and Zhen concluded that the risk of silicosis was best predicted by elapsed time since last silica exposure

together with either (1) cumulative silica exposure or (2) a combination of average silica exposure and duration of exposure. Exposure-response relationships were substantially higher using measured silica exposures (compared to using estimated silica exposures based on measured total dust exposures and assuming a constant silica proportion of dust). The risk of silicosis in this study is higher than in workforce studies having no follow-up of those leaving the mining industry (e.g., Muir *et al*., 1989) and in studies without job title-specific silica measurements (e.g., Hnizdo and Sluis-Cremer, 1993). However, the risk is comparable to several recent studies of exposure-response relationships for mining dust (e.g., Ng and Chan, 1994; Steenland and Brown, 1995) (see Summary Table 15 below). A limitation relative to other studies is the small number of subjects (100) in the group.

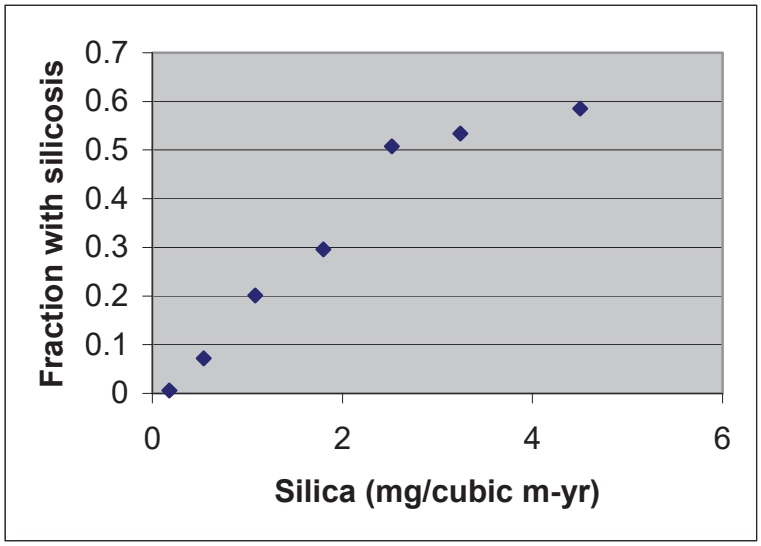
Chinese tin miners (Chen *et al*., 2001)

Chen *et al*. (2001) found a clear exposure-response relationship between silica dust exposure and silicosis in a cohort of 3010 (2795 male and 215 female) miners employed for at least 1 year during the period 1960-1965 in any of four Chinese tin mines. No other diseases due to silica or tin were mentioned. Each cohort member was followed through 1994. Historical Chinese total dust (CTD) data were used to create a job exposure matrix for each facility, job title, and calendar year. The CTD data were converted to estimates of respirable crystalline silica for comparison with findings from other epidemiological studies of silicosis (including some of those above). Each miner's work history was abstracted from employment records. The diagnosis of silicosis was based on 1986 Chinese Roentgen diagnostic criteria for pneumoconiosis. The criteria classified silicosis as stages I-III, similar to an ILO classification of 1/1 or greater. Of the 3010 miners, 1015 (33.7%) were identified as silicotic (mean age = 48.3 years, with a mean of 21.3 years after first exposure) (Table 12). Among the silicotics, 684 (67.4%) developed silicosis after their tin mine exposure had ended (mean = 3.7 years after). The risk of silicosis was strongly related to cumulative exposure to silica. The Weibull distribution gave a very good fit to the data. The risk of silicosis was less than 0.1% when CTD was less than 10 (mg/m^3)-yr (= 0.36 (mg/m^3)-yr of respirable crystalline silica). The risk of silicosis increased to 68.7% when CTD exposure was equal to 150 (mg/m^3)-yr (= 5.4 (mg/m^3)-yr of respirable crystalline silica). Latency period was not correlated to the risk of silicosis or to cumulative dose. From their data, the authors predicted a 55% risk of silicosis for 45 years exposure to 0.1 mg/m^3 respirable crystalline silica, the workplace exposure limit (4.5-(mg/m^3)-years silica). Figure 4 plots the fraction of the workers in Chen *et al*. with silicosis (column 2 in Table 12 divided by column 4) exposed to a given level of silica (mid-point – in parentheses in column 1 of Table 12), as calculated by OEHHA staff.

Table 12. Cumulative silicosis risk based on cumulative total dust (CTD) (Table 5, Chen *et al*., 2001)

Range of CTD exposure in (mg/m^3)-y/ (silica mid-point)	*Cases of silicosis (n)*	*Workers entering category*	*Workers at this level of CTD/silica*	*Cumulative risk based on Weibull model*	*Mean net exposure (years)*	*Mean latency (years)*
<10 (0.18)	2	3010	333	0.001	2.2	14.7
10-19.99 (0.54)	24	2677	334	0.010	5.3	21.3
20-39.99 (1.08)	126	2343	626	0.070	9.3	22.0
40-59.99 (1.80)	127	1717	429	0.145	11.9	21.5
60-79.99 (2.52)	196	1288	386	0.285	9.9	20.3
80-99.99 (3.24)	141	902	264	0.405	10.8	19.0
100-149.99 (4.50)	244	638	417	0.663	13.1	20.4
≥ 150 (≥ 5.4)	155	221	221	0.917	15.7	25.4

Figure 4. Percent silicosis vs. silica level from Chen *et al*.



Industrial sand workers (McDonald *et al*., 2001; Hughes *et al*., 2001; Rando *et al*., 2001)

McDonald *et al*. (2001) studied a cohort of 2670 men employed before 1980 for 3 years or more and followed through 1994 in one of nine North American sand-producing plants and in a large associated office complex (since most of the office employees had previously worked in the mines). They found 37 deaths due to silicosis and silicotuberculosis. The mean exposure of the cohort was 42 μg/m^3 silica (Rando *et al*., 2001). Odds ratios for silicosis mortality, determined using conditional multiple logistic regression (SAS software), were significantly related to cumulative silica exposure (Hughes *et al*., 2001) (Table 13). The odds ratios are in general agreement with those in the gray foundry workers of Rosenman *et al*. (1996) (Table 4).

Table 13. Median cumulative silica exposure and odds ratio (Table 3 in Hughes *et al*., 2001)

No lagging			*Lagged 15 yr*		
Median exposure in (mg/m^3)-y	***Silicotics (n)***	***Odds ratio [a] for mortality***	***Median exposure in (mg/m^3)-y***	***Silicotics (n)***	***Odds ratio [a,b] for mortality***
0.832	7	1.00	0.142	7	1.00
2.744	7	1.27	1.229	7	2.54
6.916	8	2.62	2.583	7	4.55
12.084	7	2.13	7.990	8	5.16

[a] Matched odds ratio relative to lowest cumulative exposure category. Although labeled a cohort study, the data analysis compared cases of silicosis with non-silicotic controls.
[b] Significant increasing trend across exposure categories (see Hughes *et al*. for more details)

Ceramic workers (Cavariani *et al*., 1995; Legrand-Cattan *et al*. (1998)

Cavariani *et al*. (1995) investigated the incidence of silicosis among 2,480 men in the ceramics industry in central Italy. The workers were surveyed during the period 1974-1987 and followed through 1991 with annual chest radiographs. The cumulative risk of silicosis (ILO category 1/1 or greater) was 48% after 30 years of employment. A multivariate Cox's proportional hazards model indicated that silicosis increased linearly up to the period of 25-29 years employment. A hazard risk of 14.6 was found comparing those with ≥ 30 years exposure to those employed 10 years. Smoking significantly contributed to the model, but its role was unclear.

Legrand-Cattan *et al*. (1998) examined the dose-response relationship in two French ceramic plants. A 1992 cross-sectional study included more than 200 silica-exposed workers. Three ILO certified B readers read chest radiographs. Silica was sampled in the airborne dust. The results are tabulated below (Table 14).

Table 14. Silicosis in two French ceramic plants (Legrand-Cattan *et al*., 1998)

Cumulative exposure to silica in (mg/m^3 – years)	***Number of workers at this level***	***Number with small opacities with ILO profusion ≥ 1/0***	***Percent***
< 0.35	50	2	4
0.35 – 1.08	57	8	14
1.09 – 1.77	55	11	20
> 1.77	55	17	31
Total	217	38	(18)

A dose response relationship is clear; the authors reported a p value of 0.002. However, the study is limited by the lack of follow-up of the workers.

Slate workers (Glover *et al*., 1980; Saiyed *et al*., 1985; Saiyed and Bannerjee, 1985)

Slate contains calcium carbonate, iron oxides, silicates, amorphous silica, and crystalline silica. Glover *et al*. (1980) studied slate workers in North Wales. The respirable slate dust contained 13-32% crystalline silica. In the study group were 725 current and former workers exposed only to slate dust, while the controls were 530 men from the same area who had never been exposed to dust. Pneumoconiosis was found in 239 slate workers (33 %), and 10% had degrees of pneumoconiosis (category 2 or higher using the 1971 ILO scheme) that would bring worker's compensation. The prevalence of respiratory symptoms (cough, phlegm, dyspnea) was high. There was evidence of an effect of both simple and complicated pneumoconiosis on lung function (declines in FVC and FEV_1) additional to the effect of age. The high prevalence (40-50%) of radiological lesions suggested the presence of healed tubercular lesions in men over 55. Either pneumoconiosis or old tubercular lesions (or both) could account for the symptomatology and disability of the men.

Saiyed *et al*. (1985) surveyed the slate-pencil industry in India. An industrial hygiene survey revealed very high levels of free silica (2-10 mg/m^3), while a medical survey showed that 324 of 593 workers (54.6%) had silicosis. Of these, 105 had "conglomerate" silicosis (progressive massive fibrosis, PMF). Some lung lesions were detectable after less than five years of exposure to slate dust. Saiyed and Bannerjee (1985) conducted a follow-up examination 16 months later. The progression of silicosis was very rapid, and a total of 23 workers had died during this period (mean age = 34.7 years; mean exposure = 11.9 years). The authors attributed the high mortality to high levels of silica leading to early onset of PMF. The progression of silicosis was related to the intensity and duration of dust exposure, and to the severity of silicosis found initially.

Silicosis has been reported in other groups of slate workers in Norway (Bang and Suhr, 1998; Suhr *et al*., 2003) and in Germany (Mehnert *et al*., 1990).

Silica particle size

Data on silica particle size in the various workplaces are limited. According to Witschi and Last (2001), silica particles with a diameter of 1 μm (range = 0.5 - 3 μm) appear to be the most fibrotic in humans. NIOSH (1974) reviewed the existing literature and found that in five diatomite plants the mean silica diameter was 1.1 μm (range = 0.5 - 2 μm). For nine potteries, the particle size was 1.2 μm. For 18 foundries, more than 90% of the particles were less than 3 μm. The majority of particles to which shipyard sandblasters were exposed was also less than 3 μm. In the Vermont granite sheds, 10 mppcf (million particles per cubic foot) granite dust were initially estimated to be equal to 0.1 mg/m^3 respirable quartz. Steenland and Brown (1995) used this estimate for silica in South Dakota gold mines. Assuming that the density of quartz is 2.65 g/cm^3 and that the quartz particles are spherical, the data indicate that the particles have a diameter of 0.59 μm. NIOSH (1974) listed 0.94 μm as the median particle size in metal mines. No indication was given of the dispersion of the particle sizes around the average value. Davis *et al*. (1983) used the value of 10 mppcf in granite sheds as equal to 0.075 mg/m^3 silica. For that estimation, OEHHA staff calculated the particle diameter to be 0.53 μm. Thus, existing data indicate that the majority of silica in the workplace is respirable. In most of the occupational studies examined, the exposures were measured using a calibrated cyclone sampler similar to that recommended in the current NIOSH (2003) method. This allows collection of particles

primarily in the 0.5 – 5 μm range, with a collection efficiency profile intended to match the penetration of particles into the alveolar region of the human lung. In the case of the South African gold mine studies (Beadle, 1971; Page-Shipp and Harris, 1972; Hnizdo and Sluis-Cremer, 1993), particle number was determined by an optical method selecting respirable particles (range of 0.5 to 5 μm). Thus, the risk estimates obtained from these studies refer to particles in the size range where penetration occurs into the respiratory region of the lung. This corresponds to the size range of particles thought to be responsible for silicosis. It differs from the definition of "respirable" particles (*i.e.* PM_{10}) commonly used in environmental measurements, which refers to particles capable of penetrating anywhere in the lower respiratory tract (described as "thoracic" particles in occupational studies).

Risk estimation for silicosis from epidemiologic studies

The data from the above studies have been used by a number of investigators (Finkelstein, 2000; Chen *et al*., 2001; Hughes, 1995) and by OEHHA staff to estimate percent silicosis based on cumulative silica exposure in units of (mg/m^3)-yr. The results are summarized in Table 15.

Table 15. Summary - Estimates of % silicosis based on cumulative silica exposure in (mg/m^3)-y

Study	*Population (number with silicosis)*	*Exposure of 2 (mg/m^3)-y*	*Exposure of 4 (mg/m^3)-y*	*Exposure of 4.5 (mg/m^3)-y*
Muir *et al*., 1989	2109 male Ontario hard rock miners ("15")	0.4[a,c]	1.2[a,c]	2[b]
Rosenman *et al*., 1996	1072 Midwestern foundry workers (28)	2[a]	10[a]	3[b]
Graham *et al*., 1991	408 Vermont granite workers (35)	~3[c]	—	—
Hughes *et al*., 1998	1809 white male diatomaceous earth workers (81)	1.1 (low intensity) 3.7 (high intens.)[a]	4 (low) 12 (high)[a]	—
Park *et al*., 2002	2342 white male diatomaceous earth workers (80)	~7[e]	13[e]	14[e]
Hnizdo & Sluis-Cremer 1993	2235 white male South African gold miners (313)	5[a] 10[c]	52[a] 60[c]	77[b]
Ng & Chan, 1994	338 male Hong Kong granite workers (36)	6[a]	15[a]	15-20[b]
Steenland & Brown, 1995	3330 male S. Dakota gold miners (170)	8[a]	53[a]	70[b]
Kreiss & Zhen, 1996	100 miners in Leadville, CO (32)	11[a]	53[a]	92[b]
Chen *et al*., 2001	3010 Chinese tin miners (1015)	14[d]	47[d]	55[b]
Churchyard *et al*., 2004	510 black gold miners (93)	~28	-	-

[a] From Table II of Finkelstein (2000)
[b] From Table 6 of Chen *et al*. (2001)
[c] From Tables 3 and 4 of Hughes (1995)
[d] Interpolated by OEHHA staff from Fig. 2 of Chen *et al*. (2001).
[e] Estimated by OEHHA staff from Table 4 of Park *et al*. (2002)
[f] 158 had an ILO reading ≥ 1/0, while 103 had an ILO reading ≥ 1/1.

In Table 15, more than 14,000 workers were studied, of whom approximately 12% were classified as silicotic. The 12% is likely an underestimate of the incidence of silicosis due to lack of follow-up by chest radiographs during life in some cohorts and to the lack of an autopsy after death.

Determination of LOAEL and NOAEL for silicosis (Rice and Stayner, 1995)

In another approach to the data, Rice and Stayner (1995) identified the NOAEL and LOAEL for silicosis in several studies (Table 16). The study of Hnizdo and Sluis-Cremer (1993) yielded both a LOAEL and a NOAEL.

Table 16. Estimates of NOAELs and LOAELs for silicosis (Rice and Stayner, 1995)

Study	***Subjects***	***NOAEL in μg/m³***	***LOAEL in μg/m³***
Davis *et al*., 1983	969 granite workers	67.5	
Hnizdo and Sluis-Cremer, 1993	2235 gold miners	7	20
McDonald and Oakes, 1984	1321 gold miners	-	8[a]
	64 gypsum miners	35	49
Muir *et al*., 1989	2109 gold miners	Could not determine	Could not determine
Rice *et al*., 1986	888 dusty trade workers	80-100	200-252

[a] McDonald and Oakes (1984) considered this value to be only an approximation.

Proposals to change the occupational exposure limit

Silicosis is still being diagnosed at death in workers who were supposed to be exposed to occupational levels of 50-100 μg/m^3. Thus, there have been recommendations that the occupational exposure limit for respirable, crystalline silica (specifically alpha-quartz) be lowered from the current level of 100 μg/m^3 to 50 μg/m^3 (NIOSH, 1974; Rosenman *et al*., 1996; ACGIH, 1999; Finkelstein, 2000). In 2000, the ACGIH lowered its TLV for quartz from 100 to 50 μg/m^3. In 1986, WHO recommended that the occupational level be set at 40 μg/m^3 (WHO, 1986). Greaves (2000) recommended that the TLV be lowered to 10 μg/m^3. Based on existing data Greaves (2000) estimated that at 10 μg/m^3 the incidence rate for ILO grade 1/0 silicosis would be less than 5%, while for grade 1/1 it would be less than 2%. Chen *et al*. (2001) recommended that the TLV be lowered to 5 μg/m^3. "If the lifetime risk of silicosis is to be under 1 in 1000 (a criterion used by OSHA) for a lifetime exposure of 45 years, then the mean Chinese total dust concentration must be lower than 0.14 mg/m^3 (or lower than 0.005 mg/m^3 respirable crystalline silica)" (Chen *et al*., 2001). Mannetje *et al*. (2002) pooled data from six occupational cohorts. These included four groups discussed above: diatomaceous earth workers, Vermont granite workers, U.S. industrial sand workers, and South Dakota gold miners. Among them 170 deaths from silicosis were reported. The estimated mortality risk from silicosis to age 65 after 45 years of exposure at 100 μg/m^3 silica was 13 per 1000, while the risk of death at 50 μg/m^3 was estimated at 6 per 1000. Both estimates are above the 1 per 1000 risk acceptable to OSHA. Mannetje *et al*. also concluded that the occupational standards for silica should be lowered, but they did not specify a level. They further state that their estimates of silicosis mortality are probably underestimates due to exposure misclassification and to outcome misclassification, since deaths due to silicosis might have been coded to tuberculosis or chronic obstructive pulmonary disease.

C. Silica exposure and lung cancer in workers

In 1997, IARC classified respirable crystalline silica in Class 1, a Known Human Carcinogen, based on occupational epidemiologic studies. However, chronic RELs are not based on cancer endpoints. Further, there is no approved cancer potency factor for silica.

V. Effects of Animal Exposures

Several papers have reported that freshly fractured quartz, which has increased surface activity, causes greater inflammation than "aged" quartz. Vallyathan *et al*. (1991) reported that "fresh" silica was 4.2-fold more potent than silica aged for 1-2 days in decreasing the membrane integrity of male rat macrophages; 50% more potent in activating hydrogen peroxide secretion by macrophages; and 4.6-fold more potent in stimulating cellular chemiluminescence. Vallyathan *et al*. (1995) reported that inhalation of 19.3 mg/m^3 aged (for 2 months) quartz for five hours/day for 10 days by male Fischer 344 rats increased the number of cells recoverable by bronchoalveolar lavage (BAL) (Table 17). Aged quartz also gave histopathological evidence of increased pulmonary infiltrates, showed higher levels of biochemical markers of lung injury, increased lipid peroxidation, and increased the ability of pulmonary phagocytes to produce more oxygen radicals than air-exposed controls. These pulmonary responses were significantly more pronounced after inhalation of 22.4 mg/m^3 freshly fractured quartz.

Table 17. Cells recovered in bronchoalveolar lavage from rats (Vallyathan *et al*., 1995)

Cell type	***Room air***	***Aged quartz***	***Freshly fractured***	***Fresh/aged***
Total cells	7.1±0.78*	9.3±1.2	20.4±2.2	2.2
Macrophages	6.7±0.69	4.7±0.79	5.4±0.78	1.1
Neutrophils	≥ 0.038	5.3±0.66	10.4±1.44	2.0
Lymphocytes	≥ 0.038	1.7±0.25	3.6±0.27	2.1
Red blood cells	≥ 0.038	1.7±0.26	6.0±0.57	3.5

* Cell counts are in millions. Each value is the mean ± standard error of 5 rats.

Burns *et al.* (1980) exposed female Balb/c mice for up to 39 weeks to 4.9 mg/m^3 Min-U-Sil brand crystalline silica. By 24 weeks, silica-laden macrophages were present in the lungs. After 39 weeks of exposure, silicotic lesions were seen in the lungs and adjacent lymph nodes (Table 18).

Davis *et al*. (1998) exposed mice to an aerosol of cristobalite silica (mass median aerodynamic diameter (MMAD) = 1.7 μm) for five hours/day in order to examine (1) the effects of exposure dose, (2) the evolution of disease over time, and (3) the variation in responses among strains. In C3H/HeN mice, incremental, cumulative exposure doses of cristobalite (10 mg/m^3 for 8 days, 43 mg/m^3 for 9 days, and 70 mg/m^3 for 12 days) caused (1) increased initial lung dust burden at 12 to 16 weeks post-exposure, (2) progressively intense pathological responses, and (3) increased total lung collagen (as measured by hydroxyproline).

The histopathological changes and total lung collagen increased with time after exposure. Silicosis was compared in four inbred strains of mice (BALB/c, C3H/HeN, MRL/MpJ, New

Zealand Black) 16 weeks after aerosol inhalation exposure to cristobalite (70 mg/m^3, 5 hours/day, 12 days). C3H/HeN mice had histopathological silicotic lesions, enlarged intrapulmonary lymphoid tissue, and increased lung wet weight, increased bronchoalveolar lavage (BAL) recoverable macrophages, lymphocytes, and neutrophils, and increased total lung collagen (hydroxyproline analyses). BALB/c mice developed slight pulmonary lesions. MRL/MpJ mice showed prominent pulmonary infiltrates with lymphocytes. New Zealand Black (NZB) mice developed extensive alveolar proteinaceous deposits, inflammation, and fibrosis. The authors found both dose-time-response relationships and a substantial variation of responses among mouse strains to the high level, short duration exposure.

At Brookhaven National Laboratory, groups of Fischer 344 rats were exposed to 0, 2, 10, and 20 mg/m^3 Min-U-Sil brand silica (alpha-quartz) for six months (Kutzman, 1984a; as summarized by USEPA, 1996). Other groups of rats had the same exposure, but were allowed to "recover" in air for an additional 6 months (Kutzman, 1984b; as summarized by USEPA, 1996). Significant alterations in total lung weight, total lung collagen, total elastin per unit lung dry weight, and total protein per unit lung dry weight at 2 mg/m^3 silica and microscopic evidence of silicotic lesions at the higher silica levels indicated that 2 mg/m^3 was a LOAEL for silica effects. After six months in clean air, the silica-induced lesions appeared to worsen.

Muhle *et al*. (1989) exposed groups of 50 male and 50 female rats to 1 mg/m^3 DQ12 quartz six hours/day, five days/week for 24 months. DQ12 contains 87% crystalline alpha-quartz, has a mass median aerodynamic diameter (MMAD) of 1.3 μm, and is 74% respirable. Moderate fibrosis was seen in 85 animals, slight fibrosis in 13, and very slight fibrosis in 1 (total rats with fibrosis = 99/100). Varying amounts of peribronchial granulomatous foci were noted in 95 rats.

Muhle *et al*. (1998) reported lung fibrosis in hamsters exposed to 3 mg/m^3 DQ12 silica. After 18 months of exposure to DQ12 for 6 h/day, 5 days/week, all hamsters in the group of 15-19 animals necropsied had very slight fibrosis. Approximately 100 silica-exposed animals were exposed for five more months to air only. Afterward 22.2% had very slight fibrosis, 68.7 % had slight fibrosis, and 1% had moderate fibrosis (i.e., more than 90/100 hamsters had lung fibrosis). No collagen measurements were reported. Thus, rats, mice, and hamsters show pulmonary fibrosis after crystalline silica exposure at and above 1 mg/m^3.

Wagner *et al*. (1968) exposed dogs up to 2.5 years, guinea pigs up to 18 months, and rats up to 2 years for 6 hours/day, 5 days/week to 61% cristobalite (in calcined diatomaceous earth). Dust exposures were 2 and 5 million particles per cubic foot (mppcf), equivalent to 0.2 and 0.5 mg/m^3 cristobalite (USEPA, 1996), with occasional excursions to 50 mppcf. No lung fibrosis was detected at these levels but all levels caused accumulation of inflammatory cells in the lung parenchyma. However, in dogs fibrotic nodules developed in the hilar lymph nodes with more nodules at 5 mppcf than at 2 mppcf.

Scheuchenzuber *et al*. (1985) examined immunologic responses in Balb/c mice following inhalation of 1.954 mg/m^3 silica for 150, 300, or 570 days. Mice exposed for 570 days were tested immediately post-exposure. Those exposed for 150 or 300 days were tested immediately or were rested for 30 or 150 days to allow for possible recovery from effects of dust inhalation. Silica inhalation suppressed the number of specific plaque-forming cells (PFC) in the spleen produced in response to aerosolized *E. coli*. After 570 days of inhalation, silica also reduced the

ability of alveolar macrophages to phagocytize *Staphylococcus aureus in vitro* and impaired the ability to lyse allogeneic tumor cells (from mice other than Balb/c) *in vitro*. Silica inhalation did not affect antibody-dependent cell-mediated cytotoxic and mitogenic responses by splenic lymphocytes. (Fibrosis was not an endpoint measured, but the effect level is similar to the LOAELs in other animal studies.)

Table 18. Animal studies of silica inhalation analyzed by USEPA (1996)

Study	***Species***	***Duration*** [a]	***LOAEL***
Muhle *et al*., 1989	Rat	24 mo	1.0 mg/m^3
Scheuchenzuber *et al*., 1985	Mice	150-570 d	2.0
Burns *et al*., 1980	Mice	3-39 wk	4.9
Kutzman, 1984a	Rat	6 mo	2.0
Kutzman, 1984b	Rat	6 mo + 6 mo recovery	2.0
Wagner *et al*., 1986	Dog	Up to 2.5 yr	0.2

[a] Inhalation exposure was generally for 6 h/day, 5 d/wk.

Quartz has the ability to induce the generation of free radicals and to cause oxidative stress in tissues. Many substances that affect the quartz surface can modify this ability. Some of these modifiers could originate from other minerals, which exist together with quartz in nature. Donaldson and Borm (1998) proposed that the hazard posed by quartz may vary widely depending on the origin of the silica sample or on its contact with other chemicals/minerals. Such mechanistic data could assist in the interpretation of epidemiological studies such as those above. Experimentally their group found that DQ12 quartz, a European quartz standard which is often used in experimental studies of silica effects, is much more inflammatory in rat lung than respirable silica collected from two workplaces (Clouter *et al*., 2001).

Humans appear to show adverse effects of silica exposure at lower levels than animals (compare LOAELs in Table 18 to LOAELs/NOAELs in Table 16). Rodents tend to be obligate nose-breathers and to have extensive nasal turbinates, which may result in less silica reaching the lower lung. For silica, results in animals may not be a good predictor of human effect levels.

VI. Derivation of Chronic Reference Exposure Level (REL)

Key study	Hnizdo and Sluis-Cremer, 1993
Study population	2235 white South African gold miners
Exposure method	Workplace inhalation
Critical effects	Silicosis (313 miners) (14 %)
LOAEL	3 mg/m^3-years CDE (9 miners with silicosis)
NOAEL	2 mg/m^3-years CDE (0 miners with silicosis) or 600 $\mu g/m^3$-years silica (dust = 30% silica)
$BMCL_{01}$	2.12 (mg/m^3)–yr CDE or 0.636 (mg/m^3)–yr silica
Exposure continuity	8 h/day, 5 d/wk
Exposure duration	Average of 24 years dust exposure (10-39 years)
Average experimental exposure	235 $\mu g/m^3$-yr silica at BMC_{01} (636 x 10 m^3/20 m^3 x 270 shifts/365 days) 235 $\mu g/m^3$-yr/24 yr = 9.8 $\mu g/m^3$
Human Equivalent Concentration (HEC)	9.8 $\mu g/m^3$
LOAEL uncertainty factor	Not needed in BMC approach
Subchronic uncertainty factor	1
Interspecies uncertainty factor	1
Intraspecies uncertainty factor	3
Cumulative uncertainty factor	3
Inhalation Reference Exposure Level	3 $\mu g/m^3$ (based on 30% silica in mine dust) [respirable, as defined occupationally by ACGIH/ISO]

First supportive study	Steenland and Brown, 1995
Study population	3330 S. Dakota gold miners
Exposure method	Workplace inhalation
Critical effects	Silicosis (170 miners) (5.1%)
LOAEL	0-0.2 mg/m^3-years (5 miners with silicosis)
NOAEL	Not found
$BMCL_{01}$	0.34 (mg/m^3)–yr (see text below)
Exposure continuity	8 h/day, 5 d/wk
Exposure duration	3-36 years (average 9 years underground)
Average experimental exposure	112 $\mu g/m^3$-y (340 x 10 m^3/20 m^3 x 5 d/7 d x 48 wk/52 wk) 112 $\mu g/m^3$-y/9 y = 12.4 $\mu g/m^3$
Human Equivalent Concentration (HEC)	12.4 $\mu g/m^3$
LOAEL uncertainty factor	Not needed in BMC approach
Subchronic uncertainty factor	1
Interspecies uncertainty factor	1
Intraspecies uncertainty factor	3
Cumulative uncertainty factor	3
Inhalation Reference Exposure Level	4 $\mu g/m^3$ [respirable, as defined occupationally]

Second supportive study	Hughes *et al*., 1998
Study population	1809 California diatomaceous earth workers
Exposure method	Workplace inhalation
Critical effects	Silicosis (81 workers) (4.5%)
LOAEL	> 1, ≤ 3 mg/m^3-years (17 workers with silicosis)
NOAEL	≤ 1 mg/m^3-years (6 cases). (Six cases were observed, but Hughes *et al*. assigned the group a Relative Risk = 1 for silicosis.)
Exposure continuity	8 h/day, 5 d/wk
Exposure duration	1-45 years (mean = 11.5 years)
Average experimental exposure	≤ 330 μg/m^3-y (1000 x 10/20 x 5/7 x 48/52) ≤ 330 μg/m^3-y/ 11.5years = ≤ 29 μg/m^3
Human Equivalent Concentration (HEC)	29 μg/m^3
LOAEL uncertainty factor	3 (authors' NOAEL actually is a LOAEL)
Subchronic uncertainty factor	1
Interspecies uncertainty factor	1
Intraspecies uncertainty factor	3
Cumulative uncertainty factor	10
Inhalation Reference Exposure Level	3 μg/m^3 [respirable, as defined occupationally]

Third supportive study	Chen *et al*. (2001)
Study population	3010 Chinese tin miners
Exposure method	Workplace inhalation
Critical effects	Silicosis (1015 workers) (33.7 %)
LOAEL	10-19.99 mg CTD/m^3-years (24 cases)
NOAEL	≤ 10 mg CTD/m^3-years (2 cases) ≤ 360 μg silica/m^3 - years
$BMCL_{01}$	132 μg silica/m^3 - years
Exposure continuity	8 h/day, 5 d/wk
Exposure duration	2.2 years for NOAEL group
Average experimental exposure	40 μg/m^3-y (132 x 10/20 x 5/7 x 48/52) 40 μg/m^3-y/2.2 years = 18 μg/m^3
Human Equivalent Concentration (HEC)	18 μg/m^3
LOAEL uncertainty factor	Not needed in BMC approach
Subchronic uncertainty factor	1
Interspecies uncertainty factor	1
Intraspecies uncertainty factor	3
Cumulative uncertainty factor	3
Inhalation Reference Exposure Level	6 μg/m^3 [respirable, as defined occupationally]

Fourth supportive study	Churchyard *et al*., 2004
Study population	510-520 black South African gold miners
Exposure method	Workplace inhalation
Critical effects	Silicosis (93 cases)
LOAEL	0-0.80 mg/m^3-yr (11 cases)
NOAEL	Not identified
BMCL$_{05}$	0.673 (mg/m^3)-yr
Exposure continuity	270 shifts/year
Exposure duration	21.8 yr (6.3-34.5)
Average experimental exposure	249 (μg/m^3)-yr (673 x 10/20 x 270shifts/365) 249 (μg/m^3)-yr/21.8 yr= 11.4 μg/m^3
Human equivalent concentration (HEC)	11.4 μg/m^3
LOAEL uncertainty factor	Not needed in BMC approach
Subchronic uncertainty factor	1
Interspecies uncertainty factor	1
Intraspecies uncertainty factor	3
Cumulative uncertainty factor	3
Inhalation Reference Exposure Level	4 μg/m^3 [respirable, as defined occupationally]

The study of 2235 white South African gold miners by Hnizdo and Sluis-Cremer (1993) not only determined a NOAEL of 2 (mg/m^3)-yr CDE (600 μg/m^3-yr silica), but also had sufficient dose-response data for a BMC derivation. This study was powerful enough to detect a 1.9% incidence of silicosis (9 cases out of 474 exposed) at 0.9 mg/m^3-yr silica (0/204 vs. 9/474, p = 0.064 by Fisher exact test, two-tailed). Because this incidence represents approximately the sensitivity limit of the data, and silicosis is a severe irreversible endpoint, the BMCL$_{01}$ (*i.e.,* the lower bound estimate of the concentration at which 1% of the population develops silicosis) was selected as the basis of the chronic REL. In benchmark analysis of chronic animal studies, BMCL$_{05}$ is typically regarded by OEHHA as equivalent to a NOAEL. However, the power of this large-scale study is sufficient to demonstrate measurable responses below the 5% incidence level (which cannot then be logically considered a no-effect level). Furthermore, the endpoint measured in this epidemiological study is considered to be severe, since it represents the occurrence of clinically recognizable and irreversible disease, rather than an adverse physiological or biochemical response or a histopathological result seen at autopsy.

Benchmark Concentration (BMC) models, developed by the USEPA (BMDS versions 1.3, 1.3.1, and 1.3.2), were fit to the human data in Hnizdo and Sluis-Cremer (1993) (Table 7 and Figure 2 above). Fitting the probit model to the log dose of the Hnizdo and Sluis-Cremer (1993) data yielded an MLE$_{01}$ of 2.45 (mg/m^3)–yr CDE and a BMCL$_{01}$ of 2.12 (mg/m^3)–yr CDE (χ^2 = 0.64; p value for fit = 0.9957) (Figure 5, Table 19). (For comparison the BMCL$_{05}$ was 3.73 (mg/m^3)–yr CDE.) Fitting the logistic model to the same data yielded a BMCL$_{01}$ of 1.73 (mg/m^3)–yr CDE (χ^2 = 2.71; p value for fit = 0.8446) (Table 19). The BMCL$_{01}$ from these data is about the same as the apparent NOAEL. In general, a BMC is preferred to a NOAEL because the BMC takes into account all the dose response data in a study. The apparent NOAEL may be either above or below an actual effect level, depending on the study design and distribution of the data.

Figure 5. Probit model fit to the log dose of the Hnizdo and Sluis-Cremer data.

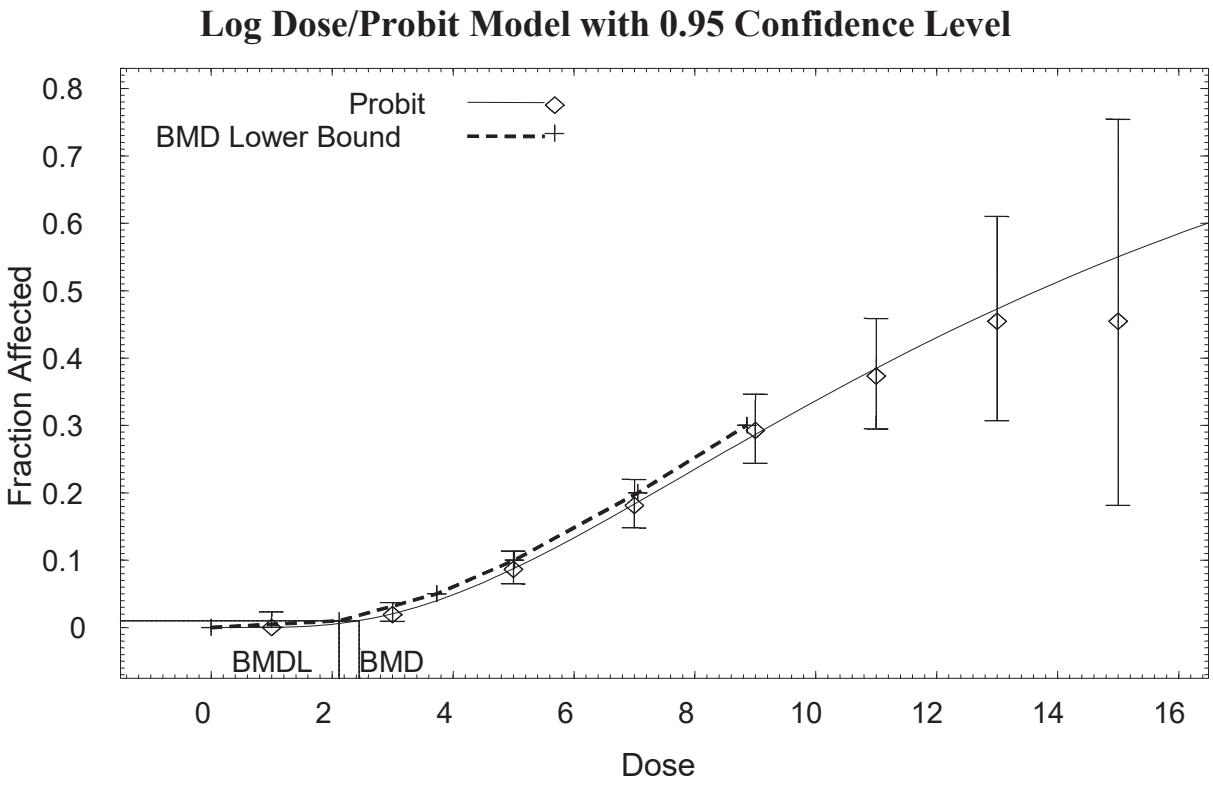


Table 19. Fits of benchmark models to the Hnizdo and Sluis-Cremer (1993) data

BMDS Model	MLE_{01}	$BMCL_{01}$	*p value for fit*
Probit-log-dose	2.45 (mg/m^3)–yr CDE	2.12 (mg/m^3)–yr CDE	0.9957
Logistic-log-dose	2.07	1.73	0.8446
Multistage (n=2)	2.47	1.89	0.7213
Quantal-quadratic	1.62	1.54	0.5017
Probit	1.56	1.32	0.0079
Logistic	1.48	1.28	0.0003
Quantal-linear	0.37	0.34	0.0000

For the estimate of 30% silica in the South African gold mine dust, Hnizdo and Sluis-Cremer (1993) relied on estimates for the years 1956-1960 by Beadle (Beadle and Bradley, 1970; Beadle, 1971). The original data, obtained by Corner House Laboratories for the South African Bureau of Mines, are partly presented by Beadle and Bradley (1970), but a more detailed presentation of exposures for various classes of workers is given by Page-Shipp and Harris

(1972). The latter paper also describes in some detail the methodology used to obtain the particle counts, and to convert those data into either respirable surface area or respirable mass values. Gibbs and Du Toit (2002) reviewed the data and methodology used by Hnizdo and Sluis-Cremer (1993) to estimate silica exposures of workers, which apparently depended on an unpublished analysis of the Corner House Laboratories' data done by Du Toit in 1991. Gibbs and Du Toit state that the exact relationship between the observed particle counts and theoretically derived mass concentrations cannot be determined, but that the uncertainties in this conversion do not appear to be severe for the dust characteristics observed in the South African mines. They accept the estimates by Beadle and Bradley (1970) of the quartz percentages in the dust, *i.e.* 54% for incinerated and acid-washed dust and 30% for unmodified dust.

However, Gibbs and Du Toit (2002) assert that Hnizdo and Sluis-Cremer (1993) incorrectly applied the 30% (total dust) silica content to figures for acid-treated dust in calculating the silica exposures of each occupational group. This contention is supported by the footnote to Table II in Hnizdo and Sluis-Cremer (1993) where the respirable dust concentration is described as "After heat and acid treatment". In order to clarify this point, OEHHA reviewed the independent reporting of the underlying data by Page-Shipp and Harris (1972). For most occupational groups, the silica exposures (shown in Table 20) calculated from Appendix I of Page-Shipp and Harris (1972), using the 54% silica content appropriate for acid-washed dust, correspond more closely to those calculated by Hnizdo and Sluis-Cremer (1993) (applying the 30% quartz content to their reported "respirable dust concentrations," i.e., the untreated dust), than to the modified, and higher, quartz exposures proposed by Gibbs and Du Toit (2002). For example, 113 exposure samples were taken for stopers.

Table 20. Estimates of silica exposures in mg/m^3 for different occupational groups in South African gold mines.

Occupation	*Shaft Sinkers*	*Developers*	*Stopers*	*Assistant miners/ Trammers*	*Shift Bosses*	*Other Officials*	*Banks/ Skips*	*Workers Near shafts*	*Boiler-makers*	*Other Artisans*	*Miscellan-eous*
Page-Shipp and Harris (1972) (Table III and Appendix I)											
Hours/shift (t)	7.70	8.00	7.80	7.70	5.20	4.00	7.50	6.50	6.30	5.70	7.20
Number of samples	10	37	113	157	43	106	33	34	41	61	11
RM x t	4.44	1.96	1.57	1.20	0.87	0.77	1.31	0.56	1.00	0.64	1.01
s.d.	3.94	1.59	1.00	0.93	0.71	0.53	1.38	0.57	0.71	0.51	0.79
Respirable Mass (RM)	0.58	0.25	0.20	0.16	0.17	0.19	0.17	0.09	0.16	0.11	0.14
Silica (54%)	0.31	**0.13**	**0.11**	**0.08**	**0.09**	**0.10**	**0.09**	**0.05**	**0.09**	**0.06**	0.08
(after acid treatment)											
Hnizdo and Sluis-Cremer (1993) (Table II)											
RM		0.48	0.37	0.27	0.30	0.30	0.13	0.10	0.19	0.19	
Silica (30%)		**0.14***	**0.11***	**0.08***	**0.09***	**0.09***	0.04	0.03	0.06	**0.06***	
(before acid treatment)											
Gibbs and Du Toit (2002) (Table 4)											
RM		0.48	0.37	0.27	0.30	0.30	0.13	0.10	0.19	0.19	
Silica (54%)		0.26	0.20	0.15	0.16	0.16	**0.07***	**0.05***	**0.10***	0.10	
(after acid treatment)											

* denotes that value is equal to or closer to the value based on Page-Shipp and Harris

The last line of Appendix I of Page-Shipp and Harris (1972) gives a mean value for stopers of 1.57 (mg/m^3)-hours respirable dust mass after acid treatment. Since the average work shift for stopers was 7.8 hours (Page-Shipp and Harris, 1972, Table III, last row), the average exposure level was 0.20 mg/m^3. If 54% of this were quartz, the quartz level would be 0.11 mg/m^3. Table II of Hnizdo and Sluis-Cremer (1993) lists 0.37 mg/m^3 respirable dust for stopers. Thirty % of 0.37 mg/m^3 equals 0.11 mg/m^3, the same value reported by Page-Shipp and Harris. In Table 4 of Gibbs and Du Toit (2002) stopers are also reported to be exposed to 0.37 mg/m^3 respirable dust. If 54% were quartz, as Gibbs and Du Toit contend, the quartz level would be 0.2 mg/m^3. For 6 of the 9 categories of workers comprising 83% of the samples taken the silica levels correspond more closely to values used by Hnizdo and Sluis-Cremer than to those suggested by Gibbs and Du Toit.

Several more recent analyses of quartz content of South African mining rock have been reported (Table 21). Kielblock *et al*. (1997) give the overall silica content of the dust as 15% for the late 1980s to early 1990s. Dr. Eva Hnizdo (personal communication, 2003), now with the U.S. National Institute of Occupational Safety and Health (NIOSH), provided a summary of various other estimates that have been made. "Past surveys indicate that the amount of airborne respirable dust in SA gold mines in 1980's and in 1970's was on average around 0.4 mg/m^3 with average quartz concentration of 0.08 mg/m^3" (about 20%). In a Ph.D. thesis submitted by the late R.E.G. Rendall (1999) on dust in the air of gold mines, the silica percentage averaged 22% during the period from 1964 to 1988. In summary,

(1) Notwithstanding some apparent contradictions in the various accounts, the silica concentrations in air proposed by Hnizdo and Sluis-Cremer, based on the Corner House Laboratory data, are a reasonable contemporary estimate of the exposures experienced by the workers examined in the study by Hnizdo and Sluis-Cremer (1993).

(2) Other, more recent estimates of percent silica in the mine dust were lower than the value of 30% used by Hnizdo and Sluis-Cremer (1993). Newer studies, which using more sophisticated methods to measure silica in the dust, indicate lower silica concentrations in the various occupational settings. Since dust levels in the mines were fairly constant for decades and quantification of silica was improving, 30% is more likely to be an overestimate than an underestimate of silica levels.

(3) Analysis of the data of Page-Shipp and Harris (1972) by OEHHA staff indicated that Hnizdo and Sluis-Cremer (1993) used the correct silica content, despite an erroneous statement in a footnote to Table II of their paper[1].

[1] Dr. Eva Hnizdo reviewed this analysis of the silica content of the dust and agrees with the assessment. ("I am very pleased that you studied carefully all the reports and came to the conclusion that our study was after all reasonably correct. Based on the Churchyard study and the measurements data I have seen in SA during the 1990s, I am also convinced that our results are reasonable estimates of the exposure of the cohort." (Hnizdo, personal communication October 2004)

Table 21. Estimates of respirable silica fraction of South African gold mine dust

Authors	*Time frame*	*% silica*		*Number of samples*	*Methods*
Beadle and Bradley, 1970	1958-1967	total dust: 25.7%; gravimetric: 28.5%; microscopy. acid-washed: 54%		142 grav; 143 elect ppt	gravimetric; precipitator + microscopy
Hnizdo and Sluis-Cremer (1993)	1956-1960	30%			precipitator + microscopy
Rendall (unpublished thesis)					
Survey 1	1987-8	17%		588	gravimetric
Survey 2	1977	20%		166	gravimetric
Survey 3	1977	17%		90	gravimetric
Survey 4a	1964-7	22%		112	gravimetric
Hnizdo (personal communication)	1970-1989	20%			
Kielblock (1997)	~1990	15.08%			Not stated
Churchyard (2004)	2000-1	14.3%			Gravimetric + X-ray diffraction

In the first supportive study Steenland and Brown (1995) found five cases of silicosis in the lowest dose group of 0 – 0.2 (mg/m^3)-yr and considered the group to be a LOAEL (Table 10 above). None of the BMDS models gave an acceptable fit at the $p \geq 0.05$ level using six or seven silica levels. The closest was the quantal quadratic model ($\chi^2 = 9.62$; $p = 0.0473$), which resulted in a BMC_{01} for silica of 0.43 (mg/m^3)–yr using the six lowest levels of silica. In risk assessment, the highest dose or doses are often dropped in order to obtain an acceptable fit of the model to the data. This is reasonable with the benchmark approach since the highest doses should be least informative and the doses in the low dose region near the benchmark should be most informative for the benchmark concentration (USEPA, 1995; Filipsson *et al*., 2003). Fitting the probit model to the log dose of the five lowest silica levels from Steenland and Brown yielded a $BMCL_{01}$ of 0.34 (mg/m^3)–yr CDE ($\chi^2 = 1.32$; p value for fit = 0.5177). [For comparison, $BMCL_{05} = 0.85$ (mg/m^3)–yr CDE.] Fitting the quantal quadratic model gave a $BMCL_{01}$ of 0.45 (mg/m^3)–yr ($\chi^2 = 3.36$; $p = 0.3395$). Use of the BMC_{01} value of 0.34 (mg/m^3)–yr CDE from the log dose probit model resulted in a chronic REL estimate for crystalline silica of 4 μg/m^3. Steenland and Brown stated that "silicosis has no background rate for non-exposed populations that changes with age or calendar time" and thus they assumed that the five silicotics in the 0 – 0.2 (mg/m^3)-yr were exposed to silica in the mines.

In a second supportive study, Hughes *et al*. (1998) found six cases of silicosis in the lowest exposure group of ≤ 1 mg/m^3-yr but considered that group to be a NOAEL, not a LOAEL. If the lowest exposure group is used as a NOAEL, a chronic REL of 10 μg/m^3 is calculated from the

data. Hughes *et al*. (1998) cite examples of possible non-occupational chest radiograph opacities (due, for example, to age or smoking) to explain the six cases in the lowest exposure group. However, due to the rarity of silicosis the six cases are biologically significant. OEHHA considers that the six cases may be work related, not cases of environmental or background silicosis. When a LOAEL to NOAEL UF of 3 is applied to the data of Hughes *et al*. (1998), the estimated REL is 3 $\mu g/m^3$.

In a third supportive study, Chen *et al*. (2001) found two cases of silicosis in the lowest exposure group of ≤ 10 mg CTD/m^3-years and considered that exposure level to be a NOAEL. One of the advantages of the benchmark dose analysis is that a NOAEL/LOAEL controversy, such as the one above with the Hughes *et al*. (1998) data, does not impact the procedure. The chart of the Chen *et al*. data above (Figure 4) indicates that the dose response is linear at low doses. Fitting the probit model to the log dose of the four lowest data points yielded a BMC_{01} of 0.132 (mg/m^3) - yr CDE ($\chi^2 = 2.19$; p value for fit = 0.335). Use of five, six, or seven data points gave BMC_{01}s of 0.14 to 0.17, but the p values were less than 0.1. For comparison, fitting the logistic model to the log dose of the four lowest data points yielded a $BMCL_{01}$ of 0.093 (mg/m^3) - yr CDE ($\chi^2 = 4.86$; p value for fit = 0.0879). An inhalation chronic Reference Exposure Level for crystalline silica of 6 $\mu g/m^3$ was estimated from the Chen *et al*. data.

The fourth supportive study is that of black South African gold miners by Churchyard *et al*. (2003, 2004). A problem with this data set is the statistical "noise" in the lower exposure groups; e.g., the lowest exposure group has a higher incidence of silicosis (11/103) than the next group (8/97). This noise causes problems in estimating a low benchmark such as the $BMCL_{01}$ used with the Hnizdo and Sluis-Cremer (1993) data and with data from Steenland and Brown (1995) and Chen *et al*. (2001). The calculation therefore uses a 5% BMCL of 0.673 (mg/m^3)-yr from the probit log dose model as the benchmark, which is reasonably well within the range of the reliably observed data and which does not differ too widely from the MLE_{05} estimate of 0.955 (mg/m^3)-yr for that parameter. This $BMCL_{05}$ is not strictly comparable to the $BMCL_{01}$ calculated from the data of Hnizdo and Sluis-Cremer, but the concerns about the severity of the effect noted in the discussion of that derivation apply with equal or greater force here.

On the other hand, the variability in the low-dose data in this study implies that an unobserved factor is affecting the data. All the models predict that the "background" is substantially (as much as 5 – 10%) above zero, which is intrinsically implausible for silicosis unless there is an unrecorded additional source of silica exposure. Possibly, there were occasional excursions in the exposure of the workers in less exposed jobs, which were not captured by the systematic assessments for these job classifications. Alternatively, perhaps the assignment of an assumed zero exposure value to "non-dusty" job classifications noted in the paper was in fact inaccurate for some individuals. There may also be some distortion of the curve resulting from the "binning" of the exposure categories; certainly, the bin widths hinder the attempt to calculate a $BMDL_{01}$ in this case. In relation to the model fit, other models (including the quantal linear model, which emphasizes the likely more reliable incidences at higher dose levels) are consistent with the $BMCL_{05}$ results from the log probit model used here. Finally, the total number of cases and controls examined is fewer than in Hnizdo and Sluis-Cremer's study, which reduces the precision.

None of these issues can be resolved without recourse to the individual data, which were not available for this analysis, and may not be resolved even then. However, the derivation of an HEC of 11.2 $\mu g/m^3$ (and thus a comparison REL of 4 $\mu g/m^3$) indicates at least that the results of the earlier analysis are unlikely to have underestimated the proper value of the REL. Although the uncertainties in the Churchyard *et al*. data prevent us from being more precise, we cannot eliminate the possibility that the REL should be set lower. However, the California ambient monitoring data, although subject to considerable uncertainty as to the relevant particle size distributions, suggest a plausible lower bound on the REL, which is consistent with our analysis of Hnizdo and Sluis-Cremer's data.

Other investigators have approached the possibility that some opacities on radiographs may be due to background influences such as age and smoking. In regard to smoking, Blanc and Gamsu (1988) reviewed the literature and concluded that smoking would not interfere with the determination of silicosis by the ILO system. Based on reading 1422 films of unexposed blue-collar workers, Castellan *et al*. (1985) stated that the use of the median result of 3 readers (the same number used by Hughes *et al*.) rarely results in interpreting a chest radiograph as ILO category $\geq$ 1/0 in workers who were not exposed to dust (and regardless of smoking status).

The USEPA (1996) did a benchmark analysis with the Hnizdo and Sluis-Cremer (1993) data. They estimated that the lower bound for a 1% risk for silicosis ($BMCL_{01}$) was 1.31 (mg/m^3)-yr, which by their methods is equivalent to a continuous, 70-year exposure to 6.7 $\mu g/m^3$ silica. However, USEPA did not do a formal Reference Concentration (RfC) derivation for silica by either the BMC/UF or NOAEL/UF approach.

The key (Hnizdo and Sluis-Cremer, 1993) and supporting (Steenland and Brown, 1995; Hughes *et al*., 1998; Chen *et al*., 2001) studies were of human adults, nearly all males, who were presumably healthy, at least initially, since they were able to work. Thus there is need to protect the sensitive members of the population, especially children, in whose airways penetration of silica particles will be greater (Phalen *et al*., 1985; Schiller-Scotland *et al*., 1994; Oldham *et al*., 1997; Bennett and Zeman, 1998). In addition, women may be more sensitive than men to the development of silicosis (Gerhardsson and Ahlmark, 1985; Katsnelson *et al*., 1986). The selection of three as the intraspecies uncertainty factor (UF_H) was based on several considerations.

(1) The workers who developed silicosis at low silica concentrations are by definition the most sensitive workers to silica-induced silicosis. Because of the large population of workers examined in these studies (more than 14,000), the sensitive individuals represent at least part of the range of sensitivity to be expected in the general population. This may justify reducing the UF_H from the default value of 10. Since these workers did not include children, the elderly, or females (except for the 215 females in Chen *et al*.), some uncertainty related to inter-individual variability remains. Therefore, a UF_H of 3 rather than 1 is chosen.

(2) Mukherji *et al*. (1993) reported mean ambient silica levels (in PM_{10}) at three locations in the northern part of Santa Barbara County, California (see the Appendix to this report). At Santa Maria (an urban site) the level was 2.3 $\mu g/m^3$; in Santa Ynez (a rural site) 0.6 $\mu g/m^3$; and in Buellton (a remote background site) 0.2 $\mu g/m^3$ crystalline silica. Thus, use

of a human intraspecies uncertainty factor (UF_H) of 10 with the data from the key study would result in an estimated chronic REL of 0.9 μg/m^3 (ACGIH method), a level in the range of ambient levels in California. Although the reported levels at the urban site may (according to the authors) have reflected some anthropogenic contributions such as disturbance and tracking of siliceous road dust, the rural and remote site values are apparently (perhaps conservatively) reflective of the natural background to which all California residents are exposed. (U.S. EPA (1996) found slightly higher average ambient levels of silica in PM_{10}; this average may include some sites affected by disturbance and emissions.) There is no evidence that these background levels of silica are causing silicosis. On the other hand, silicosis in the general population is not a target for medical attention, and autopsy rates are very low, so the possibility of a low frequency of response at these levels cannot be entirely dismissed. On balance, it appears plausible that a REL of 3 μg/m^3 (benchmark + UF_H = 3) would be protective of the general population.

(3) The dose-response curve for silicosis due to inhalation of crystalline silica is steep, and an upward curvature of this dose response was seen in some studies (Figure 7-1 in USEPA, 1996). It is notable that, whereas exposures in the 1-3 μg/m^3 range are apparently without effect (based on the benchmark calculations and the California ambient background data), Rice and Stayner (1995) described a LOAEL for silicosis of 8 μg/m^3 in gold miners (Table 16; based on data from McDonald and Oakes [1984]). This finding may partly reflect differences in physical state of the silica, and co-exposures, but it might indicate that, although the chronic REL should be protective of public health, chronic exposures only moderately exceeding the REL may lead to clinically observable disease.

The animal studies gave LOAELs for silica of 0.2 mg/m^3 in dogs and from 1 to 4.9 mg/m^3 in rodents. After extrapolation to equivalent continuous time and application of LOAEL to NOAEL, interspecies, and intraspecies UFs, the estimated chronic RELs from animal data are all less than 1 μg/m^3. This reflects in part the greater uncertainty in extrapolating from animal studies to predicted human health effects.

The silica particles of concern in the causation of silicosis are those of respirable size. California EPA defines 'respirable' as particles 10 μm or less MMAD. This reflects one usual type of sampler (for "PM_{10}") used for ambient air sampling in the general environment. The other usual type of environmental sampler, $PM_{2.5}$, collects even smaller particles. There are differences in the size range distribution between a typical PM_{10} measuring device and the NIOSH type personal samplers, or other devices with similar size selection properties, used by the investigators in the epidemiological studies. The NIOSH-type samplers capture 50% of particles with a MMAD of 4 μm, and higher percentages of smaller particles. A smaller proportion of larger particles between 4 and 10 μm in aerodynamic diameter will also be collected. Figure 5, from Volume I of U.S. EPA's Third External Review Draft of Air Quality Criteria for Particulate Matter (April 2002), includes particle penetration curves for PM_{10}, $PM_{2.5}$, and occupational samplers.

Figure 5. Size cut curves of particle penetration

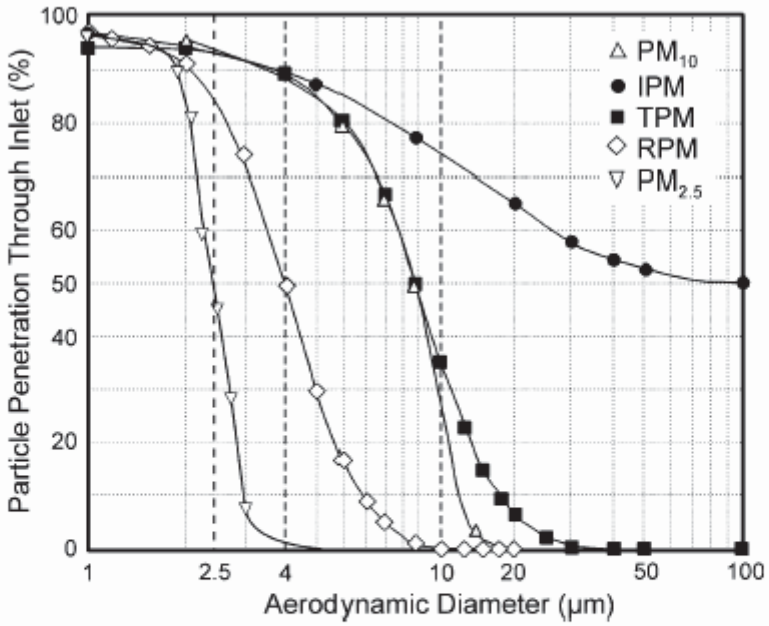


Figure 2-6. Specified particle penetration (size-cut curves) through an ideal (no-particle-loss) inlet for five different size-selective sampling criteria. Regulatory size cuts are defined in the Code of Federal Regulations; $PM_{2.5}$ (2001c), PM_{10} (2001a). $PM_{2.5}$ is also defined in the Federal Register (1997). Size-cut curves for inhalable particulate matter (IPM), thoracic particulate matter (TPM) and respirable particulate matter (RPM) size cuts are computed from definitions given by American Conference of Governmental and Industrial Hygienists (1994).

The NIOSH samplers are designed to mimic the size range of particles that reach into the bronchiolar and alveolar spaces (what the occupational community calls respirable). PM_{10} samplers are meant to capture particles that penetrate the entire length of the lower respiratory tree, including those that penetrate to the tracheobronchial and alveolar regions. Penetration (and therefore presumably deposition) by particle size is complex, and is dependent on the aerodynamic diameter, hygroscopicity, and electrostatic charge of the particles, and on a number of host factors including airway structure and geometry, as well as depth, rate, and mode of breathing (nasal vs. oronasal). The fractional penetration in the various regions of the respiratory tract is not linear with respect to size. Generally, though, larger particles impact higher in the respiratory tree (the extrathoracic and tracheobronchial regions), while smaller particles show greater penetration to the lower tracheobronchial and alveolar regions. There are a number of models of regional penetration and deposition in the respiratory tract, as well as some measurements. Chan and Lippmann (1980) showed peak alveolar deposition for particles about 3 µm MMAD with deposition dropping above and below that. Their data and model indicate

that tracheobronchial deposition rises rapidly above about 3 μm MMAD. Available data also indicate significant inter-individual variability in fractional deposition. The ICRP (1994) model used in evaluating risk from radioactive particles indicates that total deposition in the respiratory tract for particles 3 μm in activity median thermodynamic diameter (AMTD) is about 0.78 with a regional deposition fraction of 0.077 for the alveolar region for a reference male worker during nasal breathing. The same model predicts a total deposition in the respiratory tract of 0.77 for 10 μm AMTD particles and a deposition fraction of 0.024 in the alveolar region. Thus, many particles with a 10 μm MMAD get into the alveolar space. A smaller difference in regional penetration and deposition is predicted for mouth breathers. Therefore, if only the size range measured by the samplers used in the studies were considered, the measurement might underestimate the amount of silica that reaches the gas exchange regions of the lung, depending on the actual particle size distributions in the occupational studies and in the environments in which the REL is to be applied. Unfortunately, neither the occupational nor the environmental silica particle size distributions are known in detail; measurements have been reported only in terms of NIOSH sampler results or PM_{10} cutoff values.

It is generally assumed that the silicosis is induced by that fraction of the silica that reaches the alveoli. Nevertheless, no actual data exonerate the coarser particles in the 4 - 10 μm range. A fraction of these particles can enter the bronchioles and alveoli. However, some data from South African gold mines indicate that more than 99% of the crystalline silica dust can be in the $PM_{2.5}$ fraction (Sichel, 1957). Thus, the samplers used in the key study appear to be collecting the biologically relevant range of particles in that situation.

In the absence of comprehensive data on the silicosis-inducing activity of different particle sizes, it is not possible to adjust the REL for different particle size distributions, which might be found in the general environment, or for different measurement methods. The REL is therefore specified as applicable to concentrations of particles having a size range (and reactivity) similar to those measured in the occupational studies [respirable as defined occupationally (ISO, 1995; NIOSH, 2003; ACGIH, 2004)]. Results obtained by other sampling methods would need to be corrected for any difference in size selectivity of the method used. Such a correction factor would be specific to the particle size distribution present at the site studied, so no general correction factors can be proposed. A more inclusive sampling procedure, such as that used for PM_{10}, would overestimate the relevant exposure in any situation, and so would be inappropriate for precise risk quantification. However, PM_{10} would be useful as a screening method to establish that a particular situation is unlikely to present a hazard. For example, if the silica concentration in PM_{10} modeled at a receptor is less than the REL (3 $\mu g/m^3$), occupationally respirable silica will also be less than 3 $\mu g/m^3$, so a facility would not pose a risk due to silica at that receptor. If the silica concentration in $PM_{2.5}$ modeled at a receptor is less than 3 $\mu g/m^3$ but PM_{10} is greater than 3 $\mu g/m^3$, further testing would be needed. If both $PM_{2.5}$ and PM_{10} exceeded the REL, the chronic Hazard Index would exceed 1 to an undetermined extent, suggesting a need for risk management. More precise determination of the amount of material in the respirable size fraction for environmental samples may require further work on measurement methodology, since ISO (1995) and similar occupational methods have not been validated for the lower levels encountered in environmental samples.

VII. Data Strengths and Limitations for Development of the REL

The strengths of the inhalation REL for silica include:

(1) The availability of several long-term studies of inhalation in workers at varying exposure concentrations (see Summary Table 15 above), with adequate histopathological and radiological analysis, and with adequate follow-up.
(2) The finding of a dose-response effect for silicosis in several of the studies (e.g., Hnizdo and Sluis-Cremer, 1993; Steenland and Brown, 1995; Chen *et al*., 2001).
(3) The observation of a NOAEL in some studies including the key study (summarized by Rice and Stayner, 1995).
(4) The power of the Hnizdo and Sluis-Cremer (1993) data to detect a small effect.

Major areas of uncertainty are:

(1) The limited follow-up of the cohort members in some studies (e.g., Muir *et al*., 1989; Rosenman *et al*., 1996) with consequent under-ascertainment of silicosis (even to the extent that such studies are useless for determining exposure-response).
(2) The general underestimation of silicosis by radiography alone (Hnizdo *et al*., 1993), which results in higher, less health-protective chronic REL estimates.
(3) The possible underreporting of silicosis where complete radiographic data and autopsy data are not available (Steenland and Brown, 1995).
(4) The uncertainties in exposure estimation, especially when reconstructing historical levels of silica exposure (Seixas *et al*., 1997; Gibbs and Du Toit, 2002) including the variability in the estimates of percent quartz in the South African mine dust (Beadle, 1971; Hnizdo and Sluis-Cremer, 1993; Kielblock *et al*., 1997; Gibbs and Du Toit, 2002; Hnizdo, personal communication) and when converting particle counts to mass.
(5) The differences in percent silicosis in different studies at what were considered similar silica levels and similar exposure duration (see Summary Table 15 above).
(6) The variability in toxicity of various forms of silica (e.g., freshly fractured vs. aged quartz; cristobalite vs. quartz) although all forms have toxicity (Table 17).
(7) The limited information on silica particle size (including its variability) in the epidemiological studies, other than that the silica was respirable, and the variability in particle penetration and deposition as a function of particle size in the respiratory tract in the human population (e.g., Heyder *et al*., 1982; ICRP, 1994; Hattis *et al*., 2001).
(8) The use of area samplers rather than personal samplers to estimate exposure, which usually results in an underestimation of silica exposure (Cherrie, 1999).

VIII. Potential for Differential Impacts on Children's Health

Silica is a respiratory irritant and a modifier of immune function. Since the key study involved over 2000 men, some were likely to be more sensitive to silica than others. In addition, we used a benchmark of 1% adverse effect, rather than the usual 5 %. Thus, use of the human intraspecies uncertainty factor (UF_H) of 3 should result in a REL that adequately protects most members of the general population. Exacerbation of asthma, which has a more severe impact on children than on adults, is a known response to some respiratory irritants. However, there is no

data on such a response to silica in infants or children. The epidemiological studies used in the derivation of the REL did not include children. If children's susceptibility were much greater than that of adults, it would be expected that clinical disease would be evident in children following exposures in the upper range of the respirable silica levels measured in ambient air in California. No such reports have been identified in the literature. There are no data on silica's effects on the immune system of children.

OEHHA is currently evaluating its risk assessment methodology, in particular the UF_H, for its adequacy in protecting infants and children. Since children have smaller airways than adults and breathe more air on a body weight basis, penetration and deposition of particles in the airways and alveoli in children is likely greater than that in adults exposed to the same concentration (Phalen *et al*., 1985; Schiller-Scotland *et al*., 1994; Oldham *et al*., 1997; Bennett and Zeman, 1998).

IX. References

ACGIH. 1999. American Conference of Governmental Industrial Hygienists. 1999 TLVs and BEIs. Threshold Limit Values for chemical substances and physical agents and Biological Exposure Indices. Cincinnati: ACGIH.

ACGIH. 2000. American Conference of Governmental Industrial Hygienists. 2000 TLVs and BEIs. Threshold Limit Values for chemical substances and physical agents and Biological Exposure Indices. Cincinnati: ACGIH.

ACGIH. 2004. American Conference of Governmental Industrial Hygienists. 2004 TLVs and BEIs. Threshold Limit Values for chemical substances and physical agents and Biological Exposure Indices. Cincinnati: ACGIH. pp. 73-6.

American Thoracic Society. 1997. Adverse effects of crystalline silica exposure. Am Respir Crit Care Med. 155:761-5.

Balmes J, Becklake M, Blanc P, Henneberger P, Kreiss K, Mapp C, Milton D, Schwartz D, Toren K, Viegi G. Environmental and Occupational Health Assembly, American Thoracic Society. 2003. American Thoracic Society Statement: Occupational contribution to the burden of airway disease. Am J Respir Crit Care Med. 167(5):787-97.

Bar-Ziv J, Goldberg JM. 1974. Simple siliceous pneumoconiosis in Negev Bedouins. Arch Environ Health. 29:121-6.

Beadle DG. 1971. The relationship between the amount of dust breathed and the development of radiological signs of silicosis: an epidemiologic study of South African gold miners. In: Walton WH (ed). Inhaled particles III. Oxford: Pergamon Press, pp. 953-64.

Beadle DG, Bradley AA. 1970. The composition of airborne dust in South African gold mines. In: Shapiro HA (ed). Pneumoconiosis. Proceedings of the International Conference. Johannesburg 1969. Cape Town: Oxford University Press. pp. 462-6.

Begin R, Ostiguy G, Fillion R, Colman N. 1991. Computed tomography scan in the early detection of silicosis. Am Rev Respir Dis. 144(3 Pt 1):697-705.

Bennett WD, Zeman KL. 1998. Deposition of fine particles in children spontaneously breathing at rest. Inhal Toxicol. 10:831-42.

Blanc PD, Gamsu G. 1988. The effect of cigarette smoking on the detection of small radiographic opacities in inorganic dust diseases. J Thorac Imaging. 3(4):51-6.

Burns CA, Zarkower A, Ferguson FG. 1980. Murine immunological and histological changes in response to chronic silica exposure. Environ Res. 21(2):298-307.

CARB. 2001. California Air Resources Board. California Emissions Inventory Development and Reporting System (CEIDARS). Data from Data Base Year 2001.

Castellan RM, Sanderson WT, Petersen MR. 1985. Prevalence of radiographic appearance of pneumoconiosis in an unexposed blue collar population. Am Rev Respir Dis. 131(5):684-6.

Cavariani F, Di Pietro A, Miceli M, Forastiere F, Biggeri A, Scavalli P, Petti A, Borgia P. 1995. Incidence of silicosis among ceramic workers in central Italy. Scand J Work Environ Health. 21 (Suppl 2):58-62

Chan TL, Lippmann M. 1980. Experimental measurements and empirical modeling of the regional deposition of inhaled particles in humans. Am Ind Hyg Assoc J. 41:399-409.

Chen W, Zhuang Z, Attfield MD, Chen BT, Gao P, Harrison JC, Fu C, Chen JQ, Wallace WE. 2001. Exposure to silica and silicosis among tin miners in China: exposure-response analyses and risk assessment. Occup Environ Med. 58(1):31-7.

Chia KS, Ng TP, Jeyaratnam J. 1992. Small airways function of silica-exposed workers. Am J Ind Med. 22(2):155-62.

Cherrie JW. 1999. The effect of room size and general ventilation on the relationship between near and far-field concentrations. Appl Occup Environ Hyg. 14(8):539-46.

Cherrie JW, Aitken RJ. 1999. Measurement of human exposure to biologically relevant fractions of inhaled aerosols. Occup Environ Med. 56(11):747-52.

Churchyard GJ, Ehrlich R, teWaterNaude JM, Pemba L, Dekker K, Vermeijs M, White N, Myers J (2004). Silicosis prevalence and exposure-response relations in South African goldminers. Occup Environ Med. 61(10):811-6.

Churchyard G, Pembe L, Magadla B, Dekker K, Vermeijs M, Ehrlich R, teWaterNaude J, Myers J (2003). Silicosis prevalence and exposure response relationships in older black miners on a South African goldmine. Final report. Safety in Mines Research Advisory Committee. Simhealth 606. March 7, 2003.

Churg A. 1996. The uptake of mineral particles by pulmonary epithelial cells. Am J Respir Crit Care Med. 154(4 Pt 1):1124-40.

Clouter A, Brown D, Hohr D, Borm P, Donaldson K. 2001. Inflammatory effects of respirable quartz collected in workplaces versus standard DQ12 quartz: particle surface correlates. Toxicol Sci. 63(1):90-8.

Craighead JE, Vallyathan NV. 1980. Cryptic pulmonary lesions in workers occupationally exposed to dust containing silica. JAMA. 244(17):1939-41.

Davis GS, Leslie KO, Hemenway DR. 1998. Silicosis in mice: effects of dose, time, and genetic strain. J Environ Pathol Toxicol Oncol. 17(2):81-97.

Davis LK, Wegman DH, Monson RR, Froines J. 1983. Mortality experience of Vermont granite workers. Am J Ind Med. 4(6):705-23.

Ding M, Chen F, Shi X, Yucesoy B, Mossman B, Vallyathan V. 2002. Diseases caused by silica: mechanisms of injury and disease development. Int Immunopharmacol. 2(2-3):173-82.

Donaldson K, Borm PJ. 1998. The quartz hazard: a variable entity. Ann Occup Hyg. 42(5):287-94.

Elzea JM. 1997. The regulation of crystalline silica: an industry perspective. J Expo Anal Environ Epidemiol. 7(3):377-84.

Filipsson AF, Sand S, Nilsson J, Victorin K. 2003. The benchmark dose method--review of available models, and recommendations for application in health risk assessment. Crit Rev Toxicol. 33(5):505-42

Finkelstein MM. 2000. Silica, silicosis, and lung cancer: a risk assessment. Am. J. Ind. Med. 38(1):8-18.

Gerhardsson L, Ahlmark A. 1985. Silicosis in women. Experience from the Swedish Pneumoconiosis Register. J Occup Med. 27(5):347-50.

Gibbs GW, Du Toit RS. 2002. Estimating the quartz exposure of South African gold miners. Ann Occup Hyg. 46(7):597-607.

Glover JR, Bevan C, Cotes JE, Elwood PC, Hodges NG, Kell RL, Lowe CR, McDermott M, Oldham PD. 1980. Effects of exposure to slate dust in North Wales. Br J Ind Med. 37(2):152-62.

Goldsmith JR, Goldsmith DF. 1993. Fiberglass or silica exposure and increased nephritis or ESRD (end-stage renal disease). Am J Ind Med. 23(6):873-81.

Graham WG, Ashikaga T, Hemenway D, Weaver S, O'Grady RV. 1991. Radiographic abnormalities in Vermont granite workers exposed to low levels of granite dust. Chest. 100(6):1507-14.

Greaves IA. 2000. Not-so-simple silicosis: a case for public health action. Am J Ind Med. 37(3):245-51.

Green FHY, Vallyathan V. 1996. Pathologic responses to inhaled silica. In: Castranova V, Vallyathan V, Wallace WE (eds.). Silica and Silica-Induced Lung Diseases. Boca Raton: CRC Press, 1996, pp. 39-59.

Hattis D, Russ A, Goble R, Banati P, Chu M. 2001. Human interindividual variability in susceptibility to airborne particles. Risk Anal. 21(4):585-99.

HSDB. 2001. Hazardous Substances Data Bank. National Library of Medicine, Bethesda, MD. Available at: http://toxnet.nlm.nih.gov.

Heyder J, Gebhart J, Stahlhofen W, Stuck B. 1982. Biological variability of particle deposition in the human respiratory tract during controlled and spontaneous mouth-breathing. Ann Occup Hyg. 26(1-4):137-47.

Hnizdo E, Murray J, Sluis-Cremer GK, Thomas RG. 1993. Correlation between radiological and pathological diagnosis of silicosis: an autopsy population based study. Am J Ind Med. 24(4):427-45.

Hnizdo E, Sluis-Cremer GK. 1993. Risk of silicosis in a cohort of white South African gold miners. Am J Ind Med. 24(4):447-57.

Hnizdo E, Vallyathan V. 2003. Chronic obstructive pulmonary disease due to occupational exposure to silica dust: a review of epidemiological and pathological evidence. Occup Environ Med. 60(4):237-43.

Hughes JM. 1995. Radiographic evidence of silicosis in relation to silica exposure. Appl Occup Environ Hyg. 10(12) 1064-9.

Hughes JM, Weill H, Checkoway H, Jones RN, Henry MM, Heyer NJ, Seixas NS, Demers PA. 1998. Radiographic evidence of silicosis risk in the diatomaceous earth industry. Am J Respir Crit Care Med. 158(3):807-14.

Hughes JM, Weill H, Rando RJ, Shi R, McDonald AD, McDonald JC. 2001. Cohort mortality study of North American industrial sand workers. II. Case-referent analysis of lung cancer and silicosis deaths. Ann Occup Hyg. 45(3):201-7.

IARC Monographs on the Evaluation of Carcinogenic Risks to Humans. Volume 68. Silica, Some Silicates, Coal Dust and para-Aramid Fibrils. Lyon: IARC, 1997.

ICRP. 1994. International Commission for Radiological Protection. Human Respiratory Tract Model for Radiological Protection. ICRP Publication 66. Annals of the ICRP. 24(1-3):1-482.

International Labour Office. 1980. Guidelines for the use of ILO International Classification of radiographs of pneumoconiosis. Rev. Ed. Occupational Safety and Health Services, No. 22 (Rev.). ILO, Geneva.

ISO. 1995. International Organization for Standardization. Air quality – Particle size fractionation definitions for health-related sampling. ISO No. 7708:1995. ISO, Geneva.

Katsnelson BA, Polzik EV, Privalova LI. 1986. Some aspects of the problem of individual predisposition to silicosis. Environ Health Perspect. 68:175-85.

Kielblock AJ, Franz RM, Unsted AD, vander Linde A, Ashworth SGE. 1997. Quantitation of occupational health risks in the South African mining industry and assessment of sources of uncertainty in the estimates. SIMRAC Safety in Mines Advisory Committee final project report, project no. SIMRISK 401. Johannesburg: CSIR Division of Mining Technology.

Kitto PH. 1960. Methods of dust measurement. In: Orenstein AJ (ed). Proceedings of Pneumoconiosis Conference held at the University of Witwatersrand, Johannesburg 9th – 24th February, 1959. London: J & A Churchill Ltd. pp. 13-20.

Kitto PH. 1970. South African methods for the assessment of dust in gold and coal mines. In: Shapiro HA (ed). Pneumoconiosis. Proceedings of the International Conference. Johannesburg 1969. Cape Town: Oxford University Press. pp. 457-61.

Kreiss K, Greenberg LM, Kogut SJ, Lezotte DC, Irvin CG, Cherniack RM. 1989. Hard-rock mining exposures affect smokers and nonsmokers differently. Results of a community prevalence study. Am Rev Respir Dis. 139(6):1487-93.

Kreiss K, Zhen B. 1996. Risk of silicosis in a Colorado mining community. Am J Ind Med. 30(5):529-39.

Kutzman RS. 1984a. A study of Fischer 344 rats exposed to silica dust for six months at concentrations of 0, 2, 10 or 20 mg/m^3. Upton, NY: Brookhaven National Laboratory; report no. BNL 34617.

Kutzman RS. 1984b. A study of Fischer 344 rats exposed to silica dust for six months at concentrations of 0, 2, 10 or 20 mg/m^3, then maintained for six months prior to assessment. Upton, NY: Brookhaven National Laboratory; report no. BNL 35735.

Lapp NL, Castranova V. 1993. How silicosis and coal workers' pneumoconiosis develop--a cellular assessment. Occup Med. 8(1):35-56.

Legrand-Cattan K, Vuillaume M, Iwatsubo Y, Ameille J, Brochard P, Letourneux M, Housset B, Laureillard J, and Pairon J. 1998. Silicosis in the ceramic industry: dose-response relationship. In: Advances in the Prevention of Occupational Respiratory Disease. Chiyotani K, Hosoda Y, Aizawa Y (eds.). Amsterdam: Elsevier. Pp. 113-117.

Le Roux WL. 1970. Recorded dust conditions and possible new sampling strategies on South African gold mines. In: Shapiro HA (ed). Pneumoconiosis. Proceedings of the International Conference. Johannesburg 1969. Cape Town: Oxford University Press..

Lippmann N. 2001. Size-selective health hazard sampling. In: Cohen BS, McCammon Jr CS (eds.). Air Sampling Instruments for Evaluation of Atmospheric Contaminants. 9th ed. Cincinnati: ACGIH. pp. 93-134.

't Mannetje A, Steenland K, Attfield M, Boffetta P, Checkoway H, DeKlerk N, Koskela RS. 2002. Exposure-response analysis and risk assessment for silica and silicosis mortality in a pooled analysis of six cohorts. Occup Environ Med. 59(11):723-728.

McDonald AD, McDonald JC, Rando RJ, Hughes JM, Weill H. 2001. Cohort mortality study of North American industrial sand workers. I. Mortality from lung cancer, silicosis and other causes. Ann Occup Hyg. 45(3):193-9.

McDonald JC, Oakes D. 1984. Exposure-response in miners exposed to silica. In: Sixth International Pneumoconiosis Conference. 1983. Bochum, Germany. Vol 1. Geneva: International Labour Office (ILO). Pp. 114-23.

Mehnert WH, Staneczek W, Mohner M, Konetzke G, Muller W, Ahlendorf W, Beck B, Winkelmann R, Simonato L. 1990. A mortality study of a cohort of slate quarry workers in the German Democratic Republic. IARC Sci Publ. Vol. 97. pp. 55-64.

Mossman BT, Churg A. 1998. Mechanisms in the pathogenesis of asbestosis and silicosis. Am J Respir Crit Care Med. 157(5 Pt 1):1666-80.

Muhle H, Bellman B, Creutzenberg O, Koch W, Dasenbrock C, Ernst H, Mohr U, Morrow P, Mermelstein R. 1998. Pulmonary response to toner, TiO_2, and crystalline silica upon chronic inhalation exposure in Syrian golden hamsters. Inhal Toxicol. 10:699-729.

Muhle H, Takenaka S, Mohr U, Dasenbrock C, Mermelstein R. 1989. Lung tumor induction upon long-term low-level inhalation of crystalline silica. Am J Ind Med. 15(3):343-6.

Muir DC, Julian JA, Shannon HS, Verma DK, Sebestyen A, Bernholz CD. 1989. Silica exposure and silicosis among Ontario hardrock miners: III. Analysis and risk estimates. Am J Ind Med. 16(1):29-43.

Mukherji S, Petrini J, Murphy T. 1993. North Santa Barbara County Crystalline Silica Study. Prepared for U.S. Environmental Protection Agency Region IX. Goleta, CA: Santa Barbara County Air Pollution Control District.

Murray J, Kielkowski D, Reid P (1996). Occupational disease trends in black South African gold miners. An autopsy-based study. Am J Respir Crit Care Med. 153(2):706-10.

NIOSH. 1974. National Institute for Occupational for Safety and Health. Criteria for a Recommended Standard. Occupational Exposure to Crystalline Silica. DHEW (NIOSH) Publication No. 75-120.

NIOSH (1994, 1996, 1998, 2003) Manual of Analytical Methods, 4th ed. DHHS (NIOSH) Publication 94-113 (August, 1994), 1st Supplement Publication 96-135, 2nd Supplement Publication 98-119, 3rd Supplement 2003-154. Schlecht PC, O'Connor PF, eds.

Ng TP, Chan SL. 1992. Lung function in relation to silicosis and silica exposure in granite workers. Eur Respir J. 5(8):986-91.

Ng TP, Chan SL. 1994. Quantitative relations between silica exposure and development of radiological small opacities in granite workers. Ann Occup Hyg. 38 Suppl 1:857-63.

OEHHA. 1999. Air Toxics Hot Spots Program Risk Assessment Guidelines. Part I. The Determination of Acute Reference Exposure Levels for Airborne Toxicants. Available online at http://www.oehha.ca.gov.

Oldham MJ, Mannix RC, Phalen RF. 1997. Deposition of monodisperse particles in hollow models representing adult and child-size tracheobronchial airways. Health Phys. 72(6):827-34.

Olivetti L, Grazioli L, Milanesio L, Provezza A, Chiodera P, Tassi G, Bergonzini R. 1993. Anatomo-radiologic definition of minimal interstitial silicosis and diagnostic contribution of high-resolution computerized tomography [in Italian]. Radiol Med (Torino). 85(5):600-5.

Oxman AD, Muir DC, Shannon HS, Stock SR, Hnizdo E, Lange HJ. 1993. Occupational dust exposure and chronic obstructive pulmonary disease. A systematic overview of the evidence. Am Rev Respir Dis. 148(1):38-48.

Page-Shipp RJ, Harris E. 1972. A study of dust exposure of South African white gold miners. South Afr Inst Mining Metall. 73:10-24.

Park R, Rice F, Stayner L, Smith R, Gilbert S, Checkoway H. 2002. Exposure to crystalline silica, silicosis, and lung disease other than cancer in diatomaceous earth industry workers: a quantitative risk assessment. Occup Environ Med. 59(1):36-43.

Parks CG, Conrad K, Cooper GS. 1999. Occupational exposure to crystalline silica and autoimmune disease. Environ Health Perspect. 107 Suppl 5:793-802.

Phalen RF, Oldham MJ, Beaucage CB, Crocker TT, Mortensen JD. 1985. Postnatal enlargement of human tracheobronchial airways and implications for particle deposition. Anat Rec. 212(4):368-80.

Rando RJ, Shi R, Hughes JM, Weill H, McDonald AD, McDonald JC. 2001. Cohort mortality study of North American industrial sand workers. III. Estimation of past and present exposures to respirable crystalline silica. Ann Occup Hyg. 45(3):209-16.

Rendall REG. 1999. The nature of dusts in the air of gold mines and foundries and the risk of silicosis. Thesis submitted (posthumously) to the University of the Witwatersrand.

Rice CH, Harris RL, Checkoway H, Symons MJ. 1986. Dose-response relationships for silicosis from a case-control study of North Carolina dusty trades workers. In: Goldsmith DF, Winn DM, Shy CM (eds). Silica, Silicosis, and Cancer. Controversy in Occupational Medicine. New York: Prager. Pp. 77-86.

Rice FL, Stayner LT. 1995. Assessment of silicosis risk for occupational exposure to crystalline silica. Scand J Work Environ Health. 21 Suppl 2:87-90.

Rosenman KD, Reilly MJ, Rice C, Hertzberg V, Tseng CY, Anderson HA. 1996. Silicosis among foundry workers. Implication for the need to revise the OSHA standard. Am J Epidemiol. 144(9):890-900. Saiyed HN, Chatterjee BB. 1985. Rapid progression of silicosis in slate pencil workers: II. A follow-up study. Am J Ind Med. 8(2):135-42.

Saiyed HN, Parikh DJ, Ghodasara NB, Sharma YK, Patel GC, Chatterjee SK, Chatterjee BB. 1985. Silicosis in slate pencil workers: I. An environmental and medical study. Am J Ind Med. 8(2):127-33.

Saiyed HN, Sharma YK, Sadhu HG, Norboo T, Patel PD, Patel TS, Venkaiah K, Kashyap SK. 1991. Non-occupational pneumoconiosis at high altitude villages in central Ladakh. Br J Ind Med. 48(12):825-9.

Scheuchenzuber WJ, Eskew ML, Zarkower A. 1985. Effects of prolonged inhalation of silica and olivine dusts on immune functions in the mouse. Environ Res. 38(2):389-99.

Schiller-Scotland CF, Hlawa R, Gebhart J. 1994. Experimental data for total deposition in the respiratory tract of children. Toxicol. Lett. 72(1-3):137-44.

Seixas NS, Heyer NJ, Welp EA, Checkoway H. 1997. Quantification of historical dust exposures in the diatomaceous earth industry. Ann Occup Hyg. 41(5):591-604.

Sichel HS (1957). On the size distribution of airborne mine dust. J S Afr Inst Min Metal. 58(5):171-225.

Steenland K, Brown D. 1995. Silicosis among gold miners: exposure--response analyses and risk assessment. Am J Public Health. 85(10):1372-7.

Steenland K, Goldsmith DF. 1995. Silica exposure and autoimmune diseases. Am J Ind Med. 28(5):603-8.

Stratta P, Canavese C, Messuerotti A, Fenoglio I, Fubini B. 2001. Silica and renal diseases: no longer a problem in the 21st century? J Nephrol. 14(4):228-47.

Suhr H, Bang B, Moen BE. 2003. Respiratory health among quartz-exposed slate workers--a problem even today. Occup Med (Lond). 53(6):406-7.

Theriault GP, Burgess WA, DiBerardinis LJ, Peters JM. 1974. Dust exposure in the Vermont granite sheds. Arch Environ Health. 28(1):12-7.

USEPA. 1995. U.S. Environmental Protection Agency. The Use of the Benchmark Dose Method in Health Risk Assessment. EPA/630/R-94/007. Washington, DC: U.S. EPA.

USEPA. 1996. U.S. Environmental Protection Agency. Ambient Levels and Noncancer Health Effects of Inhaled Crystalline and Amorphous Silica: Health Issue Assessment. EPA/600/R-95/115. Office of Research and Development. Washington, DC: U.S. EPA.

Vallyathan V, Kang JH, Van Dyke K, Dalal NS, Castranova V. 1991. Response of alveolar macrophages to in vitro exposure to freshly fractured versus aged silica dust: the ability of Prosil 28, an organosilane material, to coat silica and reduce its biological reactivity. J Toxicol Environ Health. 33(3):303-15.

Vallyathan V, Castranova V, Pack D, Leonard S, Shumaker J, Hubbs AF, *et al*. 1995. Freshly fractured quartz inhalation leads to enhanced lung injury and inflammation. Potential role of free radicals. Am J Respir Crit Care Med. 152(3):1003-9.

Verma DK, Sebestyen A, Julian JA, Muir DC, Schmidt H, Bernholz CD, Shannon HS. 1989 Silica exposure and silicosis among Ontario hardrock miners: II. Exposure estimates. Am J Ind Med. 16(1):13-8.

Wagner G. (1995). The inexcusable persistence of silicosis. Am J Public Health. 85(10):1346-7.

Wagner WD, Fraser DA, Wright PG, Dobrogorski OJ, Stokinger HE. 1968. Experimental evaluation of the threshold limit of cristobalite--calcined diatomaceous earth. Am Ind Hyg Assoc J. 29(3):211-21.

Witschi HR, Last JA. 2001. Toxic responses of the respiratory system. In: Casarett and Doull's Toxicology. The Basic Science of Poisons. 6th Ed. Klaassen CD (ed). New York: McGraw-Hill.

World Health Organization (WHO). 1986. Recommended health-based limits in occupational exposure to selected mineral dusts (silica, coal). Geneva, Switzerland: World Health Organization, Technical Report Series 734.

X. Appendix

Particulate Levels of Interest for Exposure to Respirable Crystalline Silica Isomorphs

150 $\mu g/m^3$	Federal 24 hour PM_{10} standard
65 $\mu g/m^3$	Federal 24 hour $PM_{2.5}$ standard
50 $\mu g/m^3$	California 24 hour PM_{10} standard
50 $\mu g/m^3$	Federal PM_{10} annual standard (chronic exposure)
50 $\mu g/m^3$	8 hour TLV for quartz, cristobalite, and tridymite for workers (ACGIH Method)
50 $\mu g/m^3$	estimated workplace LOAEL for silicosis from studies by Theriault *et al.* (“)
20 $\mu g/m^3$	CA annual PM_{10} standard (chronic exposure) (arithmetic mean)
15 $\mu g/m^3$	Federal annual $PM_{2.5}$ standard (chronic exposure)
12 $\mu g/m^3$	CA annual $PM_{2.5}$ standard (chronic exposure) (arithmetic mean)
12 $\mu g/m^3$	current silica TLV adjusted to equivalent continuous exposure (50 $\mu g/m^3$ x 8 h/24 h x 5 d/7d) (ACGIH)
10 $\mu g/m^3$	TLV for silica proposed by Greaves (2000) (ACGIH)
8 $\mu g/m^3$	current silica TLV further adjusted by 46/70 years occupational exposure (ACGIH)
8 $\mu g/m^3$	estimated high-end ambient crystalline silica level in US (USEPA, 1996) (PM_{10})
6.7 $\mu g/m^3$	lower bound on 1% risk of silicosis estimated by USEPA (1996) (PM_{10})
5 $\mu g/m^3$	TLV for silica proposed by Chen *et al*. (2001) (ACGIH)
5 $\mu g/m^3$	“acceptable” ambient level for silica (10% of PM_{10}) (USEPA, 1996)
5 $\mu g/m^3$	RfC for diesel exhaust particulate, a respirable PM ($PM_{2.5}$)
3 $\mu g/m^3$	estimated average ambient exposure to crystalline silica (USEPA, 1996) (PM_{10})
3 $\mu g/m^3$	draft silica chronic REL proposed by OEHHA (ACGIH)
2.3 $\mu g/m^3$ (1.17-3.46; n=12)*	silica level during 1989 in Santa Maria, CA (urban site) (PM_{10})
0.6 $\mu g/m^3$ (0-1.44; n=16)*	silica level during 1989 in Santa Ynez, CA (rural site) (“)
0.2 $\mu g/m^3$ (0-1.15; n=18)*	silica level during 1989 in Buellton, CA (remote background)(“)

* mean, range, and number of crystalline silica measurements (Mukherji *et al*., 1993)

Valley School playground with Lane Mountain facility in the background.

Air Exposures to Particulate Matter and Silica Dust
Valley, WA
Summary of ATSDR's Exposure Investigation

In 2016 the Washington Department of Health asked ATSDR to do an exposure investigation to better characterize exposure to silica and particulate matter (PM) at the Valley School campus and in the community of Valley, WA. ATSDR responded by measuring four pollutants in air – PM_{10}, PM_4, $PM_{2.5}$ and PM_4 crystalline silica. This is a summary of the full ATSDR report.

The Valley School campus is located across the street from Lane Mountain Company, a large producer of silica sand products in the Northwest since 1961. The school district, local and state health departments, and state environmental department, have been talking about air quality concerns with the facility since 1992. Different types of air monitoring, and changes to the Lane Mountain facility and the school campus have taken place over time, see page 5 of the full report for details.

What are the community's concerns?

The Valley School District expressed concerns about the potential effect of silica dust on its students and staff to the Northeast Tri County Health District, the Washington Departments of Health and Ecology, and Lane Mountain Company in 1992 and in 2008. These organizations have been communicating about potential dust impacts for many years.

What did ATSDR do?

ATSDR staff visited Valley in 2016 to discuss concerns of the school district, residents, and Lane Mountain managers. From July through September 2016, ATSDR measured four pollutants in air – PM_{10}, PM_4, $PM_{2.5}$ and PM_4 crystalline silica – at nine locations throughout the area. We monitored air indoors and outdoors at the Valley School campus. We also monitored air outdoors in a nearby residential area, and at a more distant location west of Lane Mountain. We compared the measured concentrations with health comparison values from the World Health Organization, the U.S. Environmental Protection Agency (EPA), and the California Office of Environmental Health Hazard Assessment. When measurements were higher than health comparison values, then we compared the measured concentrations to documented health effect levels from scientific literature.

U.S. Department of Health and Human Services
Agency for Toxic Substances and Disease Registry

The Bottom Line:

- Some people who breathe PM_{10} on the Valley School campus could have health problems including respiratory symptoms and aggravation of existing lung conditions like asthma. Our main concern is for sensitive populations such as adults and children that have preexisting health conditions related to breathing.
- Depending on conditions, students and staff who breathe PM_4 crystalline silica long term on the school campus or Valley residents who breathe PM_4 crystalline silica long term in the surrounding area could be at risk for health problems like silicosis, lung cancer, and other respiratory, kidney, and autoimmune diseases.
- The general population and the sensitive population of students, staff, or residents who breathe $PM_{2.5}$ on the school campus or in the surrounding area are not at risk for health problems because the levels of $PM_{2.5}$ measured were below health comparison values. Some individuals (e.g., people with asthma or those with cardiopulmonary conditions) are unusually sensitive to changes in air quality, and may still experience transient health effects on days with poorer air quality, regardless of whether concentrations were above health comparison values.
- PM_4 crystalline silica and PM_{10} were highest on the school campus when winds were from the southwest. Winds from the southwest are more common in the spring and summer. PM_4 indoors is about four times lower than outdoors. Thus, Valley School District can limit exposure by limiting outdoor activity on campus on days when there are strong winds from the southwest.

CS306165-A July 2019

What did ATSDR find?

In Valley, PM_{10} and PM_4 crystalline silica were measured above their health comparison values, but $PM_{2.5}$ was below.

ATSDR recognizes that some individuals (e.g. people with asthma or those with other cardiopulmonary conditions) are unusually sensitive to changes in air quality, and those individuals may still experience transient health effects during days with poorer air quality, regardless of whether amounts exceed health-based health comparison values.

HEALTH HAZARD

Finding 1 – Some people who breathe PM_{10} on the Valley School campus could have health problems including respiratory symptoms, and aggravation of existing lung conditions like asthma. Our main concern is for sensitive populations.

During 2014 - 2017 at the permanent PM_{10} station, nine days measured 24-hour concentrations greater than the National Ambient Air Quality Standards for PM_{10} (150 µg/m_3), and approximately 25% of days had concentrations greater than the 24-hour World Health Organization health comparison value for PM_{10} (50 µg/m^3).

Across all sites sampled, we measured PM_{10} at amounts that could potentially harm some unusually sensitive individuals on 259 days of 1,200 days (21% of days measured). The PM_{10} amounts on 8 days could potentially harm sensitive individuals, and on 1 day, the amounts were high enough to potentially harm healthy adults.

The amounts of PM_{10} measured on Valley School Campus were highest when winds came from the southwest. These winds are more common in the spring and summer than other times of the year.

HEALTH HAZARD

Finding 2 – Breathing PM_4 crystalline silica on the campus of Valley School and in the surrounding area presents a potential long-term public health hazard to students, staff, and residents.

The amounts of PM_4 crystalline silica measured on Valley School Campus were above health comparison values at all monitoring locations on the campus of Valley School as well as at two locations in the nearby community.

The amounts of PM_4 crystalline silica measured on Valley School Campus were highest when winds came from the southwest. These winds are more common in the spring and summer than other times of the year.

The amount of PM_4 measured inside the school was approximately four times lower than the amounts of PM_4 measured outdoors.

Long term inhalation of crystalline silica particles may cause silicosis and is associated with increased risk of lung cancer. In addition, it has been associated with other respiratory diseases such as chronic obstructive pulmonary disease, bronchitis, and emphysema, as well as kidney and autoimmune diseases.

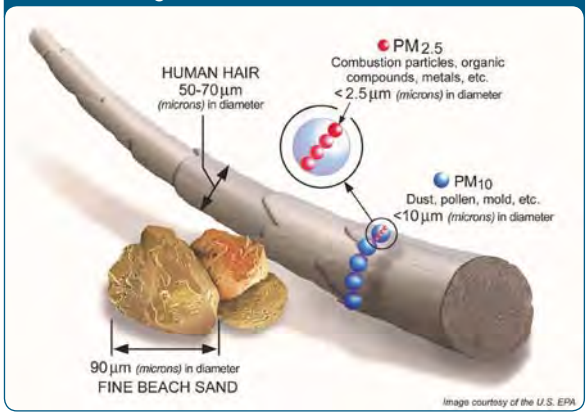
Health Comparison Values

For this EI, a health *comparison value* (CV) is the concentration of a substance – like PM_4 crystalline silica – in air that is unlikely to cause harmful health effects in people who have been exposed to it. Scientists compare measured amounts of a substance in the environment, or estimates based on measured amounts in the environment, to health comparison values.

If the amount in the environment is higher than the health comparison value, then the scientist will recommend that the substance be looked at more carefully in the health assessment process.

What is particulate matter (PM)?

Particulate matter (PM), is tiny pieces of solids or liquids that are in the air. The particles may include dust, dirt, soot, smoke, or drops of liquid. Particulate matter is described by the size of the particles in units called microns. The abbreviation is µm. In this investigation, we measured PM_{10}, PM_4, and $PM_{2.5}$. For health comparison, a human hair is 50 – 70 microns across. Image from EPA.



NO HEALTH HAZARD

Finding 3 – Breathing $PM_{2.5}$ on the campus of Valley School and in the surrounding area is not a public health hazard to the general and sensitive population of students, staff, or residents.

All 24-hour concentrations of $PM_{2.5}$ were below the World Health Organization health comparison value for $PM_{2.5}$ (25 µg/m^3). The average $PM_{2.5}$ concentration during the entire investigation was below the annual World Health Organization health comparison value at all sites but one.

Next Steps for PM_{10} and PM_4 Crystalline Silica

- Ecology and Lane Mountain agreed in November 2017 to make changes that are expected to reduce the facility's impacts on the nearby community.
- Lane Mountain should confirm that the Autumn 2018 efforts to reduce emissions of PM_{10} and PM_4 crystalline silica from their operations have resulted in reduced exposure to particulate matter and crystalline silica in the surrounding area.
- Lane Mountain should make permanent station PM_{10} data available to Valley School District in real-time to assist the District in making decisions on when to limit outdoor activity of students and staff on the school campus.
- Lane Mountain should conduct regular calibration checks and audits of the permanent PM_{10} monitor.
- Valley School District should limit outdoor activity on campus when concentrations of PM_{10} are above 50 µg/m3 in outdoor air.
- If real-time PM_{10} data are not available, Valley School District should limit outdoor activity on campus when there are strong winds from the southwest. If no wind direction measurements are available from Valley School, Lane Mountain should install a wind sock to inform decision makers about wind directions.
- If you have health concerns about this exposure talk with your doctor.

ATSDR will

- Give the report to the Valley School District, the EPA, Ecology, EI Participants, and other community members as requested.
- Meet individually with EI participants to discuss the information provided in this report and the data collected on their respective properties.
- Meet with interested stakeholders to discuss this report.
- Meet with the Valley School District and Ecology to discuss changes made through the 2017 Administrative Order issued by the Ecology to Lane Mountain Company.

If requested, ATSDR will work with the Valley School District, Ecology, and the EPA to consider options to reduce exposures in the Valley, WA.

What is crystalline silica?

Silica is the scientific name for sand. Silica is very common, it occurs naturally in our environment, and it has many uses. One form of silica, called crystalline silica, is produced by the Lane Mountain facility. People who are exposed to crystalline silica for long periods of time (typically more than 10 years) or have a heavy exposure over a short period of time may have health effects.

Air monitoring equipment on Valley School Campus.

How can I learn more about the air investigation in Valley, WA?

For more details about ATSDR's findings, see our companion report "**Ambient Airborne Exposures to Particulate Matter and Silica Dust in Valley School, Valley, WA.**"

If you have questions or comments, call ATSDR environmental health scientist, Debra Gable, at **206-553-1796**
or our toll-free number at **1-800-CDC-INFO**.

About ATSDR

The Agency for Toxic Substances and Disease Registry (ATSDR) is a federal public health agency of the U.S. Department of Health and Human Services (HHS). ATSDR works with other agencies and tribal, state, and local governments to study possible health risks in communities where people could come in contact with dangerous chemicals. For more information about ATSDR, visit our website at https://www.atsdr.cdc.gov/.

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REVIEW ARTICLE Open Access

The health effects of ultrafine particles

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Abstract

Ultrafine particles ($PM_{0.1}$), which are present in the air in large numbers, pose a health risk. They generally enter the body through the lungs but translocate to essentially all organs. Compared to fine particles ($PM_{2.5}$), they cause more pulmonary inflammation and are retained longer in the lung. Their toxicity is increased with smaller size, larger surface area, adsorbed surface material, and the physical characteristics of the particles. Exposure to $PM_{0.1}$ induces cough and worsens asthma. Metal fume fever is a systemic disease of lung inflammation most likely caused by $PM_{0.1}$. The disease is manifested by systemic symptoms hours after exposure to metal fumes, usually through welding. $PM_{0.1}$ cause systemic inflammation, endothelial dysfunction, and coagulation changes that predispose individuals to ischemic cardiovascular disease and hypertension. $PM_{0.1}$ are also linked to diabetes and cancer. $PM_{0.1}$ can travel up the olfactory nerves to the brain and cause cerebral and autonomic dysfunction. Moreover, in utero exposure increases the risk of low birthweight. Although exposure is commonly attributed to traffic exhaust, monitored students in Ghana showed the highest exposures in a home near a trash burning site, in a bedroom with burning coils employed to abate mosquitos, in a home of an adult smoker, and in home kitchens during domestic cooking. The high point-source production and rapid redistribution make incidental exposure common, confound general population studies and are compounded by the lack of global standards and national reporting. The potential for $PM_{0.1}$ to cause harm to health is great, but their precise role in many illnesses is still unknown and calls for more research.

Introduction

Air pollution can harm nearly every organ in the body[1,2], and particulate matter (PM) is the main offender. PM has been classified by particle size, which is an important factor in its health effects. PM_{10} (particles ≤10 μm in diameter), $PM_{2.5}$ (particles ≤2.5 μm in diameter), also called fine particles, and $PM_{0.1}$ (particles ≤0.1 μm in diameter), also called ultrafine particles (this term is used interchangeably with $PM_{0.1}$ in this document), have different health effects that, in part, result from how these particles navigate the small bronchioles and lung defenses. $PM_{0.1}$ are also called nanoparticles because of their size, although many authors restrict the word "nanoparticles" to the 100-nm or smaller particles produced by controlled engineering processes[3].

Ultrafine particles are dispersed atmospherically in many settings[4]. Examples are found in nature, from forest fires, ocean splashes, and viruses; combustion processes, from vehicular and power plant emissions and tobacco smoking; and synthetic sources, from toner pigment and many engineered products used for microtechnology. These particles may be formed by the coalescence of ions and gaseous molecules produced by combustion, often as acidic and basic ions or other charged species that combine to form more stable molecules or salts. This process, which usually depends on aqueous oxidation, may explain the sulfate levels of London fog of 1952[5] and the effects of humidity on the symptoms of patients with respiratory disease. Coalescing $PM_{0.1}$ are a major source of $PM_{2.5}$.

The harmful effects of the different PM categories overlap because the corresponding sizes overlap; PM_{10}, which include all smaller particles, will have similar effects to those of smaller PMs, although the effects can be distinguished by taking mass into account. PM_{10} and $PM_{2.5}$

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are measured by their mass, while $PM_{0.1}$ are measured by particle number.

Numbers of particles

A typical concentration of $PM_{0.1}$ in ambient air in rural areas is 2610 particles/cm^3, whereas a roadside concentration may be 48,180/cm^3, with a mean global concentration of 10,760/cm^3 [4]. The large numbers of $PM_{0.1}$ quickly diminish by amalgamation into larger particles and atmospheric dispersion, resulting in local concentrations or "hotspots" near traffic or sites of industrial production. It is paradoxical that $PM_{0.1}$ particle numbers decrease quickly by coalescence, yet remain airborne for extended periods and can travel to other continents. Peak concentrations of traffic emissions occur near curbsides, and these levels are often more than ten-fold higher than the background. The concentrations regress to background by ~500 m from the source[6]. High concentrations are associated with many different conditions, such as high humidity, low air movement, increased number of diesel vehicles, seasons, and traffic acceleration after stopping[3]. Improvements in fuel and engine technology and the use of catalytic converters have reduced the PM mass and carbon monoxide (CO) from automotive exhaust but have increased the number and toxicity of $PM_{0.1}$[7,8].

Occupational exposures, especially those associated with combustion or high temperature, such as welding or blast furnace operation, may be great. The highest concentrations of $PM_{0.1}$ are found in welding facilities, machine shops, basic metal industries, traffic-related occupations, and restaurants, with concentrations of $0.7–4.7 \times 10^6$, 60–450 times higher than the background levels[3].

As particle sizes decrease, the numbers increase, especially when such particles approach the size of $PM_{0.1}$. Coarse particles ($PM_{10\text{-}2.5}$) accounted for <0.1% of the total particle numbers in a study from Shenyang, China[9]. An airborne concentration of 10 μg/m^3 would require 2.4×10^6 20-nm particles, but only one 2.5-μm particle/cm^3[3,10]. With the particle number and total surface area as important parameters, the mass measurements used for PM_{10} and $PM_{2.5}$ are not useful for $PM_{0.1}$. Particle numbers are so much greater for $PM_{0.1}$ that this measurement is used to estimate $PM_{0.1}$ concentrations.

The variation in $PM_{0.1}$ number with season and location was highlighted by a European study that found that Augsburg, Helsinki, and Stockholm had mean annual concentrations of particles between 10,000 and 11,000 cm^{-3}, but winter concentrations were almost twice that much (between 10,000 and 20,000 cm^{-3}), and summer concentrations were approximately half that much (between 5000 and 6000 particles cm^{-3}. Rome and Barcelona, which have greater air pollution, had average annual concentrations of $PM_{0.1}$ of more than 43,000 and 39,000 cm^{-3}, respectively, but winter concentrations of 100,000 cm^{-3}. This study also reported diurnal and weekday variations, with daily peak levels between 7 a.m. and 10 a.m. in most places. Sunday had ~2/3 of the weekday concentrations[11]. The variation and fluctuating nature of the exposure makes monitoring, gauging their health effects, and setting standards difficult.

The hourly average exposure over a year of $PM_{0.1}$ differs from that of $PM_{2.5}$ in that the distribution of $PM_{0.1}$ is more skewed and widely spread owing to their more rapid reduction and dispersal. The greater diurnal trend for $PM_{0.1}$ results from the variation in vehicular emissions. $PM_{0.1}$ and $PM_{2.5}$ are not well correlated; the ratio of the particle number-to-mass ratio was found to be highest at roadside sites (indicating a prominence of $PM_{0.1}$) and lowest in polluted cities (indicating a prominence of $PM_{2.5}$). Regulating $PM_{2.5}$ may not significantly reduce $PM_{0.1}$[12].

Absorption and retention of $PM_{0.1}$

Most inhaled particles of 10 μm or larger in aerodynamic diameter impact the nasopharyngeal membranes. Inhaled particles of 5–10 μm usually land on the airways and are normally removed by alveolar macrophages and lung lymphatics[1]. Particles in the range of 1–2.5 μm usually make their way to the terminal bronchiole, the site of greatest accumulation and tissue destruction, as commonly seen in centrilobular emphysema. Particles <1 μm stay airborne longer and easily gain access to alveoli. Although most PM sizes can be engulfed by cells, $PM_{0.1}$ translocate transcellularly across alveolar epithelial cells by diffusion through the lipid bilayer of the cell walls[13]. It is not just phagocytic cells that pick up material. All cells absorb cellular fragments of senescent, damaged, or normal cells and exchange and recycle molecular material[14]. The cellular fragments (sometimes termed extracellular vesicles) could easily harbor $PM_{0.1}$.

In an in vitro model, investigators showed that positively charged $PM_{0.1}$ penetrated cells 20–40 times more than negatively charged particles[13]. Although this result may be specific to the cells tested, it points out the importance of surface charge. An important property of $PM_{0.1}$ is the large surface-area to mass ratio that allows the particles to carry large amounts of adsorbed materials per unit mass. The large variety of compounds that attach to these particles is likely to be a major cause of their toxicity, but the great variation of adsorbed material makes it more difficult to link $PM_{0.1}$ to specific conditions. Other properties, such as aspect ratio, charge, surface reactivity, solubility, hydrophobicity or polarity, agglomeration state, and the ability to interact with biologic tissue and generate reactive oxygen species, are important determinants of toxicity.

$PM_{0.1}$ that enter alveoli can be retained in surfactant[15], thus sidestepping the mucociliary escalator clearance mechanisms. The retention half-life of titanium dioxide (TiO_2) particles of the identical crystalline structure, deposition burden, and conditions in animal lungs was reported to be 170 days for 250-nm particles and 500 days for 20-nm particles[16]. The finer particles caused stronger and more persistent inflammation, with more type II cell proliferation and macrophage impairment and early interstitial fibrotic foci. The small particles also moved into the lung interstitium and periphery more than the larger ones[16]. TiO_2, which is considered a safe additive to foods, toothpaste, lotions, and many other household products, appears to have many major toxic effects when its particles are in the $PM_{0.1}$ range[17].

Health effects

The first interaction site for $PM_{0.1}$ is the lung. The surface area of the lung has been estimated to be more than 100 m^2 [18], but this is commonly estimated by measuring linear intercepts with a 1-μm probe by light microscopy. The estimate of the surface area becomes larger as the sampling probe becomes smaller, accounting for the additional surface area of an irregular surface. The surface area of the lung to a nanoprobe of $PM_{0.1}$ would be orders of magnitude greater than the light microscopy estimates, something Weibel referred to as the "Coast of Wales" effect[19].

$PM_{0.1}$, along with their toxic baggage, easily reach the large surface of the lung. They subsequently gain access to other organs through the lung vasculature, either through mobile cells or freely in the vasculature and lymph to directly harm distal organs. Another mechanism by which $PM_{0.1}$ cause harm is lung inflammation and the subsequent spread of inflammatory mediators to distal organs. This is considered the main cause of systemic toxicity for larger PM, which are less able to directly access other organs. In addition to having better access, $PM_{0.1}$ have more toxicity in cellular and animal models[20]. For example, low-solubility and low-toxicity $PM_{0.1}$ cause more inflammation in rat lungs than $PM_{2.5}$ of the same material[21].

Other explanations for the increased toxicity of $PM_{0.1}$ include the fact that many smaller particles may stress alveolar macrophages more than fewer large ones. This could also explain why clearance is dependent on particle size. Inflammation in response to noxious material on the surface of particles and interactions with the cell surfaces of host tissues would also be greater with ultrafine particles[22]. Another important point when considering human population studies is that $PM_{2.5}$ have more immediate effects, while $PM_{0.1}$ have more delayed effects and a greater influence on mortality[23]. Most epidemiologic studies of acute exposure to $PM_{0.1}$ take into account a lag time between exposure and symptoms of 1–5 days to account for the delayed effect.

Compared to data on $PM_{2.5}$ and PM_{10}, there is a paucity of information on the long-term health effects of $PM_{0.1}$. A major reason for this is the lack of international standards and national reporting. Although $PM_{0.1}$ can be readily measured in the atmosphere, the measuring instruments are not standardized, which means methods and protocols vary[24]. Developing standards is difficult for populations because of the variable nature of personal exposures and the silent nature of the effects of air pollution, particularly $PM_{0.1}$. Neither clinicians nor the public generally consider air pollution as a cause of a specific illness. Many exposures, such as to office printers, would go unnoticed by most workers.

The study of $PM_{0.1}$ has been aided by engineered nanoparticles. Ultrafine particles can be manufactured with a high degree of accuracy with regard to their size, shape, and composition. The biological effects of these factors can then be studied more precisely. Engineered nanoparticles can be applied to cells and tissues and given as a challenge to animals and human volunteers. For example, high doses of fibrous and tubular nanostructures can result in fibrotic lung reactions and an increased risk of carcinogenesis[25]. Single-walled nanotubes can persist deep in the lung and induce inflammatory and fibrotic reactions[26]. Many other studies have shown various intracellular effects depending on the species studied and the nature of the nanoparticle[25].

Diseases

A review of all diseases associated with $PM_{0.1}$ is beyond the scope of this paper, but the following discussion highlights major associations. Air pollution and its effects on different organ systems have recently been reviewed[1,2].

Respiratory

The respiratory system is usually the first line of entry into the body for air pollution, but ingested $PM_{0.1}$ pass through the gastrointestinal tract and can stimulate immune responses in animals and human colonic biopsies. In a Western diet, more than 10^{12} ultrafine particles are ingested daily by a single person[27]. Intact skin is generally considered protective. Short-term exposure to $PM_{0.1}$ with a high content of polycyclic aromatic hydrocarbons increases 8-hydroxyl-2-deoxyguanosine, a byproduct of DNA oxidative damage, in children with eczema but not in those without eczema[28], although particles of zinc and TiO used in topical skin care have been shown to penetrate intact skin, especially in watery or oily vehicles[29].

Metal fume fever and polymer fume fever

Metal fume fever is an example of a disease most likely caused by $PM_{0.1}$. $PM_{0.1}$ are produced by welding. Inhaling

the small particles generates a great amount of reactive oxygen species, leading to inflammation[30]. Metal fume fever usually presents with malaise, fever, chills, arthralgias, and myalgias 4–8 h after exposure to metal fumes, usually through welding. Chest radiographs are inconsistent, and the syndrome generally abates without treatment. Although zinc oxide (ZnO) has been incriminated, other metals, namely, copper, magnesium, and cadmium, have also been identified. A similar syndrome, polymer fume fever, has been associated with inhalation of fluorinated polymer products, such as polytetrafluoroethylene (Teflon®). Heated polytetrafluoroethylene contains ultrafine particles (median diameter 26 nm) that are toxic to rats, causing hemorrhagic pulmonary inflammation and death with high-dose exposure and decreased function with low-dose exposure[31]. Short-term, high-level exposure to $PM_{0.1}$, such as through diesel exhaust, also causes lung inflammation[32].

The respiratory mucociliary apparatus is a major tool to clear inhaled particles. Although ultrafine particles can be trapped in the mucous layer, its role in clearance is far less than that for PM of greater density, which impacts the airway. Many lung conditions impair mucociliary function. Ciliary dysfunction is common in smokers and those with respiratory tract infections and may account for the greater vulnerability of these individuals to air pollution. Bronchospasm and cough, which are part of asthma, are common reactions to inhaled irritants, which may account for the increased sensitivity in persons with asthma.

Particulate air pollution is a well-known cause of exacerbations and mortality in persons with chronic obstructive pulmonary disease (COPD), but the role of $PM_{0.1}$ is unclear. A study from Scotland did not find $PM_{0.1}$ to be more harmful than PM_{10} [33], but other studies have reported that indoor biological $PM_{0.1}$ in the form of bacterial extracellular vesicles do cause inflammation and emphysema[34].

Ambient exposure to $PM_{0.1}$ is associated with cough, reduced peak expiratory flow[35], and the increased use of medicines[36] and hospital admissions for persons with asthma[37]. Clinical visits for respiratory illness are associated with increased levels of $PM_{0.1}$[38]. Although most studies have found an increase in asthma symptoms, a study of more than a million adult residents of Toronto did not find evidence for an association between long-term exposure to $PM_{0.1}$ and respiratory disease after adjusting for $PM_{2.5}$, NO_2, and other covariates[39]. An Australian controlled study also did not find particle number to be independently associated with respiratory symptoms, asthma diagnosis, or lung function, although $PM_{0.1}$ was associated with an increase in inflammatory markers in atopic participants[40].

Cardiovascular

Many studies have shown that PM causes systemic inflammation and coagulation changes predisposing to ischemic cardiovascular disease, as measured by elevated C-reactive protein (CRP), circulating polymorphonuclear leukocytes, platelets, fibrinogen, plasma viscosity and other markers. PM promotes endothelial dysfunction, vascular inflammation, and atherosclerosis[1]. Past studies have attributed this effect mainly to $PM_{2.5}$, but a growing body of literature shows that $PM_{0.1}$ have a major role in essentially all of these factors[41-43]. In fact, most studies show a far greater effect for $PM_{0.1}$. $PM_{0.1}$ also cause increased heart rate variability, loss of sympathovagal balance, and altered inflammatory and hemostatic function in exposed humans[44].

Even brief exposures to $PM_{0.1}$ can cause cardiac effects. In middle-aged individuals with metabolic syndrome, exposure to $PM_{0.1}$ for 2 h caused electrocardiographic changes, a decrease in blood plasminogen and thrombomodulin and an increase in CRP and serum amyloid A[45].

Many studies have shown an association between chronic exposure to $PM_{0.1}$ and heart disease. A prospective study of 33,831 Dutch residents found that long-term exposure to $PM_{0.1}$ (measured by land use regression) was associated with an increased risk for cardiovascular disease, myocardial infarction, and heart failure[46]. In adults living in Toronto from 1996 to 2012, an increase in $PM_{0.1}$ exposure was associated with an increased incidence of heart failure and acute myocardial infarction. Adjustment for $PM_{2.5}$ and NO_2 did not change these associations, although NO_2 was also independently associated with increased heart failure incidence[47]. Mobile neighborhood monitoring found the annual average particle number exposures to be associated with stroke, ischemic heart disease, and hypertension[48]. Other studies have also found increased ischemic and thrombotic stroke with $PM_{0.1}$ exposure[49] and increased blood pressure and worse microvascular function with $PM_{0.1}$ but not with $PM_{2.5}$ and PM_{10}[43,50].

Particle size has been correlated with total and cardiovascular mortality, with the correlation becoming stronger as the particle size decreases. PM <0.50 μm had the highest correlation[9,51]. No association was found for mass concentrations ($PM_{2.5}$ and larger)[51].

Particle numbers are associated with cardiovascular disease-related emergency department visits, with a lag of 4–10 days; 10–50 nm particles mainly account for this finding. $PM_{0.1}$ were reported to account for more than 7% of emergency department visits[52]. The strongest correlate of immediate effect (within 2 days) was found with 30–100-nm particles, despite a small mass concentration. The immediate effect related to mass concentration was with the 1–5 μm particles, which had a similar delayed effect to the $PM_{0.1}$ number[52].

A study of more than 100,000 women in California found that mortality from ischemic heart disease was more strongly associated with $PM_{0.1}$ than with $PM_{2.5}$[53]. Repeated biweekly submaximal exercise tests on adult subjects with stable coronary heart disease showed that $PM_{0.1}$ were associated with electrocardiographic ST-segment depression of >0.1 mV. The researchers found that the $PM_{0.1}$ effect was independent of $PM_{2.5}$. NO_2 and CO were also associated with a risk for ST-segment depression, but coarse particles ($PM_{10-2.5}$) were not[54].

However, a study from Denmark of 6515 airport workers who were exposed to long-term $PM_{0.1}$ found no correlation with ischemic heart disease or cerebrovascular disease compared to the corresponding measures in a similar group of unskilled workers[55].

Central nervous system

Many articles have been published on the brain or neural effects of PM, and there is a great deal of animal work on the mechanisms by which $PM_{0.1}$ affect the brain and its development. Translocated $PM_{0.1}$ can be found in the brain within 4–24 h after inhalation. Nasal $PM_{0.1}$ can travel up the olfactory nerves to the brain. Animals exposed to aerosols of $PM_{0.1}$ have the greatest brain uptake in the olfactory bulb, even 7 days after exposure. In an animal inhalation study, up to 20% of the $PM_{0.1}$ deposited on the olfactory mucosa moved to the olfactory bulb[56]. This pathway, which could circumvent the blood brain barrier, may be even more direct in humans[57]. $PM_{0.1}$ not only translocate and directly damage neural tissue but also affect autonomic function. Exposure to $PM_{0.1}$ increases sympathetic nervous system activity by decreasing norepinephrine clearance, a feature that is increased with concurrent ozone (O_3) exposure[58].

Considerable animal research has been carried out on the effect of $PM_{0.1}$ on brain development. Postnatally, $PM_{0.1}$-exposed animals show short-term memory impairment, cortical and hippocampal changes, which raise the potential for excitotoxicity, and long-term glial activation, which is associated with a wide range of behavioral and other neurologic effects[59]. Pregnant mice administered carbon black nanoparticles intranasally gave birth to offspring with a dose-dependent, long-term activation of astrocytes. Many mRNA level changes associated with angiogenesis, cell migration, proliferation, chemotaxis, and growth factors caused the authors to speculate that this exposure could have wide-ranging implications for health in later life[60]. Other animal studies have shown that $PM_{0.1}$ affects emotional behavior, learning capability, neurotransmission, spontaneous motor activity, and avoidance of performance[61].

In humans, regular exercise has been shown to improve brain cognition and memory. One mechanism for this effect may be by upregulating brain-derived neurotrophic factor (BDNF). BDNF was measured in subjects before and after cycling for ~20 min near major traffic and, on another occasion, in a room with air filtered to remove $PM_{0.1}$ as well larger PM. The average $PM_{0.1}$ was 28,180 particles/cm^3 near the road and 496 particles/cm^3 in the air-filtered room. Serum BDNF concentration rose after cycling in the air-filtered room but not after cycling near the major traffic route[62].

Children

Children are more vulnerable to the health effects of air pollution, and these effects may begin with in utero exposure and have lifelong consequences[1]. Exposure of pregnant women to $PM_{0.1}$ increases the risk of low birthweight, especially in those living within 50 m of heavy traffic[63].

A study from Ghana monitored the personal exposure of 61 junior high school students 24 h per day for 10 weeks. The exposures varied greatly depending on place of residence and the type of activities in which the students were engaged. The highest exposures were in a home near a trash burning site, in a bedroom with burning coils employed to abate mosquitos, in a home with an adult smoker, and in home kitchens during domestic cooking[64]. A similar study conducted in the Pearl River delta in China that monitored school children (aged 9–13) for 2 weeks found that the highest exposures were indoors, associated with smoking adults and the use of mosquito repellent incense[65]. These studies show that the sources of $PM_{0.1}$ are not always predictable and are not only related to vehicular traffic. Microenvironments and chance exposure have important implications for epidemiologic studies.

A major source of $PM_{0.1}$ for children living in rural areas can be exhaust from school buses, especially if they are older diesel-powered vehicles. Filtration of the inside air in school buses has been shown to significantly decrease the number of in-cabin ultrafine particles[66]. Exposure also occurs when children are in the playground and standing outdoors while waiting to enter the bus. Idling engines produce more $PM_{0.1}$ than driving does[67]. These exposures can be reduced by anti-idling policies.

Electronic cigarettes are now commonly used by children and are a source of inhaled $PM_{0.1}$[68]. Electronic cigarettes use heated vehicles (usually propylene glycol and glycerol) to deliver microaerosols to the lungs. They also deliver $PM_{0.1}$ to the brain along with nicotine. High-resistance coils in electronic cigarettes and increased glycerol generate larger particles, and higher coil temperatures generate smaller particles. Particles produced by the electronic cigarettes are deposited in alveoli[69].

Diabetes

Air pollution affects many metabolic functions and has been associated with diabetes and other metabolic

illnesses. Exposure to $PM_{0.1}$ and NO_2 from traffic-related sources increases the risk for hypertension and diabetes[70]. A single 2-h inhalation of elemental carbon $PM_{0.1}$ had an effect on heart variability in diabetic subjects that lasted for hours[71]. $PM_{0.1}$ cause autonomic dysfunction[72] and affect glucose tolerance in patients with diabetes[73].

Cancer

Many studies in animals and human cells have shown that PM is mutagenic and tumorigenic[61], and it appears that the finer the particle size, the greater the mutagenic potential[74]. $PM_{0.1}$ from the atmosphere and roadside contain many mutagenic particles[75]. $PM_{0.1}$ have been shown to produce tumors in rats. The strongest tumorigenic factor was the total surface area of the retained particles, although the dose, particle type, and duration of exposure were also important. Smaller aggregated ultrafine TiO_2-induced lung tumors in rats much more than larger sized TiO_2[22]. The carcinogenic properties of nanoparticles are related to their aspect ratio and rigidness[76].

Conclusions

The potential for $PM_{0.1}$ to cause harm to health is great, but their precise role in many illnesses is still unknown. Their high point-source production and rapid redistribution make incidental exposure common for the general population and confound general population studies. This has, no doubt, contributed to the lack of global standards and national reporting. The absence of standards and reporting may account for the dearth of standardized measurements, instruments, and protocols. Air pollution is a silent epidemic[77], and $PM_{0.1}$ may be the quietest of the pollutants. $PM_{0.1}$ vary greatly with the toxins they adsorb, adding complexity to public research. In many ways, the study of $PM_{0.1}$ is at the frontier of air pollution research. As they are better understood, these particles should be more easily controlled. $PM_{0.1}$, as with other air pollution, are avoidable and correctable health risks. Halting or reducing pollution should promptly result in improved health status[78]. Undoubtedly, more research is needed.

Conflict of interest

The author declares no conflict of interest.

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References

1. Schraufnagel, D. E. et al. Air pollution and noncommunicable diseases: a review by the Forum of International Respiratory Societies' Environmental Committee, Part 1: the damaging effects of air pollution. *Chest* **155**, 409–416 (2019).
2. Schraufnagel, D. E. et al. Air pollution and noncommunicable diseases: a review by the Forum of International Respiratory Societies' Environmental Committee, Part 2: air pollution and organ systems. *Chest* **155**, 417–426 (2019).
3. Li, N. et al. A work group report on ultrafine particles (American Academy of Allergy, Asthma & Immunology): why ambient ultrafine and engineered nanoparticles should receive special attention for possible adverse health outcomes in human subjects. *J. Allergy Clin. Immunol.* **138**, 386–396 (2016).
4. Morawska, L., Ristovski, Z., Jayaratne, R., Keogh, D. U. & Ling, X. Ambient nano and ultrafine particles from motor vehicle emissions: characteristics, ambient processing and implications on human exposure. *Atmos. Environ.* **42**, 8113–8138 (2008).
5. Wang, G. et al. Persistent sulfate formation from London Fog to Chinese haze. *Proc. Natl Acad. Sci. USA* **113**, 13630–13635 (2016).
6. Karner, A. A., Eisinger, D. S. & Niemeier, D. A. Near-roadway air quality: synthesizing the findings from real-world data. *Environ. Sci. Technol.* **44**, 5334–5344 (2010).
7. Frank, B., Schuster, M. E., Schlogl, R. & Su, D. S. Emission of highly activated soot particulate—the other side of the coin with modern diesel engines. *Angew. Chem. Int. Ed. Engl.* **52**, 2673–2677 (2013).
8. Park, B. et al. Hazard and risk assessment of a nanoparticulate cerium oxide-based diesel fuel additive—a case study. *Inhal. Toxicol.* **20**, 547–566 (2008).
9. Meng, X. et al. Size-fractionated particle number concentrations and daily mortality in a Chinese city. *Environ. Health Perspect.* **121**, 1174–1178 (2013).
10. Frampton, M. W. Systemic and cardiovascular effects of airway injury and inflammation: ultrafine particle exposure in humans. *Environ. Health Perspect.* **109**(Suppl. 4), 529–532 (2001).
11. Aalto, P. et al. Aerosol particle number concentration measurements in five European cities using TSI-3022 condensation particle counter over a three-year period during health effects of air pollution on susceptible subpopulations. *J. Air Waste Manag. Assoc. (1995)* **55**, 1064–1076 (2005).
12. de Jesus, A. L. et al. Ultrafine particles and PM2.5 in the air of cities around the world: are they representative of each other? *Environ. Int.* **129**, 118–135 (2019).
13. Yacobi, N. R. et al. Mechanisms of alveolar epithelial translocation of a defined population of nanoparticles. *Am. J. Respir. Cell Mol. Biol.* **42**, 604–614 (2010).
14. van Niel, G., D'Angelo, G. & Raposo, G. Shedding light on the cell biology of extracellular vesicles. *Nat. Rev. Mol. Cell Biol.* **19**, 213–228 (2018).
15. Moller, W. et al. Deposition, retention, and translocation of ultrafine particles from the central airways and lung periphery. *Am. J. Respir. Crit. Care Med* **177**, 426–432 (2008).
16. Oberdorster, G., Ferin, J. & Lehnert, B. E. Correlation between particle size, in vivo particle persistence, and lung injury. *Environ. Health Perspect.* **102**(Suppl. 5), 173–179 (1994).
17. Shi, H., Magaye, R., Castranova, V. & Zhao, J. Titanium dioxide nanoparticles: a review of current toxicological data. *Part. Fibre Toxicol.* **10**, 15 (2013).
18. Weibel, E. W. *The Pathway for Oxygen*, Vol. 340 (Harvard University Press, 1984).
19. Weiber, E. W. *Sterologic Methods: Practical Methods for Biological Morphometry* (Academic Press, 1979).
20. Cassee, F. R. et al. Particle size-dependent total mass deposition in lungs determines inhalation toxicity of cadmium chloride aerosols in rats. Application of a multiple path dosimetry model. *Arch. Toxicol.* **76**, 277–286 (2002).
21. Donaldson, K. et al. The pulmonary toxicology of ultrafine particles. *J. Aerosol Med.* **15**, 213–220 (2002).
22. Oberdorster, G. Significance of particle parameters in the evaluation of exposure–dose–response relationships of inhaled particles. *Inhal. Toxicol.* **8** (Suppl.), 73–89 (1996).
23. Ibald-Mulli, A., Wichmann, H. E., Kreyling, W. & Peters, A. Epidemiological evidence on health effects of ultrafine particles. *J. Aerosol Med.* **15**, 189–201 (2002).
24. Viitanen, A. K., Uuksulainen, S., Koivisto, A. J., Hameri, K. & Kauppinen, T. Workplace measurements of ultrafine particles—a literature review. *Ann. Work Expo. Health* **61**, 749–758 (2017).
25. Oberdorster, G. et al. Principles for characterizing the potential human health effects from exposure to nanomaterials: elements of a screening strategy. *Part. Fibre Toxicol.* **2**, 8 (2005).
26. Shvedova, A. A. et al. Unusual inflammatory and fibrogenic pulmonary responses to single-walled carbon nanotubes in mice. *Am. J. Physiol. Lung Cell. Mol. Physiol.* **289**, L698–L708 (2005).
27. Powell, J. J. et al. Immune potentiation of ultrafine dietary particles in normal subjects and patients with inflammatory bowel disease. *J. Autoimmun.* **14**, 99–105 (2000).

28. Song, S. et al. Exposure to ambient ultrafine particles and urinary 8-hydroxyl-2-deoxyguanosine in children with and without eczema. *Sci. Total Environ.* **458-460**, 408–413 (2013).
29. Lansdown, A. B. & Taylor, A. Zinc and titanium oxides: promising UV-absorbers but what influence do they have on the intact skin? *Int. J. Cosmet. Sci.* **19**, 167–172 (1997).
30. Chang, C., Demokritou, P., Shafer, M. & Christiani, D. Physicochemical and toxicological characteristics of welding fume derived particles generated from real time welding processes. *Environ. Sci. Process. Impacts* **15**, 214–224 (2013).
31. Oberdorster, G., Gelein, R. M., Ferin, J. & Weiss, B. Association of particulate air pollution and acute mortality: involvement of ultrafine particles? *Inhal. Toxicol.* **7**, 111–124 (1995).
32. Nordenhall, C. et al. Airway inflammation following exposure to diesel exhaust: a study of time kinetics using induced sputum. *Eur. Respir. J.* **15**, 1046–1051 (2000).
33. Osunsanya, T., Prescott, G. & Seaton, A. Acute respiratory effects of particles: mass or number? *Occup. Environ. Med.* **58**, 154–159 (2001).
34. Yang, J., Kim, Y. K., Kang, T. S., Jee, Y. K. & Kim, Y. Y. Importance of indoor dust biological ultrafine particles in the pathogenesis of chronic inflammatory lung diseases. *Environ. Health Toxicol.* **32**, e2017021 (2017).
35. Peters, A., Wichmann, H. E., Tuch, T., Heinrich, J. & Heyder, J. Respiratory effects are associated with the number of ultrafine particles. *Am. J. Respir. Crit. Care Med.* **155**, 1376–1383 (1997).
36. von Klot, S. et al. Increased asthma medication use in association with ambient fine and ultrafine particles. *Eur. Respir. J.* **20**, 691–702 (2002).
37. Andersen, Z. J. et al. Size distribution and total number concentration of ultrafine and accumulation mode particles and hospital admissions in children and the elderly in Copenhagen, Denmark. *Occup. Environ. Med.* **65**, 458–466 (2008).
38. Diaz-Robles, L. A. et al. Health risks caused by short term exposure to ultrafine particles generated by residential wood combustion: a case study of Temuco, Chile. *Environ. Int.* **66**, 174–181 (2014).
39. Weichenthal, S. et al. Long-term exposure to ambient ultrafine particles and respiratory disease incidence in in Toronto, Canada: a cohort study. *Environ. Health* **16**, 64 (2017).
40. Clifford, S. et al. Effects of exposure to ambient ultrafine particles on respiratory health and systemic inflammation in children. *Environ. Int.* **114**, 167–180 (2018).
41. Duffin, R., Mills, N. L. & Donaldson, K. Nanoparticles—a thoracic toxicology perspective. *Yonsei Med. J.* **48**, 561–572 (2007).
42. Hildebrandt, K. et al. Short-term effects of air pollution: a panel study of blood markers in patients with chronic pulmonary disease. *Part. Fibre Toxicol.* **6**, 25 (2009).
43. Olsen, Y. et al. Vascular and lung function related to ultrafine and fine particles exposure assessed by personal and indoor monitoring: a cross-sectional study. *Environ. Health* **13**, 112 (2014).
44. Samet, J. M. et al. Concentrated ambient ultrafine particle exposure induces cardiac changes in young healthy volunteers. *Am. J. Respir. Crit. Care Med* **179**, 1034–1042 (2009).
45. Devlin, R. B. et al. Controlled exposure of humans with metabolic syndrome to concentrated ultrafine ambient particulate matter causes cardiovascular effects. *Toxicol. Sci.* **140**, 61–72 (2014).
46. Downward, G. S. et al. Long-term exposure to ultrafine particles and incidence of cardiovascular and cerebrovascular disease in a prospective study of a Dutch Cohort. *Environ. Health Perspect.* **126**, 127007 (2018).
47. Bai, L. et al. Associations of long-term exposure to ultrafine particles and nitrogen dioxide with increased incidence of congestive heart failure and acute myocardial infarction. *Am. J. Epidemiol.* **188**, 151–159 (2019).
48. Li, Y. et al. Association of long-term near-highway exposure to ultrafine particles with cardiovascular diseases, diabetes and hypertension. *Int. J. Environ. Res. Public Health* **14**, https://doi.org/10.3390/ijerph14050461 (2017).
49. Andersen, Z. J. et al. Association between short-term exposure to ultrafine particles and hospital admissions for stroke in Copenhagen, Denmark. *Eur. Heart J.* **31**, 2034–2040 (2010).
50. Pieters, N. et al. Blood pressure and same-day exposure to air pollution at school: associations with nano-sized to coarse PM in children. *Environ. Health Perspect.* **123**, 737–742 (2015).
51. Stolzel, M. et al. Daily mortality and particulate matter in different size classes in Erfurt, Germany. *J. Expo. Sci. Environ. Epidemiol.* **17**, 458–467 (2007).
52. Liu, L. et al. Size-fractioned particulate air pollution and cardiovascular emergency room visits in Beijing, China. *Environ. Res.* **121**, 52–63 (2013).
53. Ostro, B. et al. Associations of mortality with long-term exposures to fine and ultrafine particles, species and sources: results from the California Teachers Study Cohort. *Environ. Health Perspect.* **123**, 549–556 (2015).
54. Pekkanen, J. et al. Particulate air pollution and risk of ST-segment depression during repeated submaximal exercise tests among subjects with coronary heart disease: the Exposure and Risk Assessment for Fine and Ultrafine Particles in Ambient Air (ULTRA) study. *Circulation* **106**, 933–938 (2002).
55. Moller, K. L. et al. Cardiovascular disease and long-term occupational exposure to ultrafine particles: a cohort study of airport workers. *Int. J. Hyg. Environ. Health* **223**, 214–219 (2020).
56. Oberdorster, G. et al. Translocation of inhaled ultrafine particles to the brain. *Inhal. Toxicol.* **16**, 437–445 (2004).
57. Tian, L. et al. Correlation of regional deposition dosage for inhaled nanoparticles in human and rat olfactory. *Part. Fibre Toxicol.* **16**, 6 (2019).
58. Heusser, K. et al. Ultrafine particles and ozone perturb norepinephrine clearance rather than centrally generated sympathetic activity in humans. *Sci. Rep.* **9**, 3641 (2019).
59. Allen, J. L. et al. Developmental exposure to concentrated ambient ultrafine particulate matter air pollution in mice results in persistent and sex-dependent behavioral neurotoxicity and glial activation. *Toxicol. Sci.* **140**, 160–178 (2014).
60. Onoda, A., Takeda, K. & Umezawa, M. Dose-dependent induction of astrocyte activation and reactive astrogliosis in mouse brain following maternal exposure to carbon black nanoparticle. *Part. Fibre Toxicol.* **14**, 4 (2017).
61. Kumar, S., Verma, M. K. & Srivastava, A. K. Ultrafine particles in urban ambient air and their health perspectives. *Rev. Environ. Health* **28**, 117–128 (2013).
62. Bos, I. et al. No exercise-induced increase in serum BDNF after cycling near a major traffic road. *Neurosci. Lett.* **500**, 129–132 (2011).
63. Laurent, O. et al. Sources and contents of air pollution affecting term low birth weight in Los Angeles County, California, 2001–2008. *Environ. Res.* **134**, 488–495 (2014).
64. Nyarku, M. et al. Schoolchildren's personal exposure to ultrafine particles in and near Accra, Ghana. *Environ. Int.* **133**, 105223 (2019).
65. Mazaheri, M. et al. Characteristics of school children's personal exposure to ultrafine particles in Heshan, Pearl River Delta, China—a pilot study. *Environ. Int.* **132**, 105134 (2019).
66. Lee, E. S., Fung, C. C. & Zhu, Y. Evaluation of a high efficiency cabin air (HECA) filtration system for reducing particulate pollutants inside school buses. *Environ. Sci. Technol.* **49**, 3358–3365 (2015).
67. Londahl, J. et al. Experimental determination of the respiratory tract deposition of diesel combustion particles in patients with chronic obstructive pulmonary disease. *Part. Fibre Toxicol.* **9**, 30 (2012).
68. Glantz, S. A. & Bareham, D. W. E-cigarettes: use, effects on smoking, risks, and policy implications. *Annu. Rev. Public Health* **39**, 215–235 (2018).
69. Lechasseur, A. et al. Variations in coil temperature/power and e-liquid constituents change size and lung deposition of particles emitted by an electronic cigarette. *Physiol. Rep.* **7**, e14093 (2019).
70. Bai, L. et al. Exposure to ambient ultrafine particles and nitrogen dioxide and incident hypertension and diabetes. *Epidemiology* **29**, 323–332 (2018).
71. Vora, R. et al. Inhalation of ultrafine carbon particles alters heart rate and heart rate variability in people with type 2 diabetes. *Part. Fibre Toxicol.* **11**, 31 (2014).
72. Peters, A. et al. Elevated particle number concentrations induce immediate changes in heart rate variability: a panel study in individuals with impaired glucose metabolism or diabetes. *Part. Fibre Toxicol.* **12**, 7 (2015).
73. Sun, Y. et al. Size-fractioned ultrafine particles and black carbon associated with autonomic dysfunction in subjects with diabetes or impaired glucose tolerance in Shanghai, China. *Part. Fibre Toxicol.* **12**, 8 (2015).
74. Pagano, P., De Zaiacomo, T., Scarcella, E., Bruni, S. & Calamosca, M. Mutagenic activity of total and particle-sized fractions of urban particulate matter. *Environ. Sci. Technol.* **30**, 3512–3516 (1996).
75. Kawanaka, Y., Matsumoto, E., Sakamoto, K. & Yun, S. J. Estimation of the contribution of ultrafine particles to lung deposition of particle-bound mutagens in the atmosphere. *Sci. Total Environ.* **409**, 1033–1038 (2011).
76. Stanton, M. F. et al. Relation of particle dimension to carcinogenicity in amphibole asbestoses and other fibrous minerals. *J. Natl. Cancer Inst.* **67**, 965–975 (1981).
77. World Health Organization. *Infographic: Air Pollution—the Silent Killer* http://www.euro.who.int/en/health-topics/environment-and-health/air-quality/news/news/2018/5/over-half-a-million-premature-deaths-annually-in-the-european-region-attributable-to-household-and-ambient-air-pollution/infographic-air-pollution-the-silent-killer (2019).
78. Schraufnagel, D. E. et al. Health benefits of air pollution reduction. *Ann. Am. Thorac. Soc.* **16**, 1478–1487 (2019).

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Prevalence of wide area impacts downwind of freeways under pre-sunrise stable atmospheric conditions

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Abstract

In urban areas, a large fraction of the population typically lives within 1.5–2 km of a freeway. In an earlier study, Hu et al. (2009) showed that in the pre-sunrise hours, with stable, nocturnal meteorological conditions, the vehicle-related pollution from a freeway in southern California extended more than 2 km downwind. This was in striking contrast to earlier studies in the U.S. and Australia showing that during the daytime pollutant concentrations extended no more than ~300 m downwind of major roadways. The present study was designed to explore the variability of the large downwind pollutant impact zone observed by Hu et al. (2009) in the pre-sunrise hours on a wider geographic scale. An instrumented mobile platform was employed to measure vehicle-related pollutant concentrations on transects running upwind and downwind of four freeways in the coastal, central and eastern areas of the California South Coast Air Basin (SoCAB) during the pre-sunrise period (04:30–06:30). Extended freeway plumes were observed for ultrafine

particles (UFP), nitric oxide and particle-bound polycyclic aromatic hydrocarbons (PAH) at all four sites during stable pre-sunrise periods. Plume lengths were measured to be ~2 km or more with a dilution rate coefficient about a factor of ten lower than commonly observed for daytime. An average of 39 and 19% (±8–9%) of freeway plume peak concentrations of UFP remained 500 and 1500 m downwind, respectively, from the freeways for the four transects studied here. Because a large fraction of UFP and other vehicle related pollutants typically penetrate into indoor environments, and nocturnal surface inversions are widespread across the globe, our findings have significant implications for more extensive human exposures to vehicle-related pollutants than previously indicated based on daytime measurements of roadway plumes.

Highlights

▸ Pollutant plumes from major roadways extend up to 2 km under stable conditions. ▸ Wide area impacts of freeway plumes are commonly observed in the SoCAB. ▸ Dilution rate coefficients in stable atmospheres are less than a tenth of daytime. ▸ Decay rates of UFP vary with size, changing size distributions. ▸ Mixed fleet emissions of UFP for a given traffic flow have decreased over time.

Previous Next

Keywords

Ultrafine particles; Traffic emissions; Freeway plume extension; Human exposure; Plume decay rates; Emissions reduction

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Nitrogen Dioxide (NO2) Pollution

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Basic Information about NO2

On This Page:

What is NO_2 and how does it get in the air?

Nitrogen Dioxide (NO_2) is one of a group of highly reactive gases known as oxides of nitrogen or nitrogen oxides (NO_x). Other nitrogen oxides include nitrous acid and nitric acid. NO_2 is used as the indicator for the larger group of nitrogen oxides.

NO_2 primarily gets in the air from the burning of fuel. NO_2 forms from emissions from cars, trucks and buses, power plants, and off-road equipment.

Effects of NO_2

Health effects

Breathing air with a high concentration of NO_2 can irritate airways in the human respiratory system. Such exposures over short periods can aggravate respiratory diseases, particularly asthma, leading to respiratory symptoms (such as coughing, wheezing or difficulty breathing), hospital admissions and visits to emergency rooms. Longer exposures to elevated concentrations of NO_2 may contribute to the

development of asthma and potentially increase susceptibility to respiratory infections. People with asthma, as well as children and the elderly are generally at greater risk for the health effects of NO_2.

NO_2 along with other NO_x reacts with other chemicals in the air to form both particulate matter and ozone. Both of these are also harmful when inhaled due to effects on the respiratory system.

- Learn more about Particulate Matter <https://epa.gov/pm-pollution> and Ozone <https://epa.gov/ground-level-ozone-pollution>.

Environmental effects

NO_2 and other NO_x interact with water, oxygen and other chemicals in the atmosphere to form acid rain. Acid rain harms sensitive ecosystems such as lakes and forests.

- Learn more about Acid Rain <https://epa.gov/acidrain>.

The nitrate particles that result from NO_x make the air hazy and difficult to see though. This affects the many national parks that we visit for the view.

- Learn more about Visibility and Haze <https://epa.gov/visibility>.

NO_x in the atmosphere contributes to nutrient pollution in coastal waters.

- Learn more about Nutrient Pollution <https://epa.gov/nutrientpollution>.

What is being done to reduce NO_2 pollution?

EPA's national and regional rules to reduce emissions of NO_2 and NO_x will help state and local governments meet the National Ambient Air Quality Standard (NAAQS).

- Learn about how air quality standards help reduce NO_2 <https://epa.gov/no2-pollution/setting-and-reviewing-standards-control-no2-pollution>

EPA identifies areas where the air quality does not meet the national NO_2 standards. For these areas, state, local, and tribal governments develop plans to reduce the amount of NO_2 in the air.

- Learn more about NO_2 air quality designations and state implementation plans (SIPs) <https://epa.gov/no2-pollution/applying-or-implementing-nitrogen-dioxide-standards>.

Nitrogen Dioxide Home <https://epa.gov/no2-pollution>

Nitrogen Dioxide Basics

Setting & Reviewing NO2 Standards <https://epa.gov/no2-pollution/setting-and-reviewing-standards-control-no2-pollution>

Primary NO2 Standard Regulatory Actions <https://epa.gov/no2-pollution/primary-national-ambient-air-quality-standards-naaqs-nitrogen-dioxide>

Secondary NO2 and SO2 Standard Regulatory Actions <https://epa.gov/so2-pollution/secondary-national-ambient-air-quality-standards-naaqs-nitrogen-dioxide-no2-and-sulfur>

Implementing NO2 Standards <https://epa.gov/no2-pollution/applying-or-implementing-nitrogen-dioxide-standards>

Primary NO2 Standard Implementation Actions <https://epa.gov/no2-pollution/2010-nitrogen-dioxide-no2-national-ambient-air-quality-standards-naaqs-implementation>

Implementation Data and Reports <https://epa.gov/no2-pollution/technical-data-and-reports-nitrogen-dioxide-measurements-and-sip-status>

Other Criteria Air Pollutants <https://epa.gov/criteria-air-pollutants>

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Current state of NOx emissions from in-use heavy-duty diesel vehicles in the United States

White paper

Published: 2019.11.26 • By Huzeifa Badshah, Francisco Posada, Rachel Muncrief

Download File NOx_Emissions_In_Use_HDV_US_20191125.pdf

This report assesses the real-world nitrogen oxide (NOx) emissions behavior of heavy-duty diesel vehicles in the United States measured using portable emissions measurement systems (PEMS). This assessment is based on 160 PEMS tests from engines certified to 0.2 grams per brake horsepower-hour (g/bhp-hr) of NOx. The tests include data from eight manufacturers and 26 unique engine families certified between 2010 and 2016. Second-by-second data was analyzed to compare against the compliance evaluation results and to assess the impact of vehicle speed, vehicle type, and manufacturer on real-world NOx emissions.

Key findings:

- The Not-to-Exceed (NTE) protocol evaluates less than 10% of the total emissions data to determine compliance for heavy-duty in-use NOx emissions. The average emission value of 0.18 g/bhp-hr obtained from the NTE evaluation is significantly lower than the value of 0.42 g/bhp-hr obtained when evaluated on a total route basis.
- A disproportionate amount of NOx emissions from heavy-duty vehicles is emitted during the low-speed operation characteristic of urban driving. Vehicle operation at speeds of less than 25 mph results in NOx emissions of more than five times the certification limit for the average heavy-duty vehicle in the study.
- At mid-speed driving conditions, between 25 and 50 mph and characteristic of suburban driving, average NOx emissions from heavy-duty vehicles (HDVs) are 2.7 times the certification limit. Only at highway speeds, above 50 mph, do HDVs present average NOx emissions at the certification limit and below the in-use NTE emissions limit of 0.3 g/bhp-hr.
- Line-haul trucks have the highest average NOx emissions at less than highway speeds. Their average NOx emissions of 1.41 g/bhp-hr are more than 7 times the engine certification limit in urban driving and more than 3 times the limit in suburban driving. Only during high-speed operation do line-haul trucks emit NOx at engine certification limit levels.

- A single line-haul truck emits the NOx equivalent of 100 cars for each mile driven in urban driving. Under urban driving conditions, line-haul trucks are emitting on average 7.0 g/mi of NOx, compared with less than 0.07 g/mi for a gasoline car. The PEMS data shows that these trucks, which are optimized for highway driving, spend on average 43% of their time and emit 40% of the total mass of NOx in urban-like operation, including low-speed driving and idling.

The current NTE in-use testing protocol is inadequate to evaluate the in-use performance of HDVs in the United States, especially at low-speed conditions. The prospect of future NOx standards set over low-load cycle and idle tests, in addition to the traditional federal test procedure, demands the adoption of a different tool for proper in-use compliance evaluation. That tool should ensure that in-use compliance is evaluated not only with highway data but also includes low-speed, low-load, and idle data. This would ensure that engine dynamometer emission results obtained in the laboratory translate to real-world benefits.

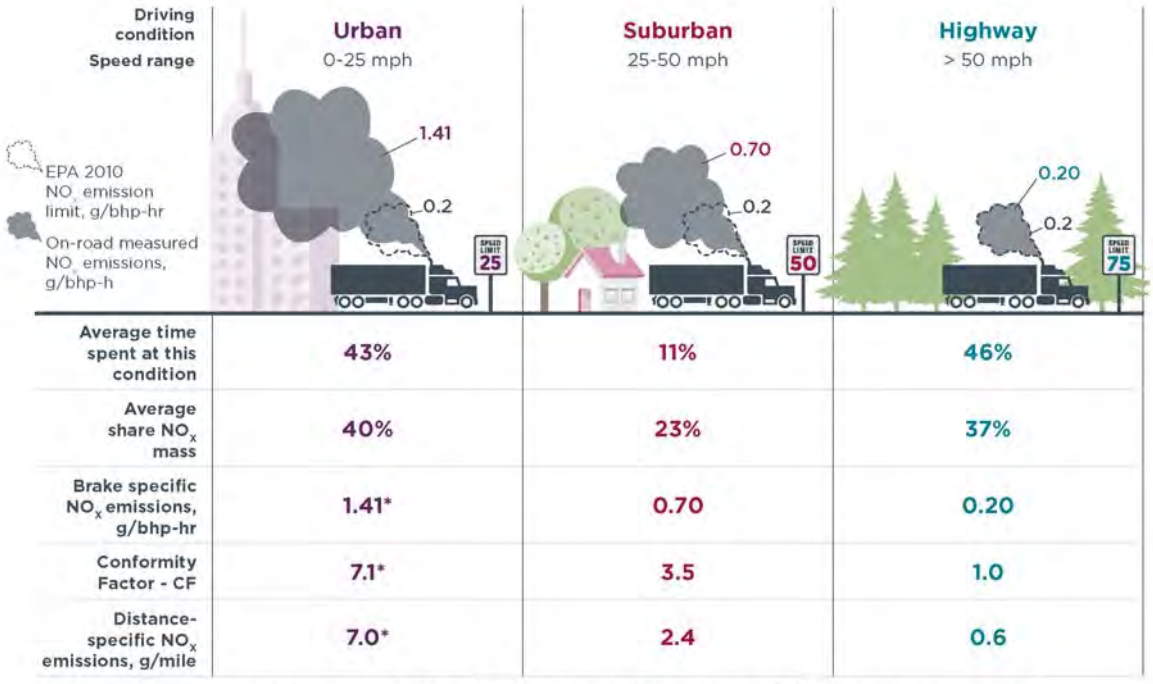

Figure ES-1 Comparison of line-haul vehicle NO_x emissions under urban, suburban, and highway driving conditions. Conformity factor is defined as ratio of measurement to engine dynamometer emission limits.

TAGS: *Air pollution / CO2 standards / GHG emissions / GHG standards / Heavy-duty vehicle efficiency / Regulatory design / Vehicle emissions standards / Vehicle test procedures*

WHITE PAPER

NOVEMBER 2019

CURRENT STATE OF NO_x EMISSIONS FROM IN-USE HEAVY-DUTY DIESEL VEHICLES IN THE UNITED STATES

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EXECUTIVE SUMMARY

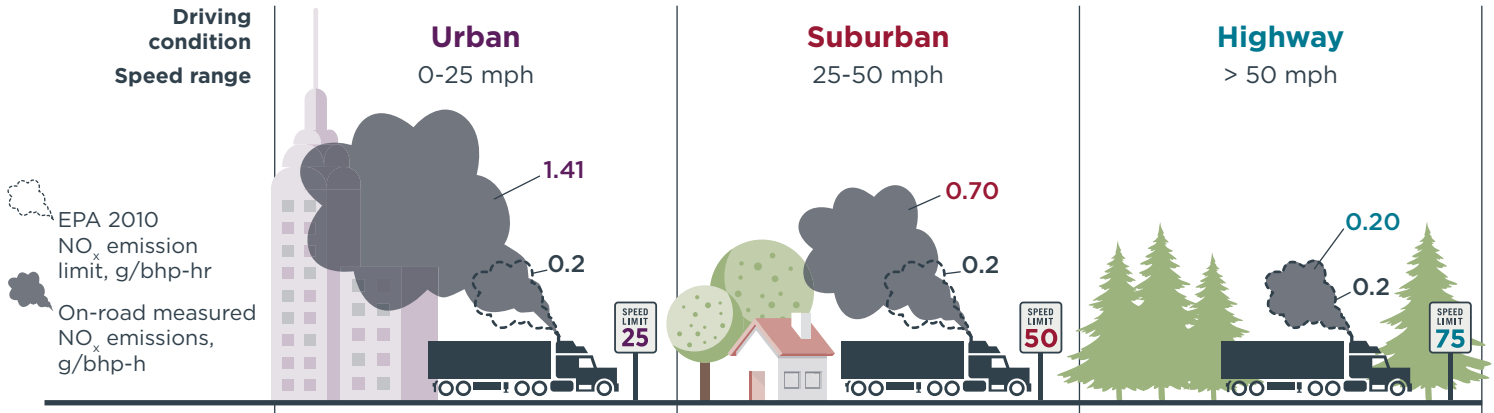
Emissions of nitrogen oxides (NO_x) from heavy-duty diesel engines are a significant contributor to ambient air quality issues and ozone pollution in many areas of the United States. Although the EPA's 2010 emissions standard for heavy-duty engines went a long way toward reducing NO_x emissions, there is still a significant gap between real-world and certified NO_x emissions from these engines. The two main regulatory agencies in the United States responsible for addressing NO_x from heavy-duty engines, the California Air Resources Board (CARB) and the U.S. Environmental Protection Agency (EPA), are developing new regulations to address weaknesses in the current standard. An important element in this process is to have a good understanding of the current real-world baseline emissions from existing diesel vehicles.

This report presents an assessment of the real-world NO_x emissions behavior of heavy-duty diesel vehicles in the United States measured using portable emissions measurement systems (PEMS). The real-world emissions data source is the publicly available Heavy Duty In-Use Testing (HDIUT) program, managed by the EPA. This assessment is based on 160 PEMS tests from engines certified to 0.2 grams per brake horsepower-hour (g/bhp-hr) of NO_x. The tests include data from eight manufacturers and 26 unique engine families certified between 2010 and 2016. Second-by-second data was analyzed to compare against the compliance evaluation results and to assess the impact of vehicle speed, vehicle type, and manufacturer on real-world NO_x emissions.

KEY FINDINGS:

- The Not-to-Exceed (NTE) protocol evaluates less than 10% of the total emissions data to determine compliance for heavy-duty in-use NO_x emissions. The average emission value of 0.18 g/bhp-hr obtained from the NTE evaluation is significantly lower than the value of 0.42 g/bhp-hr obtained when evaluated on a total route basis (total mass of NOx emitted divided by total work).
- A disproportionate amount of NO_x emissions from heavy-duty vehicles is emitted during the low-speed operation characteristic of urban driving. Vehicle operation at speeds of less than 25 mph results in NO_x emissions of more than five times the certification limit for the average heavy-duty vehicle in the study.
- At mid-speed driving conditions, between 25 and 50 mph and characteristic of suburban driving, average NO_x emissions from heavy-duty vehicles (HDVs) are 2.7 times the certification limit. Only at highway speeds, above 50 mph, do HDVs present average NO_x emissions at the certification limit and below the in-use NTE emissions limit of 0.3 g/bhp-hr.
- Line-haul trucks, defined as class 8 trucks for long-distance goods transport, have the highest average NO_x emissions at less than highway speeds. Their average NO_x emissions of 1.41 g/bhp-hr are more than 7 times the engine certification limit in urban driving and more than 3 times the limit in suburban driving (Figure ES-1)**.** Only during high-speed operation do line-haul trucks emit NO_x at engine certification limit levels.
- A single line-haul truck emits the NO_x equivalent of 100 cars for each mile driven in urban driving. The data shows that under urban driving conditions, line-haul trucks are emitting on average 7.0 g/mi of NO_x, compared with less than 0.07 g/mi for a gasoline car. The PEMS data shows that these trucks, which are optimized

for highway driving, spend on average 43% of their time and emit 40% of the total mass of NOx in urban-like operation, including low-speed driving and idling.



	Urban	Suburban	Highway
Average time spent at this condition	**43%**	**11%**	**46%**
Average share NO_x mass	**40%**	**23%**	**37%**
Brake specific NO_x emissions, g/bhp-hr	**1.41***	**0.70**	**0.20**
Conformity Factor - CF	**7.1***	**3.5**	**1.0**
Distance-specific NO_x emissions, g/mile	**7.0***	**2.4**	**0.6**

* Brake and distance specific NO_x emissions for Urban bin do not include Idle operation, only 1-25 mph operation is included

Figure ES-1 Comparison of line-haul vehicle NO_x emissions under urban, suburban, and highway driving conditions. Conformity factor is defined as ratio of measurement to engine dynamometer emission limits.

As a result of this analysis, it is evident that the current NTE in-use testing protocol is inadequate to evaluate the in-use performance of HDVs in the U.S., especially at low-speed conditions. The prospect of future NO_x standards set over low-load cycle and idle tests, in addition to the traditional federal test procedure, demands the adoption of a different tool for proper in-use compliance. That tool should ensure that in-use compliance is evaluated not only with highway data but also includes low-speed, low-load, and idle data. This would ensure that engine dynamometer emission results obtained in the laboratory translate to real-world benefits.

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INTRODUCTION

Nitrogen oxides (NO_x) are precursors to particulate matter with diameter of less than 2.5 micrometers ($PM_{2.5}$) and ground-level ozone, both of which are known to have adverse effects on human health. Long-term exposure to $PM_{2.5}$ and ozone is associated with increased risk of premature death from cardiovascular, lung, and kidney diseases (Burnett et al., 2018; Turner et al., 2016). In addition, direct NO_2 exposure is associated with asthma incidence among children and asthma emergency department visits (Anenberg et al., 2018). Reduction of NO_x emissions can lead to substantial public health benefits from improved air quality, including fewer hospitalizations and emergency room visits, fewer missed days at work, and lowered risk of premature death from cardiovascular, lung, and kidney diseases (EPA, 2018c)(EPA, 2018c). These benefits are the main drivers for decreasing NO_x emissions.

Heavy-duty diesel vehicles are a major source of NO_x emissions. In California alone, HDVs are responsible for more than 70% of NO_x emissions from on-road mobile sources (CARB, 2019). U.S. regulatory agencies have put in place regulations aimed at reducing NO_x emissions from heavy-duty diesel engines. In 2000, the U.S. Environmental Protection Agency (EPA) adopted heavy-duty engine emission standards for model years 2007–2010 and later engines—known as the EPA 2010 emissions regulation—aimed at reducing NO_x emissions by 90% compared with the model year 2004 standard.

A key part of the EPA 2010 regulation was the requirement that manufacturers conduct testing on in-use HDVs to prove compliance with the regulation and to help ensure that real-world emissions are in line with the limits set by the engine emission standards. This testing program is known as the manufacturer-run heavy duty in-use testing program (HDIUT). Under the HDIUT, selected vehicles are operated under normal driving conditions while engine emissions are continuously measured with portable emissions measurement systems (PEMS). The PEMS data collected is then evaluated under the Not-to-Exceed (NTE) protocol for regulatory compliance determination.

The NTE protocol is used to compare the results of the PEMS test data evaluation against the NTE limit. The NTE limit is defined as 0.30 g/bhp-hr, or 1.5 times the engine certification limit, plus an instrument accuracy allowance of 0.15 g/bhp-hr, for a total of 0.45g/bhp-hr, or 2.25 times the FTP certification limit (CFR, 2014).

Compliance with the NTE limit is determined by first calculating the average emissions of each valid NTE event that occurred during testing. A valid NTE event is a segment of PEMS data that meets a number of pre-defined boundary conditions during a minimum of 30 continuous seconds. These boundary conditions are described in Table 1. The average emissions from each valid NTE event window are then compared against the NTE emissions limit (CFR, 2005). A test is considered to pass if a minimum of 90% of time-weighted NTE events result in emissions below the NTE limit.

Table 1 Conditions for PEMS data to be considered valid for an NTE event.

Parameter	PEMS data condition
Engine speed	Engine speed must be greater than a given engine speed (called n_{15}) defined from the engine full load curve **(~850 to 950 rpm)**
Engine torque	Engine torque must be greater than or equal to **30%** of the peak torque
Engine power	Engine power must be greater than or equal to **30%** of the peak power
Exhaust gas temperature	Exhaust gas temperature has to be above **250°C** (measured 1.0 foot from SCR outlet)
Intake manifold temperature[a]	Intake temperature must be greater than a function of intake pressure **(~38°C / 100°F)**
Engine coolant temperature[a]	Coolant temperature must be greater than a function of intake pressure **(~60°C / 140°F)**

Note: For details please see the code of federal regulations 40 CFR Part 86
[a] For engines with an exhaust gas recirculation system (EGR) to reduce NO_x emissions.

The adoption of the EPA 2010 heavy-duty regulations has resulted in positive outcomes, with total fleet NO_x emissions dropping by more than 40% (EPA, 2018b). Remote sensing data from HDVs in California, covering a wide span of vehicle model years, shows significant improvements in average NO_x emissions, moving from close to 20 grams of NO_x per kilogram (gNO_x/kg) of fuel for model year 2004 to a range of 3.8-13.9 gNO_x/kg of fuel for vehicles post-model year 2010 (Bishop, 2019).

At the same time, in-use emissions testing has shown that there is still a gap between real-world NO_x emission levels and certified levels. Remote sensing data from HDVs in California show that the best performers at 3.8 gNO_x/kg of fuel emit about 3.3 times more than the FTP emissions standard would require (Bishop, 2019). Moreover, PEMS testing data on post-model year 2010 line-haul and delivery trucks shows that NO_x emissions reach an average of 0.45 g/bhp-hr, or twice the FTP standard (Besch, 2018; Duncan & Hamady, 2019; Quiros et al., 2016). Using a full-scale mobile laboratory housed in a tractor-trailer for operation on the roadway, CARB and West Virginia University researchers found that NO_x emissions from a group of four Class 8 trucks operating on Californian roads ranged from 0.16– 0.96 g/bhp-hr (Quiros et al., 2016).

As a result, air quality is still a significant problem in certain regions of the United States. California's Los Angeles South Coast Air Basin and San Joaquin Valley are the two areas most affected, classified as "extreme" under the national 8-Hour Ozone standard (EPA, 2019a). In the South Coast Air Basin, reductions of 70% of NO_x emissions from today's levels would be needed by 2023 to meet the national ambient air quality standard for ozone (CARB, 2017; Heroy-Rogalski, Lemieux, & Robertson, 2019).

There is potential to further reduce real-world NO_x emissions from heavy-duty diesel engines. CARB and the EPA are working to update and improve existing regulations targeting this issue. California has announced the HDV Low NO_x regulation aimed at reducing FTP-based NO_x standards by as much as 90% and introducing testing conditions that would require improved emissions control at low loads and low vehicle speeds, as well as an improved in-use testing protocol (CARB, 2016). At the federal level, the EPA announced the development of the Cleaner Trucks Initiative, which explores similar changes to the EPA 2010 NO_x emissions standard, also aiming at real-world emission reductions (EPA, 2018a).

An important input into those regulatory processes is to have a good understanding of the real-world baseline emissions from existing heavy-duty diesel vehicles certified under the EPA 2010 regulation. The objective of this report is to present an independent assessment of the real-world NO_x emissions behavior of EPA 2010-certified diesel HDVs in the United States and identify critical driving conditions that result in excessive NO_x emissions.

HDUIT DATA SUMMARY

The analysis presented here focuses on PEMS data from HDVs with post-2010 model year engines as reported by the EPA (EPA, 2019b). These data are a product of the manufacturer-run HDIUT program from which in-use compliance is determined.[1] Each test file consists of second-by-second data including parameters such as vehicle speed (mph), NO_x and CO_2 emissions (g/s), and engine power (hp). Vehicle and engine specifications as well as the results of the NTE compliance calculations are also provided. For each test, there was no prescribed driving cycle as the PEMS data was being collected while the vehicles were in normal operation as per regulatory requirements. Vehicle loading was not reported.

A total of 287 tests were carried out between 2010 and 2019 for 2010-2016 model year engines, of which 98 were conducted on engines compliant under the banking credit program and 189 were from noncredit engines certified to the 0.2 g/bhp-hr NO_x standard. The credit program allowed manufacturers to use banked credits from prior years to certify engines to a higher NO_x emissions limit, 0.3–0.5 g/bhp-hr. As the credit engines have now been phased out, the noncredit engines are the most representative of the current baseline. Therefore, the analysis we present here is focused on the emissions of noncredit engines only.[2]

We removed all Cummins non-credit engines in the HDIUT dataset because all the diesel engines families tested were covered by recalls or were natural gas engines. The HDIUT dataset contained 18 tests from two engine families manufactured by Cummins that are part of a voluntary recall involving trucks with 2010–2015 model year engines with selective catalytic systems.[3] The removal of those tests ensures that the data reported here corresponds to diesel vehicles and engines operating with no mechanical or technical issues. We also removed 11 PEMS tests that were conducted on vehicles with Cummins natural gas engines and included in the HDIUT Diesel dataset.

Removing the Cummins recalled and natural gas engine tests from the noncredit engine test pool results in 160 tests from 26 unique engine families. Results in this study are based solely on this dataset excluding the recalled and natural gas engines. Appendix A provides a summary of NOx emissions from Cummins engine families available in the HDIUT dataset and their relative performance against the fleet studied here.

Figure 1 shows the breakdown of engine tests in the dataset differentiated by manufacturer and model year. Vehicle types studied are listed by manufacturer. Most PEMS tests were carried out in vehicles with engines manufactured by Volvo, and Detroit Diesel. Line-haul and delivery trucks were more frequently tested under the program. The “other” vehicle category includes applications such as drayage, refuse, and transit buses. PEMS tests were carried out on 76 line-haul trucks, 67 delivery, and 17 in the other vehicle category. The prevalent engine model years in the HDIUT dataset cover

1 The HDIUT program requires manufacturers to test up to 25% of engine families per year. The EPA certified 290 engine families between 2010 and 2016. During this period 51 engine families were tested under the HDIUT program (EPA, 2019b).

2 Tests from credit engines were mainly concentrated in the 2010 and 2011 model years, and a few during the 2013-2015 model years. Credit engines were manufactured by General Motors, Navistar and Cummins.

3 The U.S. EPA announced in July 2018 that Cummins will voluntarily recall roughly 500,000 model year 2010-2015 medium- and heavy-duty trucks. The recall involved the replacement of SCR systems that were found less durable than is required by the emission regulations (EPA, 2018d).

2010–2014 engines. There were 16 PEMS tests available from 2015 engines and three from 2016 engines.

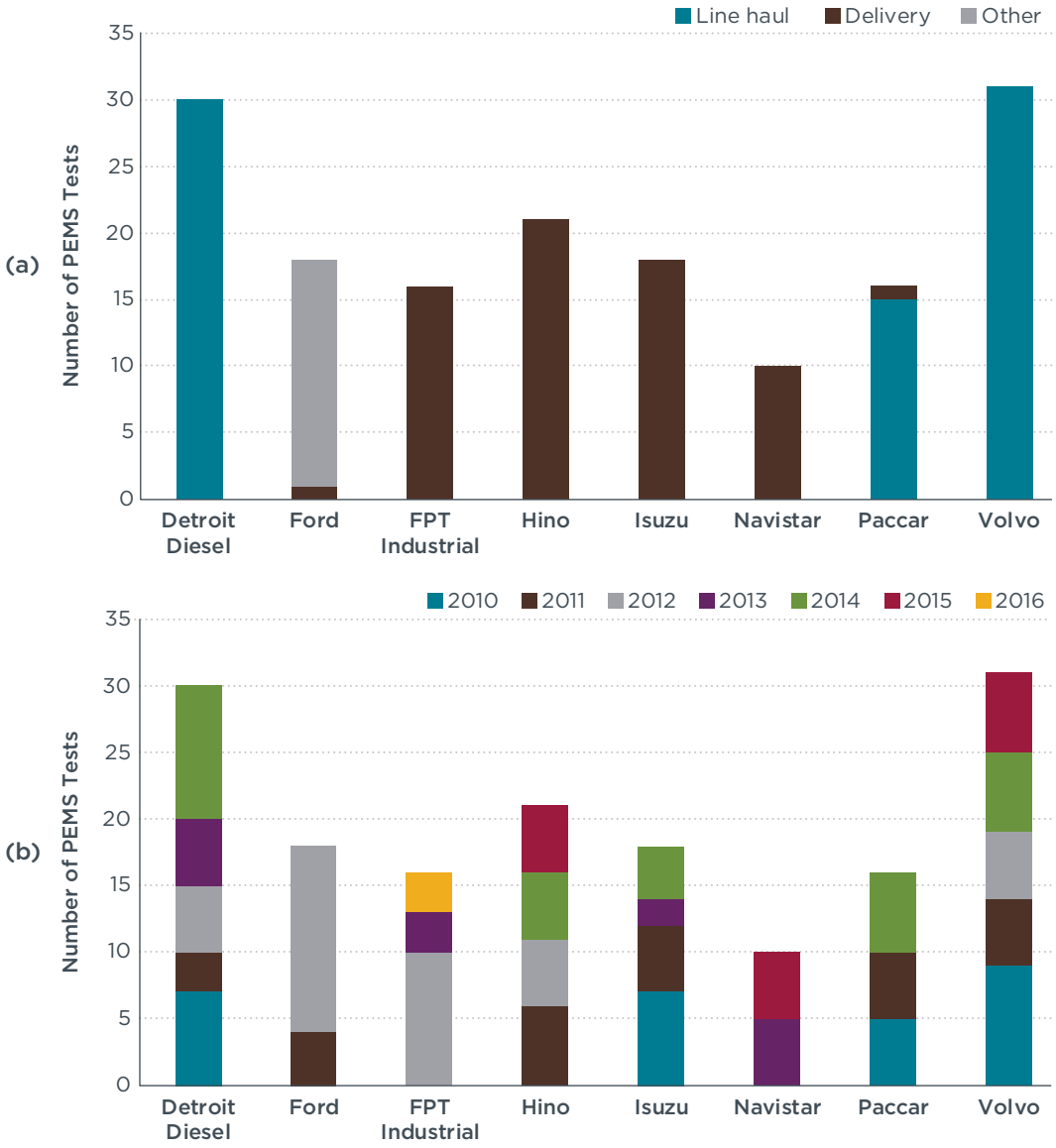


Figure 1 Number of PEMS tests in the HDIUT Database for noncredit engines by (a) vehicle type and (b) engine model year

A breakdown of the number of engine tests and the respective engine families tested by year and by manufacturer is provided in Table 2.

Table 2 Number of Tested Families and Tests

Engine manufacturer	Engine families	PEMS tests
Detroit Diesel	4	30
Ford	3	18
FPT Industrial	2	16
Hino	4	21
Isuzu	3	18
Navistar	2	10
PACCAR	3	16
Volvo	5	31
Total dataset	**26**	**160**

Appendix B presents a table summarizing each PEMS test used in the analysis. The table describes vehicles tested, engine characteristics, driving conditions, environmental conditions, and emission results as reported in the HDIUT tests and as calculated following our methods.

METHODOLOGY

In this paper we analyze second-by-second data from vehicles with heavy-duty diesel engines certified to 0.2 g/bhp-hr NO_x. The analysis includes data from all engine and vehicle operation conditions. This includes all power, torque, and engine rpm conditions, cold-start periods, low exhaust temperature (T < 250°C), and DPF regeneration events. This is done to best capture the overall picture of emissions under real-world driving conditions.

There were two cases when PEMS data was filtered out for the purposes of our analysis:

- The "zero-check" condition, in which the PEMS instrument auto-zeros itself while the rest of the OBD parameters remain live.
- Times when the OBD stream is interrupted for various reasons resulting in invalid data ("blanks," "null," or "NA" strings in the data output).

A key focus of our analysis was to gain an understanding of emissions performance in urban driving conditions. Urban driving is typically defined by low vehicle speed and low engine load. Those conditions lead to lower exhaust temperatures where effective emissions control may be more challenging (Anderson, 2018; Dixit et al., 2017). As the HDIUT dataset does not contain torque values, we rely on reported engine power for load.

As shown in Figure 2, we determined that there is a strong correlation between engine power and vehicle speed in the dataset. Average normalized power values for each speed bin are plotted versus the average speed within each bin. The analysis indicates that low-speed operation is generally associated with low-power operation and high-speed operation is associated with higher-power operation. With this being the case, we chose to report the majority of our analysis on a vehicle speed basis only.

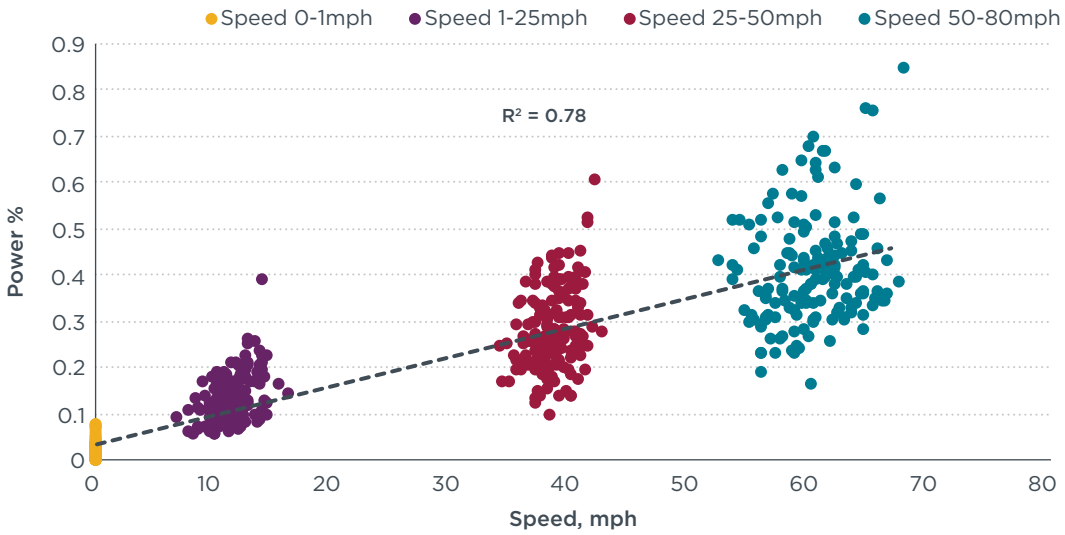


Figure 2 Speed versus normalized power. Each point represents the average engine power and average speed within the given speed bin for all 171 PEMS tests.

Data is segregated by vehicle speed parameter in the following ways throughout the report with approximate engine power ranges in percentage of P_{max} - maximum engine power listed for reference:

- Idle: vehicle speed <1mph, <10% of P_{max}
- Urban: vehicle speed of 1–25 mph, 5–25% of P_{max}
- Suburban: vehicle speed of 25–50 mph, 10–45% of P_{max}
- Highway: vehicle speed ≥50 mph, >25% of P_{max}
- Route w/o idle: all vehicle speeds excluding idle
- Route: all vehicle speeds

Wherever applicable, the data presented in these bins have been compared against current U.S. engine certification and in-use NTE standards for reference. Although there are no obligations for the vehicle to meet any regulatory standards under this binned metric, it allows for a useful comparison between certified and real-world emissions.

RESULTS AND DISCUSSION

NTE VERSUS TOTAL ROUTE NO_x EMISSIONS

As previously mentioned, HDIUT data is assessed for regulatory compliance purposes via the NTE protocol, where a series of exclusions must be applied to evaluate the real-time PEMS dataset. These exclusions reduce the useable portion of the data for compliance evaluation. Figure 3 shows the percentage of total test time spent within a valid NTE event for each PEMS test, differentiated by manufacturer. On average, the total amount of time spent in valid NTE events was 8.7% of the total test time.[4] This number can vary significantly from test to test with a minimum of zero valid NTE events to as much as 50% of the test time within valid NTE events. The average time spent in valid NTE events by different manufacturers varied from 3% (Ford) to 22% (FPT Industrial).

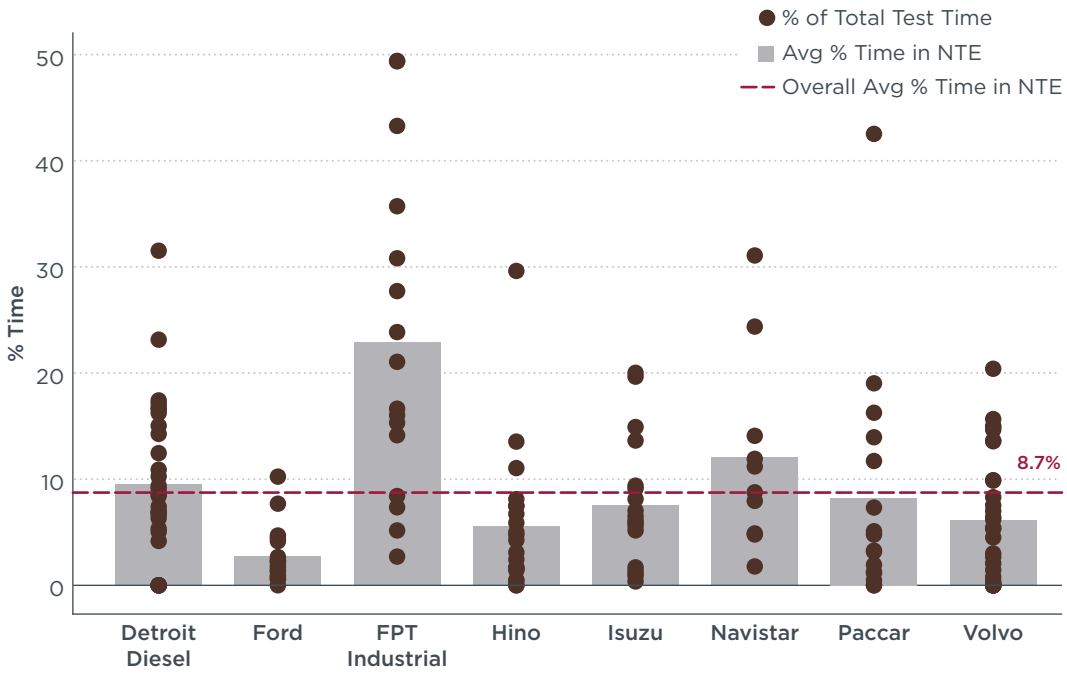


Figure 3 Percentage of time in valid NTE events, manufacturer average (bars) and individual PEMS results (dots).

In Figure 4, we compare the average NO_x over the total route without idle to the average NO_x during valid NTE events by manufacturer. The whiskers on all figures throughout the paper represent the 95% confidence interval for the metric of interest. The average NTE NO_x emissions were 0.18 g/bhp-hr across all manufacturers, while the average NO_x emissions for the entire route without idle were approximately 1.9 times higher, at 0.34 g/bhp-hr. Including idle, total route NO_x emissions increased to 0.42 g/bhp-hr. Our route results are similar to those found in previous studies that have analyzed subsets of this public dataset (Besch, 2018; Duncan & Hamady, 2019; Lee et al., 2019; Spears, 2018).

4 Tests with zero time spent in NTE events are included. Excluding these tests would yield 9.8% as the overall average percentage of time in NTE events.

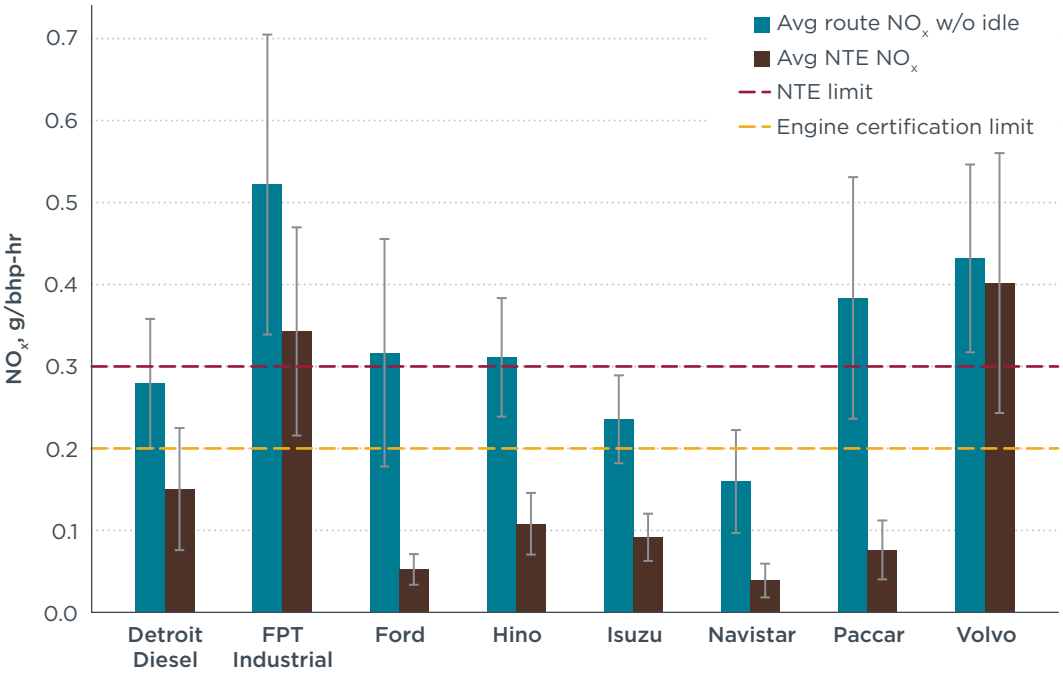


Figure 4 Comparison of Route NO_x to NTE NO_x by manufacturer. Whiskers represent the 95% confidence interval of the mean.

The variability between NTE and overall route emissions can be further assessed by differentiating by engine family (Figure 5). As previously mentioned, 26 engine families were included in the testing database, and each family was tested three to 10 times. Out of 160 tests, 22 had NO_x emissions above the NTE threshold of 0.3 g/bhp-hr when calculated using the NTE methodology (Figure 5a). The HDIUT database shows that 24 of 160 tests (15%) had no valid NTE events, which under the existing regulation triggers an additional PEMS test.

Tests that exhibit NO_x emissions higher than the NTE limit when evaluated using the NTE analysis do not automatically result in a "fail." An additional PEMS accuracy margin tolerance of 0.15 g/bhp-hr is added to the 0.30 limit, which enables NO_x emissions to be as high as 0.45 g/bhp-hr and still result in a pass. Additionally, specific procedures outline the method to be followed if tests lead to NO_x emissions above the NTE limit + accuracy margin limit.[5]

While exceeding the NO_x limit based on NTE type analysis is problematic, the situation is exacerbated when NO_x emissions are evaluated based on the full route analysis, leading to a significant increase in the number of tests and families exceeding the in-use limit. Comparing the test NO_x average route values against the NTE limit shows that 67 tests had NO_x emissions above the limit even after excluding idling (Figure 5b). Of the 26 engine families, 21 had at least one test exceeding the in-use limit when calculated using total route NO_x emissions data excluding idling.

5 The initial test campaign is carried out with the selection of five engines from a certain family. If a vehicle test were to fail the NTE limits, a series of conditionals regulate the number of additional engines that must be tested for an engine family to successfully declare a "pass." Refer to 40 CFR § 86.1915 "What are the requirements for Phase 1 and Phase 2 testing?" for further details regarding the pass/fail criteria.

Significant test differences within a single family were also observed, with the highest route average NO_x being 1.4–16.4 times the lowest route average NO_x value. The largest difference in route average NO_x was found for the Volvo AVPTH12.8S01 (D13H 425) family, where average test results ranged from 0.09–1.4 g/bhp-hr.

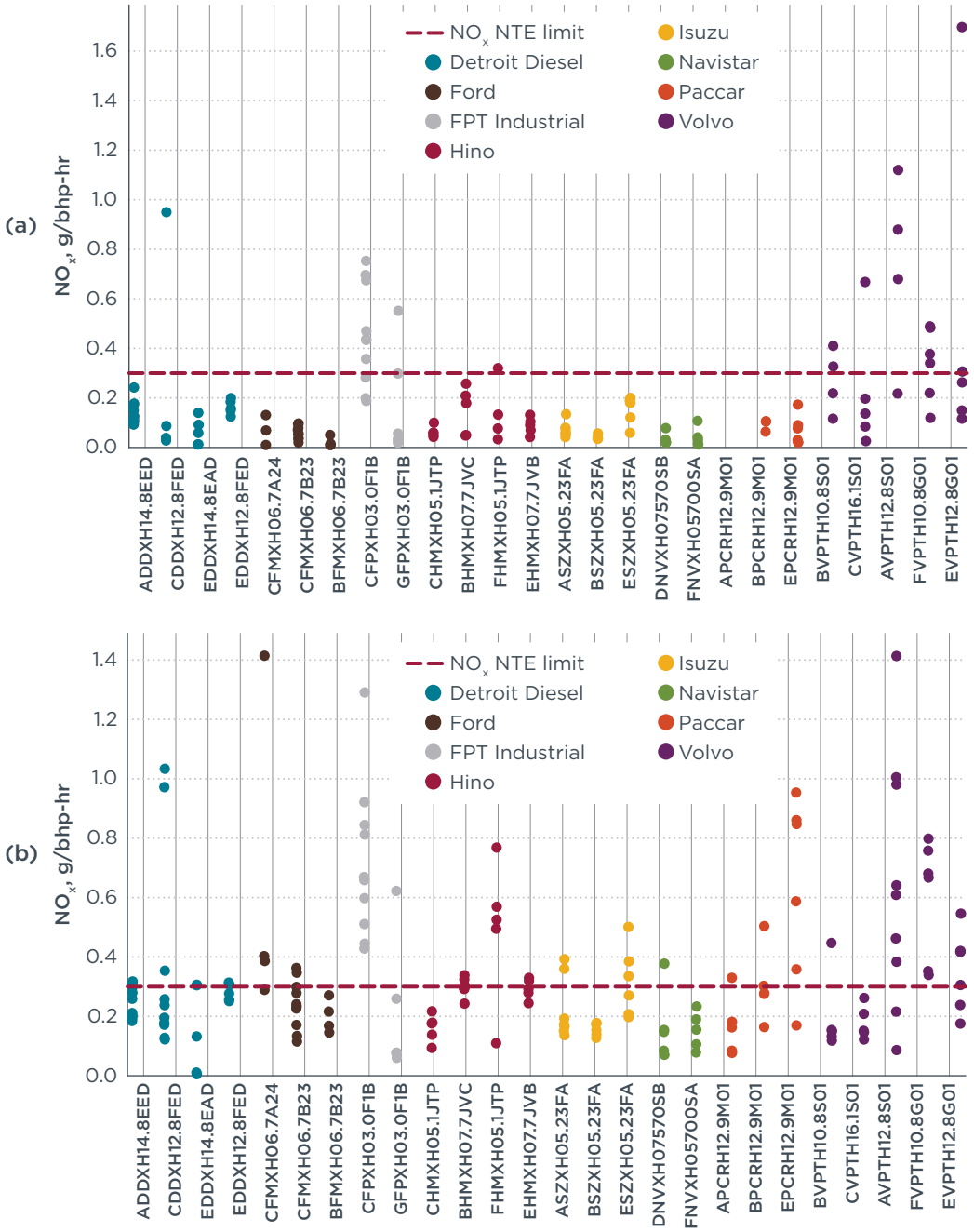


Figure 5 Engine family NOx emissions reported from (a) NTE data from HDIUT, and (b) calculated from route analysis (no data exclusions) of PEMS tests

Because the NTE approach excludes significant amounts of data from the emissions analysis, we focus the remainder of this report on analysis of full datasets differentiated by vehicle speed bins.

NO_x EMISSIONS VERSUS VEHICLE SPEED

Average emissions factor

The average NO_x emissions for all 160 PEMS tests segregated by speed bins are shown in Figure 6. The bars show the average per test NO_x emissions for all the data points that meet the given speed condition. Regulatory limits for engine certification of 0.20 g/bhp-hr and for in-use NTE of 0.30 g/bhp-hr are also included for reference. The average total route NO_x emissions across all manufacturers was 0.42 g/bhp-hr, and 0.34 g/bhp-hr excluding idling (speed < 1.0 mph)[6]. Average brake-specific NO_x emissions ranged from 0.20 g/bhp-hr at highway speeds to 1.05 g/bhp-hr at urban driving speeds. This is most likely because higher speeds result in higher exhaust temperatures where catalytic NO_x control is less challenging. This trend of higher brake-specific NO_x emissions at lower vehicle speeds has also been reported by researchers in the United States and Europe for HDVs with SCR systems (Grigoratos, 2019; Mendoza-Villafuerte, 2017; Sandhu & Sonntag, 2019).

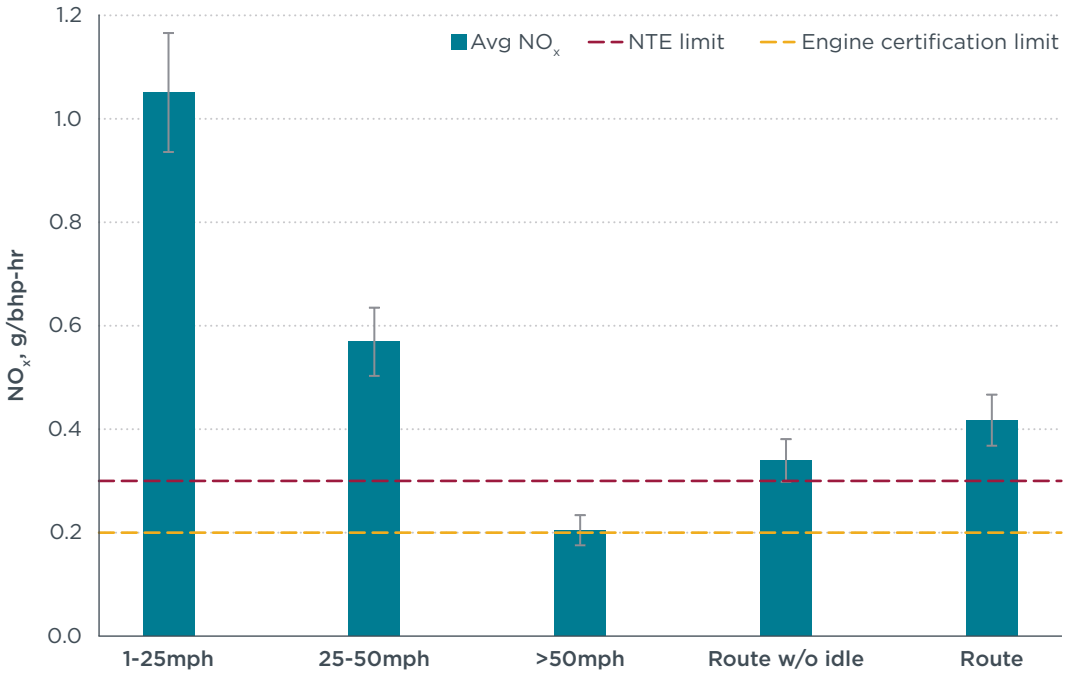


Figure 6 Average NO_x emissions in g/bhp-hr differentiated by vehicle speed. Whiskers represent the 95% confidence interval of the mean.

Exhaust temperature

The average exhaust temperature for each speed bin is shown in Figure 7. Average exhaust temperature increased from 213°C in urban driving to 290°C under highway driving conditions. The average route temperature, excluding idling data, was close to 260°C. Exhaust temperature data is measured within 12 inches downstream of the SCR system, according to in-use testing regulations. The existing NTE protocol does not consider emissions data collected below exhaust temperatures of 250°C, shown as a green dashed line in Figure 7. Catalytic NO_x control is more challenging at exhaust

6 At idle conditions, the work term may be small and may yield an artificial increase in NO_x. Brake-specific emissions would asymptotically approach infinite under idle conditions.

temperatures below 250°C. Therefore, it is not surprising that driving conditions at speeds below 25 mph or with higher percentage of idle result in higher average in-use NO_x emissions.

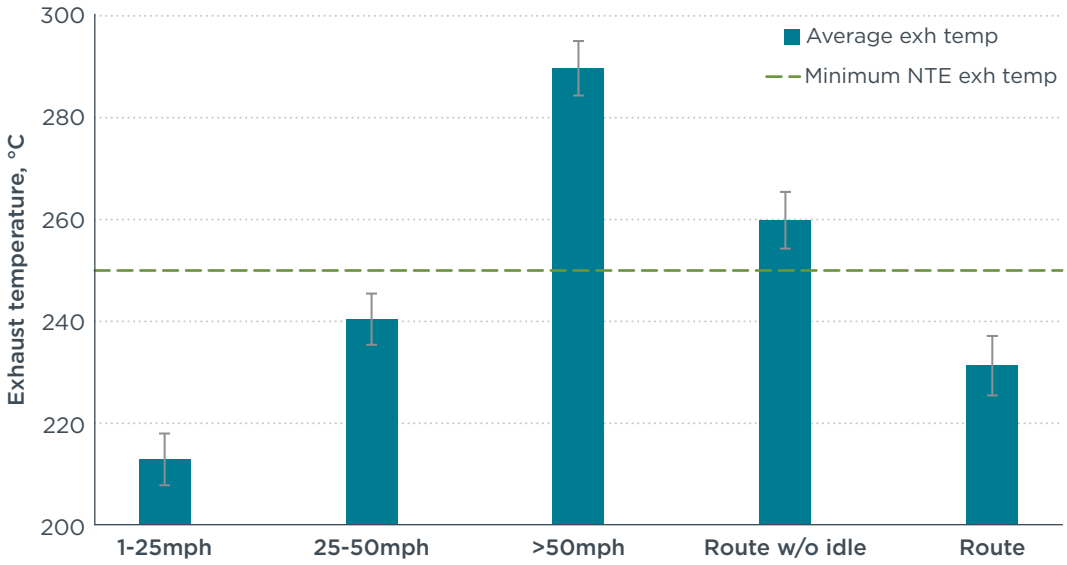


Figure 7 Average exhaust temperature per speed bin. NTE protocol minimum exhaust temperature required for individual datapoint validity is shown as a green dashed line at 250°C. Whiskers represent the 95% confidence interval of the mean.

Vehicle type

Average NO_x emissions in different speed bins were further analyzed by vehicle types: line haul, delivery, and other vehicles.[7] For the line-haul trucks, we analyzed 76 PEMS tests from 12 engine families; for the delivery trucks, 67 PEMS tests from 13 engine families; and for the other vehicle types, 17 PEMS test from three engine families. Note that the number of families does not add up to the previously mentioned count of 26 because several engine families are found in multiple vehicle types.

Line-haul trucks exhibited almost twice the average brake specific NO_x emissions as delivery or other vehicle types during low speed (1–25 mph) urban driving conditions (Figure 8a). Their average NO_x emissions of 1.41 g/bhp-hr are more than 7 times the engine certification limit in urban driving and more than 3 times the limit in suburban driving. At highway speeds, NO_x emissions seemed very similar across vehicle types. The total route average showed almost no differences among the vehicles studied.

Line-haul vehicles also had the highest CO_2-specific emission factors for 1–25 mph and 25–50 mph operation, whereas the emission factors of all three vehicle types were similar for highway-speed operation (Figure 8b). The CO_2-specific metric allows for direct comparison of different PEMS tests by lessening the impact of variations in the duty cycle or other test-to-test variables that impact emissions. It also allows comparisons with available data on HDVs. Bishop et al. captured on-road emissions from

7 Line-haul vehicles include class 8 tractor-trailers with a median gross vehicle weight rating (GVWR) of 80,000 lbs. Delivery vehicles includes box, drayage, and local transport trucks with a median GVWR of 19,500 lbs. The "other vehicle" category includes garbage collectors, bucket trucks, and various other localized applications also with a median GVWR of 19,500 lbs.

28 HDVs of model years 2014 and later operating at speeds around 10 mph and reported fuel-specific values of 3.8 gNO_x/kg of fuel, or 1.2 gNO_x/kg of CO_2 (Bishop, 2015). This value is lower than the 1.8 gNO_x/kg of CO_2 average emission values reported across all vehicles under low-speed conditions in the HDIUT dataset but well within the spread of the PEMS-based results.

Under urban driving conditions, line-haul trucks emitted, on average, 7.0 gNO_x/mile (Figure 8c). To put this in perspective, this is 100 times the Tier 2 Bin 5 emissions limit of passenger vehicles in the United States.[8] Previous studies have confirmed that real-world emissions from gasoline light-duty vehicles are generally at or below the Tier 2 standards, even in urban driving (McCaffery, 2019).

8 Tier 2 NO_x emissions standard over the FTP 75 for full useful life of Bin 5 passenger vehicles is at 0.07 g/mile.

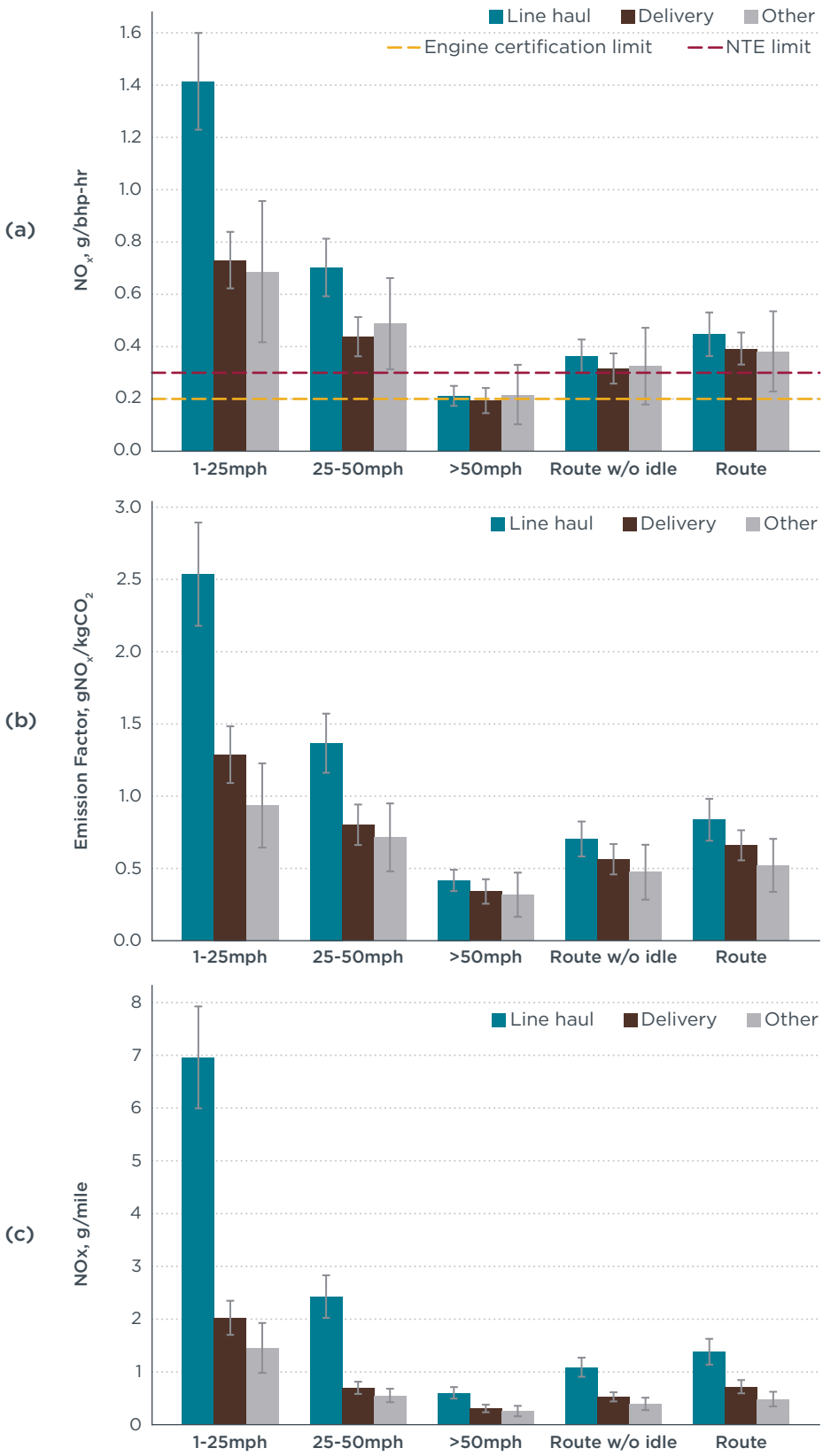


Figure 8 NO_x emissions by vehicle type in speed bins: (a) brake-specific (b) CO_2 specific, and (c) distance specific. Whiskers represent the 95% confidence interval of the mean.

Average exhaust temperature, shown in Figure 9, does not fully explain the higher urban NO_x values for line-haul vehicles than for delivery trucks as both types had very similar average exhaust temperatures. It is likely that the engine and aftertreatment control strategy for line-haul vehicles are optimized for controlling NO_x under higher speeds in line with their primary intended use. Another contributing factor to poor NO_x emissions control under low-speed urban driving could be rooted in the inability of the NTE in-use testing protocol to evaluate that driving condition for compliance purposes which disincentivizes manufacturers from optimizing NO_x around that operating region.

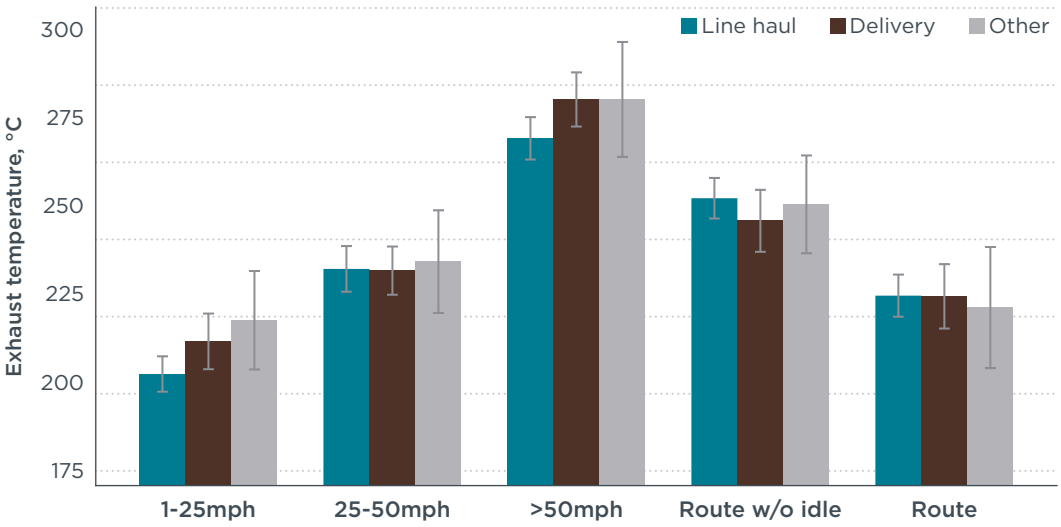


Figure 9 Average exhaust temperature (downstream of the SCR) by vehicle type in speed bins. Whiskers represent the 95% confidence interval of the mean.

Percentage of time spent and mass emitted by driving condition

As mentioned earlier, the EPA's in-use HDV testing program mandates that vehicles be driven as normally in everyday use. Figure 10 shows the percentage of time that each vehicle type spent in the different speed bins. All vehicle types spent a third or more of their time at speeds of less than 1 mph or idling. Combining idle (0–1 mph) with low-speed driving (1–25 mph), the percentage of time rises to 41%–55% depending on vehicle type. Line-haul trucks spent almost twice as much time in highway driving conditions as the two other vehicle types, offset by less time in urban and suburban driving.[9]

9 One caveat on this duty cycle summary is that these times are affected by the fact that the vehicles tested under the HDIUT program are expected to return to base at the end of the day for data collection and PEMS calibration work. This daily return to base may not be representative of normal driving for some long-haul and line-haul operations. This may result in idle and low speed shares for line-haul trucks in the HDIUT database to be higher than those found in real-world normal driving. A better source of HDV activity for class 8 heavy-duty trucks can be found in the Fleet DNA database managed and maintained by the National Renewable Energy Laboratory (NREL). The share of time operating at idle for long-haul trucks as measured by NREL fleet DNA data is 15.3% (Kotz & Kelly, 2019). The HDIUT data for line-haul class 8 trucks better matched the time share of combined-unit short-haul trucks in the NREL fleet DNA study, at 31%.

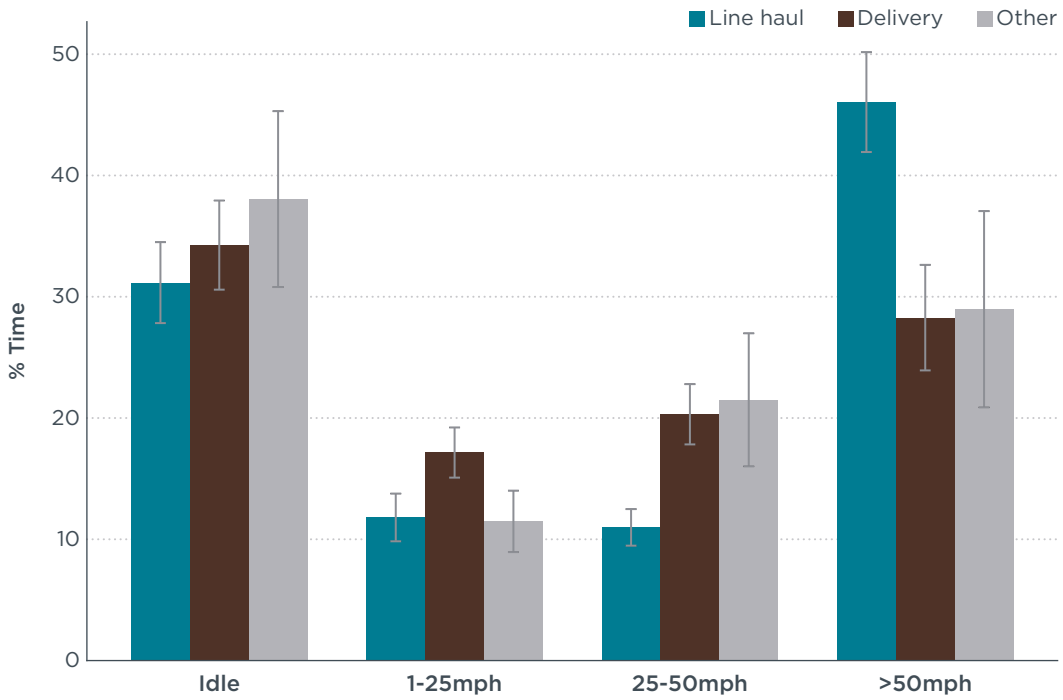


Figure 10 Percentage of time spent at each speed condition by vehicle type. Whiskers represent the 95% confidence interval of the mean.

Figure 11 shows the percentage of average total NO_x and CO_2 emitted in each speed bin by vehicle type. The percentage of total emissions in each speed bin does not necessarily align with the percentage of time spent in each speed bin. For all vehicle types, the vehicles were idling at least 30% of the time, but NO_x emissions during idling represent about 20% of the total, and CO_2 emissions represent around 10% of the total. Combining idling and low-speed driving, which was observed 41%–55% of the time, represented 40%–44% of total average NO_x emissions and 15%–20% of total average CO_2 emissions for all vehicle types. For line-haul trucks, which spent 47% of their time at highway speeds, high-speed NO_x emissions represent about 37% of the total and more than 70% of the CO_2 emissions.

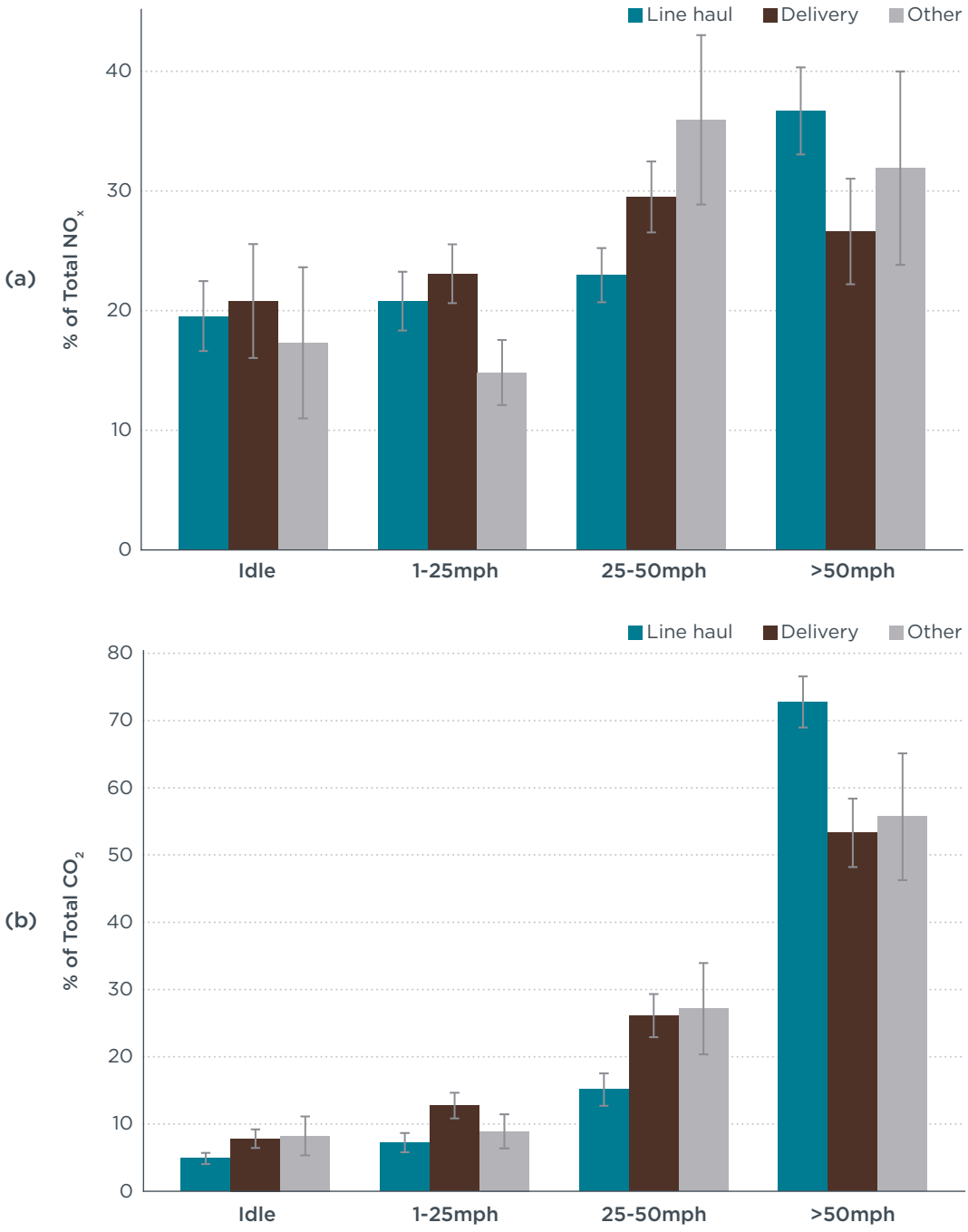


Figure 11 Percentage of total average (a) NO_x and (b) CO_2 emitted at each vehicle speed by vehicle type. Whiskers represent the 95% confidence interval of the mean.

Manufacturer

Because of the significance of urban emissions in terms of mass and share of operating time, we conducted a more extensive analysis of urban NO_x emissions excluding idle by manufacturer and vehicle type (Figure 12). Urban NO_x emissions of less than 0.3 g/bhp-hr were achieved only during 13 of the 160 tests. The high dispersion of urban NO_x performance was found in most manufacturer results, highlighting the wide range of low

vehicle speed solutions for NO_x control. This dispersion suggests that porous regulatory provisions for in-use emissions control are generating inconsistent technical solutions for NO_x control across the sector.

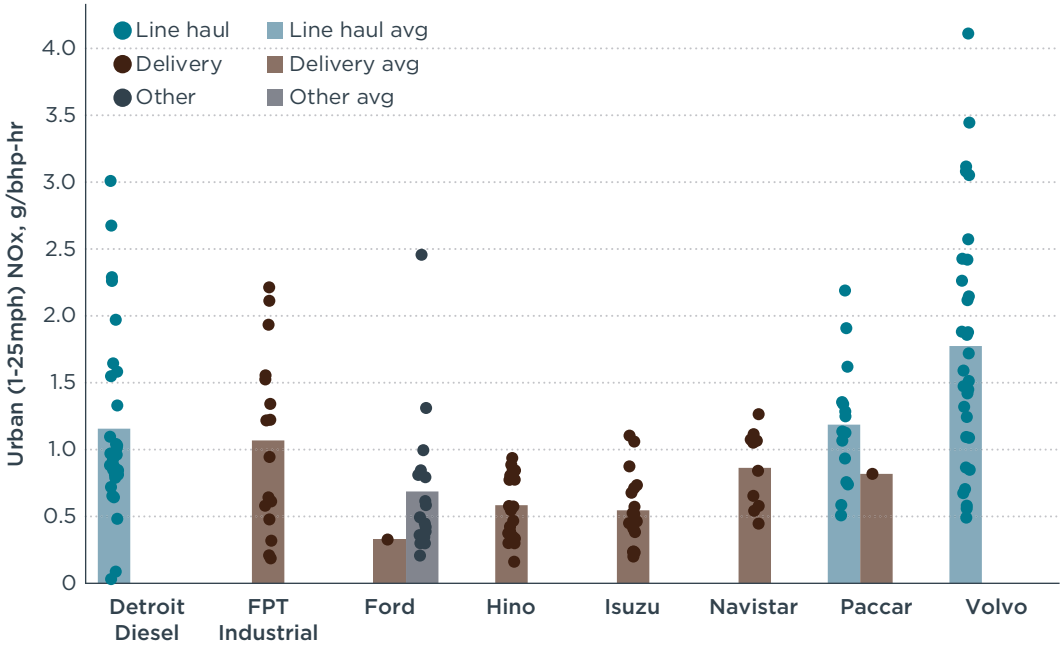


Figure 12 Urban NO_x emissions – g/bhp-hr

The dispersion of NO_x results among line-haul urban driving deserves further analysis. The three manufacturers with line-haul vehicle emissions data were Detroit Diesel, PACCAR, and Volvo. Table 3 shows summary statistics for number of tests, test-to-test differences indicated by minimum and maximum, and averages. Only two of the 76 tests for line-haul trucks had average urban NO_x emissions of less than 0.3 g/bhp-hr. The range of urban emission results between minimum and maximum values for the same manufacturer was orders of magnitude apart, from four to 100 times. Some of the maximum values shown in Table 3 are close to typical engine-out NO_x values found at low-load conditions, 3–4 g/bhp-hr. This suggests that the NO_x aftertreatment system may have been inactive during those sections of the tests.

Table 3 Summary statistics on urban driving (1–25 mph) brake-specific NO_x emissions (g/bhp-hr) for line-haul vehicles

Statistic	Detroit Diesel	Paccar	Volvo
# of Tests	30	15	31
Min	0.03	0.51	0.49
Max	3.01	2.19	4.11
Average	**1.16**	**1.19**	**1.77**

NO_x emissions of less than 0.3 g/bhp-hr under urban driving conditions were found in 13 PEMS tests. Those PEMS came from engine families manufactured by five of the eight companies and cover a wide range of engine displacements and applications.

The 0.3 g/bhp-hr in-use limit was achieved in urban driving excluding idle by Detroit Diesel with one engine family (15.8 L), Ford with two engine families (6.7 L), FPT industrial with one engine family (3.0 L), Hino with two engine families (5.0 L), and Isuzu with two engine families (5.2 L). These results indicate that real-world urban NO_x emissions in line with the existing in-use emissions standard limits are feasible with existing emissions-control technology.

Idle NO_x emission rates in gNO_x/hr are shown in Figure 13 for the different manufactures and vehicle types. Also shown for reference are the current voluntary limits set by CARB for idling emission rates (30 gNO_x/hr) as well as the proposed 2024 limit (10 gNO_x/hr) (CARB, 2019).[10] On average, line-haul trucks produced the highest NO_x emission rates at idle at 24 g/hr, while delivery trucks produced 9 g/hr and other vehicles 5 g/hr. While line-haul trucks are expected to have higher emission rates reflecting larger engine displacements, the CARB idling limit does not have separate targets based on engine characteristics. These findings indicate that future NO_x emission regulations are needed to ensure that idling emissions are accounted for and reductions are incentivized.

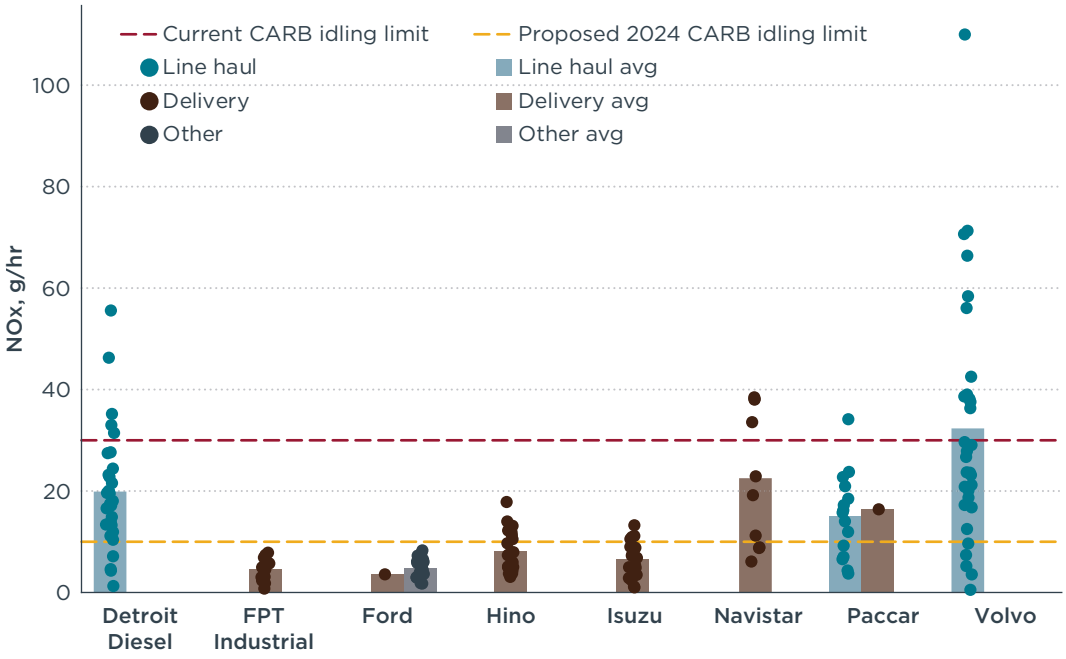


Figure 13 Idling NO_x emission rates (g/hr). The bars represent manufacturer average idle rate values for the entire PEMS dataset. Each dot represents the average idle emissions rate for each PEMS test.

10 The CARB idling limit is an optional NO_x standard for HDVs, which can also be fulfilled by implementation of an engine shutdown system that automatically turns off the engine after 300 seconds of continuous idling. The idling limit does not discriminate on engine size, flowrate, etc., and all HDVs must meet the limit in the absence of an automatic engine shutdown strategy.

CONCLUSIONS AND POLICY RECOMMENDATIONS

This report presents a baseline evaluation of NO_x emissions in the United States from on-road hheavy-duty vehicles with diesel engines certified to meet the EPA 2010 emission standards. PEMS data from the manufacturer-run HDIUT program made available by the EPA was the sole source of data for this report. Data from 160 PEMS tests and 26 unique engine families certified under the 0.2 g/bhp-hr FTP standard and the 0.3 g/bhp-hr NTE standard were studied. This evaluation provides an understanding of real-world NO_x emissions, highlighting conditions that contribute significantly to the NO_x inventory.

We compared the NO_x emissions evaluated with the NTE protocol to calculate total route NO_x emissions. The NTE method resulted in average NO_x emissions of 0.18 g/bhp-hr. This NTE compliance evaluation value was calculated, on average, based on less than 10% of the total PEMS data available. Comparatively, total route NO_x for all vehicles tested was 0.34 g/bhp-hr with the exclusion of idle operating conditions and 0.42 g/bhp-hr with idle.

We also analyzed the emissions data with no exclusions and looked at the impact of vehicle speed, vehicle type, and manufacturer on average NO_x emissions. In general, we found:

- On average, NO_x emissions increase to more than five times the certification limit in urban driving.
- Line-haul trucks have the highest urban NO_x emissions at almost seven times the engine certification limit
- Heavy-duty trucks spend, on average, 41%–55% of their total operational time in driving conditions slower than 25 mph, which is typically found in urban settings.
- On average, 40%–44% of total NO_x from HDVs is emitted during urban driving at speeds of less than 25 mph.
- NO_x emissions at levels at or below the 0.3 g/bhp-hr NTE limit during urban driving were found on 13 of 160 tests, corresponding to eight engine families produced by five manufacturers. Only four of the 160 tests had urban NO_x emissions at or below the 0.2 g/bhp-hr engine certification limit.

Based on these findings, we put forward the following policy recommendations for the development of future NO_x standards for heavy-duty engines:

- **A more stringent heavy-duty engine NO_x emissions standard is justified** based on emissions data showing a significant gap between real-world and engine-certified emissions.
- **Adoption of a supplemental low-load cycle with a corresponding emissions standard is necessary** to provide level ground for future development and adoption of emission controls for urban driving.
- **Adoption of a new in-use testing evaluation protocol that purposely targets the most challenging conditions for NO_x control is needed.** The current NTE protocol rejects more than 90% of the data captured during in-use tests. The new evaluation tool should focus on evaluating emissions where NO_x control is more challenging, such as low vehicle speed and engine load. The adoption of a low-load cycle also calls for an in-use protocol that evaluates data captured under such operating condition.

» **Controlling idling emissions requires a separate emissions limit and metric.** A work-specific limit for idling NO_x emissions is not appropriate during near-zero work conditions. A limit based on NO_x emissions rate, analogous to CARB's low NO_x idling standard, would be suitable for this application. Meeting an idle emissions limit would also incentivize compliance with CARB's proposed low-load cycle.

REFERENCES

Anderson, John. (2018, March). *Expectations for actual Euro 6 vehicle emissions*. Presented at the Concawe - Ricardo.

Anenberg, S. C., Henze, D. K., Tinney, V., Kinney, P. L., Raich, W., Fann, N., ... Kuylenstierna, J. C. I. (2018). Estimates of the global burden of ambient $PM_{2.5}$, ozone, and NO_2 on asthma incidence and emergency room visits. *Environmental Health Perspectives*, *126*(10), 107004. https://doi.org/10.1289/EHP3766

Besch, M. C. (2018). *Real-world emissions from heavy-duty in-use testing program*. Presented at the 8th Annual PEMS Conference & Workshop.

Bishop, G. A. (2015). On-road heavy-duty vehicle emissions monitoring system. *Environmental Science & Technology*, *49*(3), 1639–1645. https://doi.org/10.1021/es505534e

Bishop, G. A. (2019). Three decades of on-road mobile source emissions reductions in South Los Angeles. *Journal of the Air & Waste Management Association*, *69*(8), 967–976. https://doi.org/10.1080/10962247.2019.1611677

Burnett, R., Chen, H., Szyszkowicz, M., Fann, N., Hubbell, B., Pope, C. A., ... Spadaro, J. V. (2018). Global estimates of mortality associated with long-term exposure to outdoor fine particulate matter. *Proceedings of the National Academy of Sciences*, *115*(38), 9592–9597. https://doi.org/10.1073/pnas.1803222115

CARB. (2016). *2016 state strategy for the state implementation plan for federal ozone and $PM_{2.5}$ standards (State SIP Strategy)*. Retrieved from https://ww3.arb.ca.gov/planning/sip/2016sip/2016sip.htm

CARB. (2017). CEPAM: 2016 SIP - Standard emission tool, emission projections. Retrieved from https://www.arb.ca.gov/app/emsinv/fcemssumcat2016.php

CARB. (2019). *California Air Resources Board staff current assessment of the technical feasibility of lower NO_x standards—White paper*.

Code of Federal Regulations (CFR) (2014) Section § 86.1370 - Not-To-Exceed test procedures 65 FR 59961, Oct. 6, 2000, as amended at 66 FR 5188, Jan. 18, 2001; 70 FR 40441, July 13, 2005; 75 FR 68457, Nov. 8, 2010; 77 FR 34146, June 8, 2012. Redesignated and amended at 79 FR 23705, Apr. 28, 2014

Code of Federal Regulations (CFR) (2005). Subpart T - Manufacturer-Run In-Use Testing Program fo Heavy-Duty Diesel Engines Section § 86.1912 - How do I determine whether an engine meets the vehicle-pass criteria? 70 FR 34619, June 14, 2005, as amended at 73 FR 13450, Mar. 13, 2008; 75 FR 68460, Nov. 8, 2010

Dixit, P., Miller, J. W., Cocker, D. R., Oshinuga, A., Jiang, Y., Durbin, T. D., & Johnson, K. C. (2017). Differences between emissions measured in urban driving and certification testing of heavy-duty diesel engines. *Atmospheric Environment*, *166*, 276–285. https://doi.org/10.1016/j.atmosenv.2017.06.037

Duncan, A., & Hamady, F. (2019). A comprehensive study of manufacturers in-use testing data collected from heavy-duty diesel engines using portable emissions measurement system (PEMS). *CRC Real World Emissions Workshop*. U.S. EPA.

EPA. (2017). *Progress report vehicle engine & compliance activities*. U.S. Environmental Protection Agency - Compliance Division.

EPA. (2018a). Cleaner trucks initiative. *Regulations for emissions from vehicles and engines*. Retrieved from https://www.epa.gov/regulations-emissions-vehicles-and-engines/cleaner-trucks-initiative

EPA. (2018b). *EPA Acting Administrator Wheeler launches cleaner trucks initiative*. Retrieved from https://www.epa.gov/newsreleases/epa-acting-administrator-wheeler-launches-cleaner-trucks-initiative

EPA. (2018c). Progress cleaning the air and improving people's health. Retrieved from EPA website: https://www.epa.gov/clean-air-act-overview/progress-cleaning-air-and-improving-peoples-health

EPA. (2018d). EPA News Releases. EPA Announces Largest Voluntary Recall of Medium- and Heavy-Duty Trucks. Office of Air and Radiation (OAR). https://www.epa.gov/newsreleases/epa-announces-largest-voluntary-recall-medium-and-heavy-duty-trucks

EPA. (2019a). EPA activities for cleaner air. Retrieved from San Joaquin Valley website: https://www.epa.gov/sanjoaquinvalley/epa-activities-cleaner-air

EPA (2019b). Manufacturer-run in-use testing program data for heavy-duty diesel engines.

Grigoratos, T. (2019). Real world emissions performance of heavy-duty Euro VI diesel vehicles. *Atmospheric Environment*, *201*, 348–359. https://doi.org/10.1016/j.atmosenv.2018.12.042

Heroy-Rogalski, K., Lemieux, S., & Robertson, W. (2019). CARB heavy-duty low NO_x program. *NGO –ARB HDV NO_x Strategy Session*. California Air Resources Board.

Kotz, A. J., & Kelly, K. J. (2019). *MOVES activity updates using fleet dna data: Interim report* (No. NREL/TP-5400--70671, 1491376; p. NREL/TP-5400--70671, 1491376). https://doi.org/10.2172/1491376

Lee, A. D., Cain, J. E. O., Avila, J., Lemieux, S. C., Castillo, I., Fofie, E., … Berdahl, S. (2019). Comparison of CARB HDIUC and manufacturer-run HDIUT programs. *CRC Real World Emissions Workshop*. California Air Resources Board.

McCaffery, C. (2019). *Real-world gaseous and particulate emissions from GDI vehicles with and without catalyzed GPFs*. Presented at the CRC Real World Emissions. CRC Real World Emissions.

Mendoza-Villafuerte, P. (2017). NO_x, NH_3, N_2O and PN real driving emissions from a Euro VI heavy-duty vehicle. Impact of regulatory on-road test conditions on emissions. *Science of The Total Environment*, *609*, 546–555. https://doi.org/10.1016/j.scitotenv.2017.07.168

Quiros, D. C., Thiruvengadam, A., Pradhan, S., Besch, M., Thiruvengadam, P., Demirgok, B., … Hu, S. (2016). Real-world emissions from modern heavy-duty diesel, natural gas, and hybrid diesel trucks operating along major California freight corridors. *Emission Control Science and Technology*, *2*(3), 156–172. https://doi.org/10.1007/s40825-016-0044-0

Sandhu, G. S., & Sonntag, D. (2019). *Updates to MOVES heavy duty running exhaust rates: diesel, gasoline, and natural gas*. 53.

Spears, M. (2018). Needs and opportunities for reducing real world NO_x emissions from heavy-duty on- highway engines. *28th CRC Real World Emissions Workshop*. Truck & Engine Manufacturers Association.

Rodriguez, F. & Posada, F. (2019). *Future heavy-duty emission standards: An opportunity for international harmonization*. Retrieved from the International Council on Clean Transportation https://theicct.org/publications/ future-hdv-standards-harmonization

Transport Topics (2019) "Cummins Top Overall Supplier of Class 8 Diesel Engines in 2018." https://www.ttnews.com/articles/cummins-top-overall-supplier-class-8-diesel-engines-2018

Turner, M. C., Jerrett, M., Pope, C. A., Krewski, D., Gapstur, S. M., Diver, W. R., ... Burnett, R. T. (2016). Long-term ozone exposure and mortality in a large prospective study. *American Journal of Respiratory and Critical Care Medicine*, *193*(10), 1134–1142. https://doi.org/10.1164/rccm.201508-1633OC

APPENDIX A

The analysis presented in the body of this report excludes all Cummins non-credit engines available in the HDIUT dataset. This was done because all their diesel engines families tested under the HDIUT program were either covered by voluntary recalls or were natural gas engines. The HDIUT dataset contained 18 tests from two engine families manufactured by Cummins that are part of a voluntary recall and 11 PEMS from two CNG engine families.

This appendix presents NO_x emissions values from Cummins diesel and CNG engines excluded from the analysis. This is relevant as Cummins market share is one of the largest in the United States, where it supplies engines for four of every ten class-8 trucks sold in the United States (Transport Topics, 2019). The data from the recalled engines provides an indication of what NOx emission values can be found in HDVs experiencing SCR deterioration. The CNG results provide a comparison point on NO_x emissions between stoichiometric CNG engines and diesel.

Figure A1 shows NO_x emissions by vehicle speed bins for Cummins diesel recalled engines and Cummins CNG engines compared against the fleet average diesel values presented in this report. Recalled engines emit almost two times more than the fleet average during low speed driving. Comparatively, vehicles with CNG engines emit 40% lower during low speed driving. Total route values compared to the average diesel values are also two times higher for recalled engines while CNG engines are roughly half.

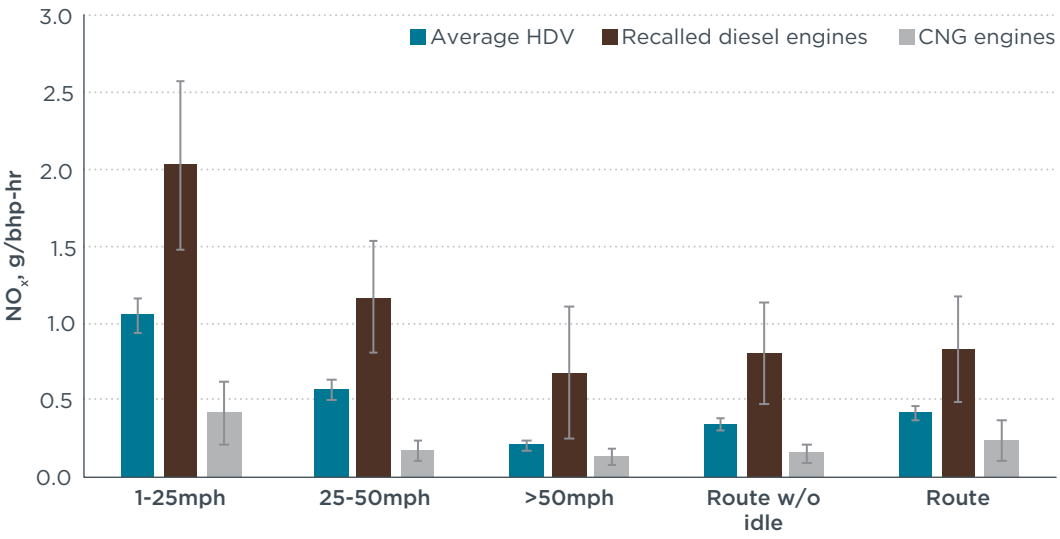


Figure A1. NO_x emissions by vehicle speed bin from the HDIUT dataset. Only non-credit engines. Includes recalled Cummins diesel engines, Cummins CNG engines, and average HDV values as shown in Figure 6. Whiskers represent the 95% confidence interval of the mean

APPENDIX B

This appendix presents a description for each of the 160 PEMS tests used in the analysis. All the tests correspond to vehicles with heavy-duty diesel engines certified to meet the EPA 0.2 g/bhp-hr NO_x emissions standard. Engines certified under the NO_x credit provision program were excluded. Engine families that are the subject of recalls were excluded from this analysis.

Table B1 describes the vehicles tested, engine characteristics, vehicle driving conditions, environmental conditions, and emission results as reported in the HDIUT tests and as calculated by our team. The HDIUT emissions results listed here show the results of the NTE evaluation process and are part of the data available from all PEMS tests studied.

Table B 1. Summary of PEMS Tests

Engine Manufacturer	Test Selection Year	Test Date	Engine Family	Engine Model	Engine Year	Engine Disp (L)	Max Power (hp)	Engine Hours	Vehicle Type	Max Weight (lbs)	Vehicle Miles	Test Duration (secs)	NTE Avg NO_x (g/hp-hr)	NTE Avg CO_2 (g/hp-hr)	Number of NTEs	Total NTE Duration (secs)	Test Duration (secs)	Amb Temp C	Avg Speed	Avg %Power	Avg Exh T[C]	Route NO_x (g/hp-hr)	Route CO_2 (kg/hp-hr)
Detroit Diesel	2011	20120410	ADDXH14.8EED	D472903-------	2010	14.8	475		Line haul	61000	68876	46980	0.11	477	47	2709	42775	19.6	31.9	0.22		0.236	0.516
Detroit Diesel	2011	20120412	ADDXH14.8EED	D472903-------	2010	14.8	475		Line haul	66000	67038	35450	0.17	478	71	3650	33505	19.6	29.1	0.18		0.326	0.546
Detroit Diesel	2011	20120925	ADDXH14.8EED	D472903	2011	14.8	475		Line haul	93107	104796	45217	0.11	493	61	3575	28704	20.1	25.3	0.20		0.294	0.545
Detroit Diesel	2011	20120926	ADDXH14.8EED	D472903	2011	14.8	475		Line haul	78130	5000	45979	0.09	505	95	4900	28612	23.5	35.1	0.25		0.212	0.545
Detroit Diesel	2011	20120927	ADDXH14.8EED	D472903	2011	14.8	475		Line haul	86155	115883	45544	0.10	502	50	2695	31960	19.7	27.7	0.19		0.367	0.545
Detroit Diesel	2012	20130813	ADDXH14.8EED	D472903	2010	14.8	475		Line haul	81000	173235	45162	0.18	453	59	3137	30572	21.9	31.0	0.20	260	0.331	0.480
Detroit Diesel	2012	20131030	ADDXH14.8EED	D472903	2010	14.8	475		Line haul	77238	173538	43024	0.13	457	57	3337	35569	24.4	32.0	0.22	233	0.251	0.520
Detroit Diesel	2012	20131031	ADDXH14.8EED	D472903	2010	14.8	475		Line haul	72272	217707	53955	0.15	455	124	7475	42906	23.3	35.3	0.24	246	0.293	0.510
Detroit Diesel	2012	20131103	ADDXH14.8EED	D472903	2010	14.8	475		Line haul	64825	180115	40568	0.09	484	31	1806	33842	12.7	28.1	0.17	208	0.389	0.550
Detroit Diesel	2013	20140806	ADDXH14.8EED	D472903	2010	14.8	475		Line haul	63365	245549	40308	0.24	478	80	4276	25565	23.8	37.4	0.22	239	0.406	0.510
Detroit Diesel	2014	20150922	CDDXH12.8FED	D471927	2013	12.8	450		Line haul	53000	160472	29355	0.04	498	28	2040	27508	21.0	30.8	0.21	234	0.215	0.496
Detroit Diesel	2014	20150923	CDDXH12.8FED	D471927	2013	12.8	450		Line haul	32000	164307	35541	0.03	504	33	2368	34538	21.1	30.0	0.21	232	0.220	0.489
Detroit Diesel	2014	20150924	CDDXH12.8FED	D471927	2013	12.8	450		Line haul	47227	168205	33501	0.09	495	30	1647	32617	21.1	34.0	0.22	245	0.167	0.485
Detroit Diesel	2014	20151120	CDDXH12.8FED	D471927	2013	12.8	370		Line haul	58577	206168	33857					33067	3.4	33.5	0.30	231	0.214	0.488
Detroit Diesel	2014	20151124	CDDXH12.8FED	D471927	2013	12.8	370		Line haul	58577	215650	25252					24586	7.4	27.6	0.23	260	0.443	0.475
Detroit Diesel	2015	20171102	CDDXH12.8FED	D471903	2012	12.8	470		Line haul	80000	424550	44353					43720	16.4	33.5	0.29	238	0.281	0.494
Detroit Diesel	2015	20171107	CDDXH12.8FED	D471903	2012	12.8	470		Line haul	80000	312937	32796	0.95	498	69	2070	30822	9.6	28.5	0.23	218	0.322	0.521
Detroit Diesel	2015	20171109	CDDXH12.8FED	D471903	2012	12.8	470		Line haul	80000	300904	73672					71067	3.4	29.0	0.23	207	0.465	0.486
Detroit Diesel	2015	20171114	CDDXH12.8FED	D471903	2012	12.8	380		Line haul	80000	246817	40169					39406	8.5	15.5	0.09	192	1.639	0.734
Detroit Diesel	2015	20171116	CDDXH12.8FED	D471903	2012	12.8	380		Line haul	80000	196360	40052					39228	5.0	12.5	0.09	155	2.008	0.625
Detroit Diesel	2016	20170511	EDDXH14.8EAD	D472906	2014	15.8	475		Line haul	56000	186709	27866	0.14	485	26	1056	25276	15.6	32.2	0.16	213	0.494	0.534
Detroit Diesel	2016	20170515	EDDXH14.8EAD	D472906	2014	15.8	475		Line haul	56000	228238	26777	0.06	475	34	1804	25911	21.2	28.4	0.14	212	0.489	0.552
Detroit Diesel	2016	20170518	EDDXH14.8EAD	D472906	2014	15.8	475		Line haul	56000	196523	23207	0.09	479	57	3139	22002	32.6	34.8	0.18	222	0.238	0.545
Detroit Diesel	2016	20170523	EDDXH14.8EAD	D472906	2014	15.8	475		Line haul	56000	228349	39199	0.01	492	94	5562	34265	22.5	50.8	0.26	283	0.015	0.524
Detroit Diesel	2016	20170525	EDDXH14.8EAD	D472906	2014	15.8	475		Line haul	56000	256589	40704	0.01	487	106	6291	37896	16.7	49.8	0.27	278	0.007	0.511
Detroit Diesel	2017	20180710	EDDXH12.8FED	D471927	2014	12.8	410		Line haul	80000	313551	43947	0.20	494	151	12927	41006	32.7	40.6	0.29	283	0.335	0.519
Detroit Diesel	2017	20180712	EDDXH12.8FED	D471927	2014	12.8	410		Line haul	80000	345273	41952	0.18	490	60	3509	40842	28.1	24.5	0.19	228	0.422	0.547
Detroit Diesel	2017	20180718	EDDXH12.8FED	D471927	2014	12.8	410		Line haul	80000	428243	28893	0.15	570	84	6529	28210	23.6	34.7	0.24	243	0.342	0.601
Detroit Diesel	2017	20180719	EDDXH12.8FED	D471927	2014	12.8	410		Line haul	80000	439060	25364	0.12	590	68	3735	24847	28.2	33.5	0.21	256	0.316	0.636
Detroit Diesel	2017	20180723	EDDXH12.8FED	D471927	2014	12.8	410		Line haul	80000	452591	21575	0.16	572	33	1677	18425	22.8	38.0	0.23	255	0.301	0.618
Ford	2013	20140716	CFMXH06.7A24	4V	2012	6.7	300	3034	Other	19000	54089	52249	0.13	671	20	1058	47617	22.0	11.3	0.07		0.521	0.763
Ford	2013	20140718	CFMXH06.7A24	4V	2012	6.7	300	1604	Other	11100	30843	29605	0.01	677	4	207	25098	21.9	26.3	0.11	210	0.467	0.727
Ford	2013	20140722	CFMXH06.7A24	4V	2012	6.7	300	1775	Other	15500	65237	23019	0.07	685	8	286	21765	26.9	30.4	0.13	239	0.407	0.732
Ford	2013	20141023	CFMXH06.7A24	4V	2012	6.7	300	571	Other	8500	47047	22683					21533	11.9	26.5	0.08	195	1.490	0.814
Ford	2015	20160726	CFMXH06.7B23	4VF	2012	6.7	300	19	Other	19500	86975	19371	0.10	628	24	1280	16645	26.5	35.9	0.17	260	0.144	0.696
Ford	2015	20160812	CFMXH06.7B23	4VF	2012	6.7	300	23	Other	19500	82481	22392	0.05	654	3	115	19380	32.0	26.4	0.11	201	0.390	0.733
Ford	2015	20160818	CFMXH06.7B23	4VF	2012	6.7	300	1280	Other	19500	84929	26591	0.02	688	28	1057	25411	26.0	46.7	0.20	274	0.128	0.746
Ford	2015	20160920	CFMXH06.7B23	4VF	2012	6.7	300	83	Other	16500	88736	23663	0.04	594	10	418	17855	25.1	35.8	0.14		0.287	0.663
Ford	2015	20160922	CFMXH06.7B23	4VF	2012	6.7	300	689	Other	16500	73651	30949	0.07	628	37	2911	28432	27.7	24.6	0.12		0.370	0.702
Ford	2011	20121003	BFMXH06.7B23	BFMXH067B23	2011	6.7	303	547	Other	19500	8619	29739	0.01	598	11	418	24772	24.3	16.4	0.06	159	0.457	0.869
Ford	2011	20121004	BFMXH06.7B23	BFMXH067B23	2011	6.7	303	1306	Other	19500	16875	32379	0.01	586	4	167	18923	25.6	21.6	0.09	222	0.260	0.753
Ford	2011	20121002	BFMXH06.7B23	BFMXH067B23	2011	6.7	303	2018	Other	19500	39606	29658	0.05	600	22	954	20325	21.8	31.6	0.14	215	0.197	0.708
Ford	2011	20120829	BFMXH06.7B23	BFMXH067B23	2011	6.7	303	96	Other	19500	4435	22924	0.02	547	4	137	22309	27.0	19.4	0.10	261	0.219	0.704
Ford	2017	20180620	CFMXH06.7B23	4VF	2012	6.7	300		Delivery	19500	74080	26512		579	7	483	25889	26.3	23.8	0.12	242	0.200	0.643
Ford	2017	20180628	CFMXH06.7B23	4VF	2012	6.7	300		Other	19500	67858	25848	0.10	586	19	920	20698	26.4	32.3	0.16	257	0.280	0.624
Ford	2017	20180727	CFMXH06.7B23	4VF	2012	6.7	300		Other	19500	48196	29258	0.06	566	11	744	27872	21.9	20.8	0.12	280	0.368	0.631
Ford	2017	20180822	CFMXH06.7B23	4VF	2012	6.7	300		Other	19500	55488	27031	0.08	561	9	364	26295	21.9	29.7	0.12	225	0.270	0.652
Ford	2017	20180911	CFMXH06.7B23	4VF	2012	6.7	300		Other	19500	49372	25543	0.04	576	10	543	24074	19.9	23.5	0.11	194	0.235	0.644

Engine Manufacturer	Test Selection Year	Test Date	Engine Family	Engine Model	Engine Year	Engine Disp (L)	Max Power (hp)	Engine Hours	Vehicle Type	Max Weight (lbs)	Vehicle Miles	Test Duration (secs)	NTE Avg NO_x (g/hp-hr)	NTE Avg CO_2 (g/hp-hr)	Number of NTEs	Total NTE Duration (secs)	Test Duration (secs)	Amb Temp C	Avg Speed	Avg %Power	Avg Exh T[C]	Route NO_x (g/hp-hr)	Route CO_2 (kg/hp-hr)
FPT Industrial	2013	20150417	CFPXH03.0F1B	F1C	2012	3	156		Delivery	16000	26187	43695	0.44	678	55	6019	25228	31.9	31.6	0.32	261	0.685	0.694
FPT Industrial	2013	20150430	CFPXH03.0F1B	F1C	2012	3	156		Delivery	16000	34302	32716	0.67	619	21	1087	14795	35.9	22.5	0.23	248	0.935	0.604
FPT Industrial	2013	20150616	CFPXH03.0F1B	CFPXH030F1B	2012	3	156		Delivery	16000	62622	24843	0.70	523	32	1992	23688	24.6	17.2	0.18	227	1.267	0.551
FPT Industrial	2013	20150619	CFPXH03.0F1B	CFPXH030F1B	2012	3	156		Delivery	16000	14427	29763	0.36	557	51	4177	27272	24.2	18.3	0.21	249	0.900	0.584
FPT Industrial	2013	20150623	CFPXH03.0F1B	CFPXH030F1B	2012	3	156		Delivery	16000	124128	31788	0.43	553	62	4725	29485	20.8	27.6	0.32	276	0.620	0.576
FPT Industrial	2013	20150714	CFPXH03.0F1B	CFPXH030F1B	2012	3	156		Delivery	16000	82596	30546	0.75	543	13	702	25951	24.5	16.4	0.19	236	0.840	0.518
FPT Industrial	2013	20150630	CFPXH03.0F1B	CFPXH030F1B	2012	3	156		Delivery	16000	23068	34260	0.19	572	52	5100	30645	20.9	23.1	0.27	248	0.486	0.596
FPT Industrial	2013	20150702	CFPXH03.0F1B	CFPXH030F1B	2012	3	156		Delivery	16000	4960	26551	0.47	563	45	3500	24755	19.3	21.4	0.22	250	0.714	0.602
FPT Industrial	2013	20150708	CFPXH03.0F1B	CFPXH030F1B	2012	3	156		Delivery	16000	84248	26420	0.28	550	64	7952	22264	19.2	36.6	0.36	283	0.534	0.569
FPT Industrial	2013	20150710	CFPXH03.0F1B	CFPXH030F1B	2012	3	156		Delivery	16000	85678	24311	0.20	521	64	7214	23413	24.0	35.8	0.36	304	0.455	0.543
FPT Industrial	2018	20180911	GFPXH03.0F1B	F1C	2013	3	161	191	Delivery	19500	6352	43603	0.04	458	175	15376	31120	19.5	46.6	0.53	322	0.079	0.466
FPT Industrial	2018	20181128	GFPXH03.0F1B	F1C	2013	3	153	2077	Delivery	19500	59785	23759	0.55	478	21	954	18456	-1.9	33.2	0.39	262	0.616	0.438
FPT Industrial	2018	20181129	GFPXH03.0F1B	F1C	2013	3	153	1948	Delivery	19500	63091	23188	0.30	409	55	3566	16936	1.0	45.0	0.48	286	0.262	0.399
FPT Industrial	2019	20190205	GFPXH03.0F1B	F1C	2016	3	153	899	Delivery	19500	28189	27674	0.02	449	47	3993	14405	24.1	28.0	0.35	267	0.076	0.423
FPT Industrial	2019	20190206	GFPXH03.0F1B	F1C	2016	3	153	401	Delivery	19500	15580	15550	0.06	544	39	6518	13203	23.7	52.7	0.67	382	0.079	0.560
FPT Industrial	2019	20190207	GFPXH03.0F1B	F1C	2016	3	153	519	Delivery	19500	14601	17134	0.03	459	54	5402	12481	20.7	44.5	0.52	331	0.066	0.475
Hino	2013	20141216	CHMXH05.1JTP	J05E	2012	5	210		Delivery	19500	22681	29642	0.06	387	15	1130	22571	10.2	29.2	0.29	240	0.217	0.473
Hino	2013	20141217	CHMXH05.1JTP	J05E	2012	5	210		Delivery	19500	35102	23846	0.04	406	21	989	16814	14.7	29.9	0.29	246	0.220	0.489
Hino	2013	20141215	CHMXH05.1JTP	J05E	2012	5	210		Delivery	19500	32784	30813	0.05	424	28	1517	20270	12.8	21.9	0.23	223	0.184	0.491
Hino	2013	20141214	CHMXH05.1JTP	J05E	2012	5	210		Delivery	19500	22940	28069	0.06	473	97	6463	21821	11.2	47.1	0.51	302	0.155	0.489
Hino	2013	20141213	CHMXH05.1JTP	J05E	2012	5	210		Delivery	19500	61638	25479	0.10	471	58	3153	23287	7.2	29.8	0.29	233	0.279	0.501
Hino	2014	20150722	BHMXH07.7JVC	J08E	2011	7.7	220		Delivery	25950	119258	22743	0.21	512	8	366	21678	27.3	28.6	0.26	205	0.320	0.496
Hino	2014	20150723	BHMXH07.7JVC	J08E	2011	7.7	220		Delivery	25950	159955	22332	0.05	506	6	308	19795	27.0	27.5	0.27	211	0.256	0.500
Hino	2014	20150721	BHMXH07.7JVC	J08E	2011	7.7	220		Delivery	25950	107438	22269	0.05	483	2	100	20963	30.3	24.6	0.24	205	0.293	0.474
Hino	2014	20150720	BHMXH07.7JVC	J08E	2011	7.7	220		Delivery	25950	77880	22320	0.18	506	10	611	19718	31.3	24.0	0.24	185	0.379	0.473
Hino	2012	20131206	BHMXH07.7JVC	J08E	2011	7.7	220		Delivery	25950	32576	20098	0.26	569	16	749	17570	10.4	28.4	0.19		0.377	0.816
Hino	2014	20150724	BHMXH07.7JVC	J08E	2011	7.7	220		Delivery	25950	113738	20709					16329	27.7	25.3	0.24	211	0.348	0.481
Hino	2017	20181010	FHMXH05.1JTP	J05E	2015	5.1	210		Delivery	14500	16497	21264	0.08	543	18	847	17775	25.8	36.1	0.20	232	0.115	0.557
Hino	2017	20181017	FHMXH05.1JTP	J05E	2015	5.1	210		Delivery	14500	76545	45111	0.03	544	8	341	24119	19.2	16.7	0.10	169	0.568	0.641
Hino	2017	20181016	FHMXH05.1JTP	J05E	2015	5.1	210		Delivery	14500	81812	38228					19435	15.9	13.4	0.09	170	0.839	0.641
Hino	2017	20181015	FHMXH05.1JTP	J05E	2015	5.1	210		Delivery	14500	28564	43056	0.13	584	6	347	21031	21.1	15.9	0.11	194	0.602	0.650
Hino	2016	20170708	EHMXH07.7JVB	J08E	2014	7.7	260		Delivery	33000	17684	22176	0.13	492	12	486	19919	30.7	24.4	0.17	243	0.351	0.574
Hino	2016	20170710	EHMXH07.7JVB	J08E	2014	7.7	260		Delivery	33000	40019	25739	0.10	503	35	1606	21474	32.2	31.4	0.20	250	0.359	0.593
Hino	2016	20170707	EHMXH07.7JVB	J08E	2014	7.7	260		Delivery	33000	34689	28700	0.09	498	62	2706	24504	28.7	40.7	0.26	270	0.288	0.571
Hino	2016	20170711	EHMXH07.7JVB	J08E	2014	7.7	260		Delivery	33000	60629	25638	0.07	494	43	1746	21505	32.7	35.0	0.22	254	0.263	0.579
Hino	2016	20170712	EHMXH07.7JVB	J08E	2014	7.7	260		Delivery	33000	47611	26310	0.04	507	34	1319	19605	32.6	38.6	0.24	264	0.305	0.586
Hino	2017	20181011	FHMXH05.1JTP	J05E	2015	5.1	210		Delivery	14500	43860	46329	0.32	565	2	85	18000	26.9	19.2	0.12	186	0.601	0.625
Isuzu	2011	20120514	ASZXH05.23FA	4HK1-TC	2010	5.2	210		Delivery	19500	59809	35715	0.08	542	27	1372	23330	21.5	22.7	0.19	223	0.185	0.595
Isuzu	2011	20120507	ASZXH05.23FA	4HK1-TC	2010	5.2	210		Delivery	19500	32655	45698	0.05	546	33	1882	20049	16.1	22.5	0.17	183	0.181	0.637
Isuzu	2011	20120515	ASZXH05.23FA	4HK1-TC	2010	5.2	210		Delivery	19500	33177	38499	0.06	508	29	1639	26878	24.7	17.6	0.15	202	0.153	0.576
Isuzu	2013	20140617	BSZXH05.23FA	4HK1-TC	2011	5.2	210		Delivery	19500	30200	26579	0.04	529	28	1797	22094	30.0	17.7	0.13	200	0.241	0.636
Isuzu	2011	20121019	ASZXH05.23FA	4HK1-TC	2010	5.2	210		Delivery	19500	40397	27320	0.13	692	7	285	25592	13.1	9.5	0.06	155	0.525	0.885
Isuzu	2011	20121022	ASZXH05.23FA	4HK1-TC	2010	5.2	210		Delivery	19500	40468	27491	0.07	703	8	390	23003	19.7	10.6	0.07	157	0.524	0.834
Isuzu	2011	20120521	ASZXH05.23FA	4HK1-TC	2010	5.2	210		Delivery	19500	72062	42256	0.08	528	53	3773	27661	17.8	26.2	0.19	187	0.311	0.674
Isuzu	2011	20120523	ASZXH05.23FA	4HK1-TC	2010	5.2	210		Delivery	19500	52922	29833	0.04	504	23	1189	13044	21.0	32.6	0.23	190	0.239	0.619
Isuzu	2013	20140219	BSZXH05.23FA	4HK1-TC	2011	5.2	210		Delivery	19500	40719	38864	0.04	595	28	3491	17758	23.2	25.3	0.19	212	0.239	0.656
Isuzu	2013	20140224	BSZXH05.23FA	4HK1-TC	2011	5.2	210		Delivery	19500	53760	40592	0.06	595	46	4389	21921	24.5	28.7	0.21	234	0.216	0.651
Isuzu	2013	20140226	BSZXH05.23FA	4HK1-TC	2011	5.2	210		Delivery	19500	37884	41428	0.03	605	26	2529	16959	21.8	22.8	0.18	212	0.230	0.662
Isuzu	2013	20140228	BSZXH05.23FA	4HK1-TC	2011	5.2	210		Delivery	19500	32978	36633	0.04	576	22	1084	19336	18.1	20.2	0.15	215	0.207	0.588
Isuzu	2016	20171030	ESZXH05.23FA	4HK1-TC	2013	5.2	210		Delivery	19500	73036	47486	0.20	667	2	102	28157	5.6	14.7	0.09	162	0.825	0.796
Isuzu	2016	20171103	ESZXH05.23FA	4HK1-TC	2013	5.2	210		Delivery	19500	70435	26642	0.18	576	20	1495	21272	9.2	29.6	0.24	252	0.310	0.620
Isuzu	2016	20180911	ESZXH05.23FA	4HK1-TC	2014	5.2	210		Delivery	19500	54398	32430	0.18	559	7	292	21387	26.1	19.7	0.13	209	0.442	0.643
Isuzu	2016	20180912	ESZXH05.23FA	4HK1-TC	2014	5.2	210		Delivery	19500	44674	30576	0.06	565	17	1013	19581	26.7	24.6	0.17	235	0.240	0.646
Isuzu	2016	20180913	ESZXH05.23FA	4HK1-TC	2014	5.2	210		Delivery	19500	53124	14506	0.19	584	3	108	12306	25.8	14.1	0.10	226	0.486	0.905
Isuzu	2016	20180914	ESZXH05.23FA	4HK1-TC	2014	5.2	210		Delivery	19500	53255	25552	0.12	551	9	659	10114	25.1	32.9	0.23	238	0.245	0.616

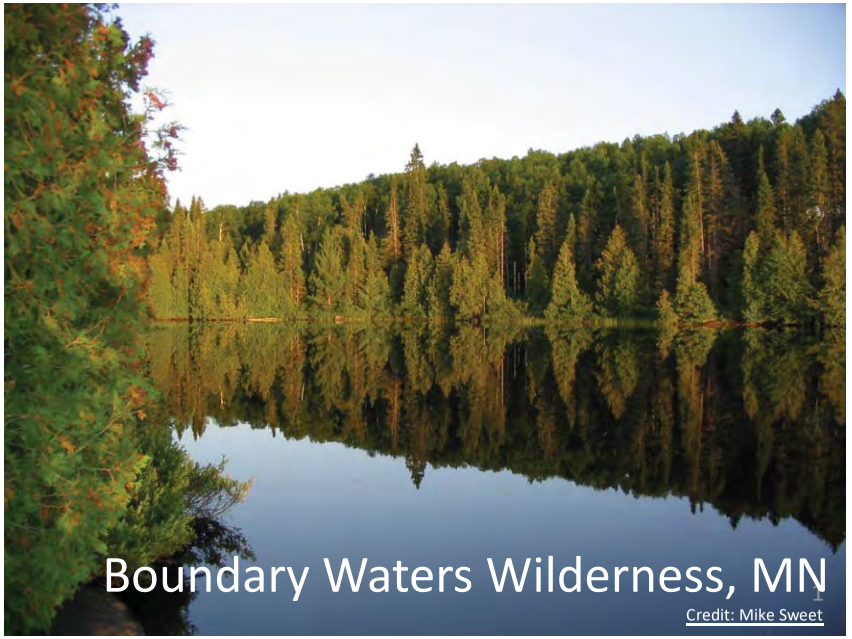
Engine Manufacturer	Test Selection Year	Test Date	Engine Family	Engine Model	Engine Year	Engine Disp (L)	Max Power (hp)	Engine Hours	Vehicle Type	Max Weight (lbs)	Vehicle Miles	Test Duration (secs)	NTE Avg NO_x (g/hp-hr)	NTE Avg CO_2 (g/hp-hr)	Number of NTEs	Total NTE Duration (secs)	Test Duration (secs)	Amb Temp C	Avg Speed	Avg %Power	Avg Exh T[C]	Route NO_x (g/hp-hr)	Route CO_2 (kg/hp-hr)
Navistar	2014	20150522	DNVXH07570SB	A475	2013	12.4	483	4577	Delivery	47400	191598	43722	0.03	587	150	10201	41850	19.5	46.0	0.24	253	0.125	0.622
Navistar	2014	20150528	DNVXH07570SB	A475	2013	12.4	483	4840	Delivery	47400	230123	38006	0.03	598	202	11301	36365	28.1	53.0	0.28	263	0.083	0.624
Navistar	2014	20150730	DNVXH07570SB	A410	2013	12.4	411	3650	Delivery	47400	97906	39174	0.02	572	57	4001	33576	27.6	31.4	0.20	254	0.188	0.631
Navistar	2014	20150731	DNVXH07570SB	A410	2013	12.4	411	3171	Delivery	47400	75118	31796	0.02	562	49	2330	29278	27.7	31.9	0.20	231	0.169	0.619
Navistar	2014	20150921	DNVXH07570SB	A410	2013	12.4	411	3171	Delivery	47400	75022	26517	0.08	585	24	1244	25443	18.1	26.0	0.15	212	0.438	0.674
Navistar	2016	20170725	FNVXH05700SA	S330	2015	9.3	330		Delivery	80000	32459	36996	0.04	517	16	645	36286	20.0	15.1	0.13	186	0.769	0.574
Navistar	2016	20171121	FNVXH05700SA	S330	2015	9.3	330		Delivery		58213	40877	0.02	585	31	1920	40104	7.6	15.0	0.15	188	0.638	0.621
Navistar	2016	20170801	FNVXH05700SA	S330	2015	9.3	330	1470	Delivery		21544	49184	0.11	513	64	4012	45782	26.5	13.3	0.12	184	0.823	0.583
Navistar	2016	20170919	FNVXH05700SA	S300	2015	9.3	300	1382	Delivery		48141	58936	0.03	562	101	6205	55412	24.8	17.9	0.16	204	0.521	0.630
Navistar	2016	20170922	FNVXH05700SA	S330	2015	9.3	300	1922	Delivery		50970	26211	0.01	563	63	3586	25461	25.9	21.1	0.19	213	0.284	0.602
Paccar	2010	20111001	APCRH12.9M01	MX 280 T	2010	12.9	395	2311	Line haul	75000	98956	31156					22349	24.0	45.9	0.26		0.184	0.477
Paccar	2010	20111003	APCRH12.9M01	MX 280 T	2010	12.9	405	2379	Line haul	68000	102141	19430			11	522	16421	19.9	45.0	0.25		0.081	0.475
Paccar	2010	20111006	APCRH12.9M01	MX 280 T	2010	12.9	405	896	Line haul	68000	34175	21753			32	2159	18433	15.1	44.1	0.28		0.087	0.482
Paccar	2010	20111010	APCRH12.9M01	MX 280 T	2010	12.9	405	1067	Line haul	68000	39544	31511			29	1667	22751	24.2	44.8	0.32		0.165	0.512
Paccar	2010	20111031	APCRH12.9M01	MX 360 T	2010	12.9	491	1013	Line haul	64000	23804	31581			22	1356	28354	6.5	28.5	0.20		0.377	0.544
Paccar	2015	20151113	BPCRH12.9M01	MX 360 T	2011	12.9	491	6268	Line haul	60000	268474	40878					8047	13.3	27.2	0.27	237	0.550	0.521
Paccar	2015	20160525	BPCRH12.9M01	MX 321 T	2011	12.9	449	11374	Line haul	79920	394653	32719	0.11	492	17	959	29375	29.1	28.0	0.20	251	0.364	0.537
Paccar	2015	20160606	BPCRH12.9M01	MX 360 T	2011	12.9	491	8723	Line haul	80000	370400	35750	0.06	491	127	10064	23662	16.9	50.5	0.32	253	0.179	0.503
Paccar	2015	20160609	BPCRH12.9M01	MX 360 T	2011	12.9	491	8927	Line haul	80000	368831	37480	0.10	495	45	3828	27442	27.4	32.7	0.20	250	0.302	0.530
Paccar	2015	20160811	BPCRH12.9M01	MX 340 T	2011	12.9	491	7559	Delivery	60000	308336	25698		511	2	81	16762	21.4	34.4	0.18	232	0.336	0.549
Paccar	2017	20180627	EPCRH12.9M01	MX-13 360 V	2014	12.9	482	8170	Line haul	99060	288943	34335	0.17	452	14	838	16447	19.4	22.6	0.16	211	0.611	0.500
Paccar	2017	20180628	EPCRH12.9M01	MX-13 360 V	2014	12.9	482	8176	Line haul	102560	289056	41616	0.02	451	5	252	20535	20.2	15.5	0.09	181	1.023	0.549
Paccar	2017	20180702	EPCRH12.9M01	MX-13 360 V	2014	12.9	482	6844	Line haul	105500	230438	40369	0.09	462	7	398	20609	19.3	24.1	0.13	219	0.897	0.566
Paccar	2017	20180703	EPCRH12.9M01	MX-13 360 V	2014	12.9	482	8041	Line haul	105500	282861	33389	0.03	482	3	237	19505	20.6	16.0	0.11	224	0.873	0.584
Paccar	2017	20181112	EPCRH12.9M01	MX-13 340 V	2014	12.9	455	9127	Line haul		332716	37165	0.02	458	76	6454	33912	16.5	38.7	0.28	233	0.171	0.466
Paccar	2017	20181115	EPCRH12.9M01	MX-13 340 V	2014	12.9	455	4685	Line haul	80000	260050	40799	0.08	474	60	5815	35766	12.5	30.4	0.24	219	0.390	0.485
Volvo	2014	20141029	BVPTH10.8S01	MP7-395C	2011	10.8	409	5069	Line haul	50000	203488	39938	0.41	502	2	72	38840	16.6	21.9	0.15	212	0.504	0.551
Volvo	2014	20141027	BVPTH10.8S01	MP7-395C	2011	10.8	409	5038	Line haul	50000	192844	39929	0.12	504	49	2066	38748	22.9	34.7	0.20	246	0.154	0.562
Volvo	2013	20140501	CVPTH16.1S01	D16H 500	2012	16.1	529	4019	Line haul	80000	166211	40095	0.08	536	54	2384	37053	17.7	38.6	0.20	257	0.162	0.593
Volvo	2013	20140513	CVPTH16.1S01	D16H 550	2012	16.1	579	3304	Line haul	80000	126711	33548	0.67	525	50	2246	32095	17.9	28.8	0.17	261	0.165	0.597
Volvo	2013	20140429	CVPTH16.1S01	D16H 500	2012	16.1	529	2140	Line haul	80000	78364	39511	0.03	550	62	3184	38369	8.7	37.2	0.22	249	0.218	0.592
Volvo	2013	20140515	CVPTH16.1S01	D16H 500	2012	16.1	529	5107	Line haul	102000	153331	40480	0.14	495	70	3893	39315	17.0	29.7	0.16	223	0.369	0.654
Volvo	2013	20140516	CVPTH16.1S01	D16H 500	2012	16.1	529	3255	Line haul	102000	106731	36802	0.20	538	44	2568	34078	20.6	33.7	0.18	230	0.322	0.660
Volvo	2011	20120626	AVPTH12.8S01	D13H 425	2010	12.8	435	5952	Line haul	80000	256633	43200					48953	21.6	13.5	0.12	171	0.765	0.460
Volvo	2012	20131120	AVPTH12.8S01	D13H 425	2010	12.8	435	8550	Line haul	107000	354249	43200	0.88	556	15	952	45817	1.2	40.6	0.40	246	0.640	0.615
Volvo	2012	20131001	AVPTH12.8S01	D13H 425	2010	12.8	435	3028	Line haul	60000	310226	43200	0.68	510	26	1407	46838	19.8	16.7	0.12	180	1.064	0.531
Volvo	2011	20120628	AVPTH12.8S01	D13H 425	2010	12.8	435	4986	Line haul	60000	217660	43200					43752	25.1	50.2	0.23	250	0.106	0.545
Volvo	2012	20131119	AVPTH12.8S01	D13H 425	2010	12.8	435	7430	Line haul	107000	314482	43200	0.22	489	14	719	49341	-0.3	37.5	0.33	226	0.504	0.523
Volvo	2011	20120510	AVPTH12.8S01	D13H 425	2010	12.8	438	1862	Line haul	77160	53485	32714					31704	14.9	30.1	0.18	220	0.357	0.536
Volvo	2012	20131126	AVPTH12.8S01	D13H 425	2010	12.8	438	4769	Line haul	78000	124864	32443					30423	2.2	21.4	0.17	179	1.286	0.499
Volvo	2011	20120508	AVPTH12.8S01	D13H 425	2010	12.8	438	697	Line haul	70000	21222	42492					41594	19.2	22.0	0.17	210	0.488	0.542
Volvo	2012	20131022	AVPTH12.8S01	D13H 425	2010	12.8	438	3408	Line haul	70000	103657	37603	1.12	513	6	288	36520	7.6	15.4	0.09	151	1.640	0.551
Volvo	2014	20141012	BVPTH10.8S01	D11H 385	2011	10.8	396	7923	Line haul	80000	344060	27183	0.22	491	26	1640	26464	15.1	47.6	0.38	270	0.130	0.512
Volvo	2014	20141015	BVPTH10.8S01	D11H 385	2011	10.8	396	8596	Line haul	80000	397935	31839	0.33	481	27	1675	30952	11.9	49.5	0.35	254	0.153	0.515
Volvo	2014	20141014	BVPTH10.8S01	D11H 385	2011	10.8	396	6124	Line haul	80000	253493	26386					25665	14.1	46.9	0.36	247	0.158	0.617
Volvo	2017	20180410	FVPTH10.8G01	MP7-395C	2015	10.8	403	9554	Line haul	65000	332822	39887	0.34	483	58	5805	37066	11.9	42.5	0.37	237	0.764	0.490
Volvo	2017	20180522	FVPTH10.8G01	MP7-395C	2015	10.8	403	7207	Line haul	88640	238632	49098	0.22	493	117	9122	44710	21.7	28.8	0.33	268	0.397	0.502
Volvo	2017	20180410	FVPTH10.8G01	MP7-395C	2015	10.8	403	7689	Line haul	65000	270151	44530	0.49	471	87	5347	39401	12.0	43.6	0.30	223	0.799	0.489
Volvo	2017	20180412	FVPTH10.8G01	MP7-395C	2015	10.8	403	5101	Line haul	65000	177991	42814	0.12	475	124	6102	40716	17.5	32.9	0.30	233	0.449	0.488
Volvo	2017	20180412	FVPTH10.8G01	MP7-395C	2015	10.8	403	5885	Line haul	65000	202762	41911	0.48	478	95	5584	37423	20.0	39.4	0.26	217	0.863	0.480
Volvo	2017	20180411	FVPTH10.8G01	MP7-395C	2015	10.8	403	6109	Line haul	65000	202583	45007	0.38	485	90	5666	41580	13.2	34.3	0.28	214	0.770	0.505
Volvo	2016	20170417	EVPTH12.8G01	D13J425	2014	12.8	435	4450	Line haul	79000	292385	43060	0.26	454	29	1575	34907	16.3	41.6	0.28	231	0.461	0.468
Volvo	2016	20170419	EVPTH12.8G01	D13J425	2014	12.8	435	8161	Line haul	79000	225317	43621	1.70	439	4	137	36047	10.9	39.4	0.29	235	0.458	0.454
Volvo	2016	20170719	EVPTH12.8G01	D13J425	2014	12.8	435	6881	Line haul	60000	229362	40598	0.15	443	91	4529	30889	27.0	41.5	0.22	238	0.335	0.473
Volvo	2016	20161108	EVPTH12.8G01	D13J425	2014	12.8	435	7013	Line haul	50000	302670	13459					13123	9.4	52.5	0.34	244	0.601	0.467
Volvo	2016	20161115	EVPTH12.8G01	D13J425	2014	12.8	435	5042	Line haul	80000	212457	39938	0.31	453	3	445	16600	4.5	49.4	0.40	254	0.367	0.467
Volvo	2016	20161115	EVPTH12.8G01	D13J425	2014	12.8	435	5186	Line haul	80000	218119	23188	0.12	436	38	2034	20605	17.7	50.0	0.40	273	0.255	0.450

Mobile Source Modeling: Strengths, Weaknesses, and Opportunities for Improvement

Mark Janssen
Emissions Director

LADCO Regional Air Quality Meeting – Spring 2019
April 17th, 2019

LADCO | LAKE MICHIGAN AIR DIRECTORS CONSORTIUM



State of Technology

- LADCO works directly with US EPA to build onroad and nonroad emissions inventories for modeling; we pool resources and expertise.
- MOVES is EPA's Java/MYSQL software for calculating onroad (cars/trucks) emissions.
- Needs fleet age/technology, activity, fuel, vehicle population and speed.
- LADCO has worked with Coordinating Research Council (CRC) to improve MOVES inputs like speed, age, and temporal profiles.

Coordinating Research Council (CRC)
Auto/Oil industry group

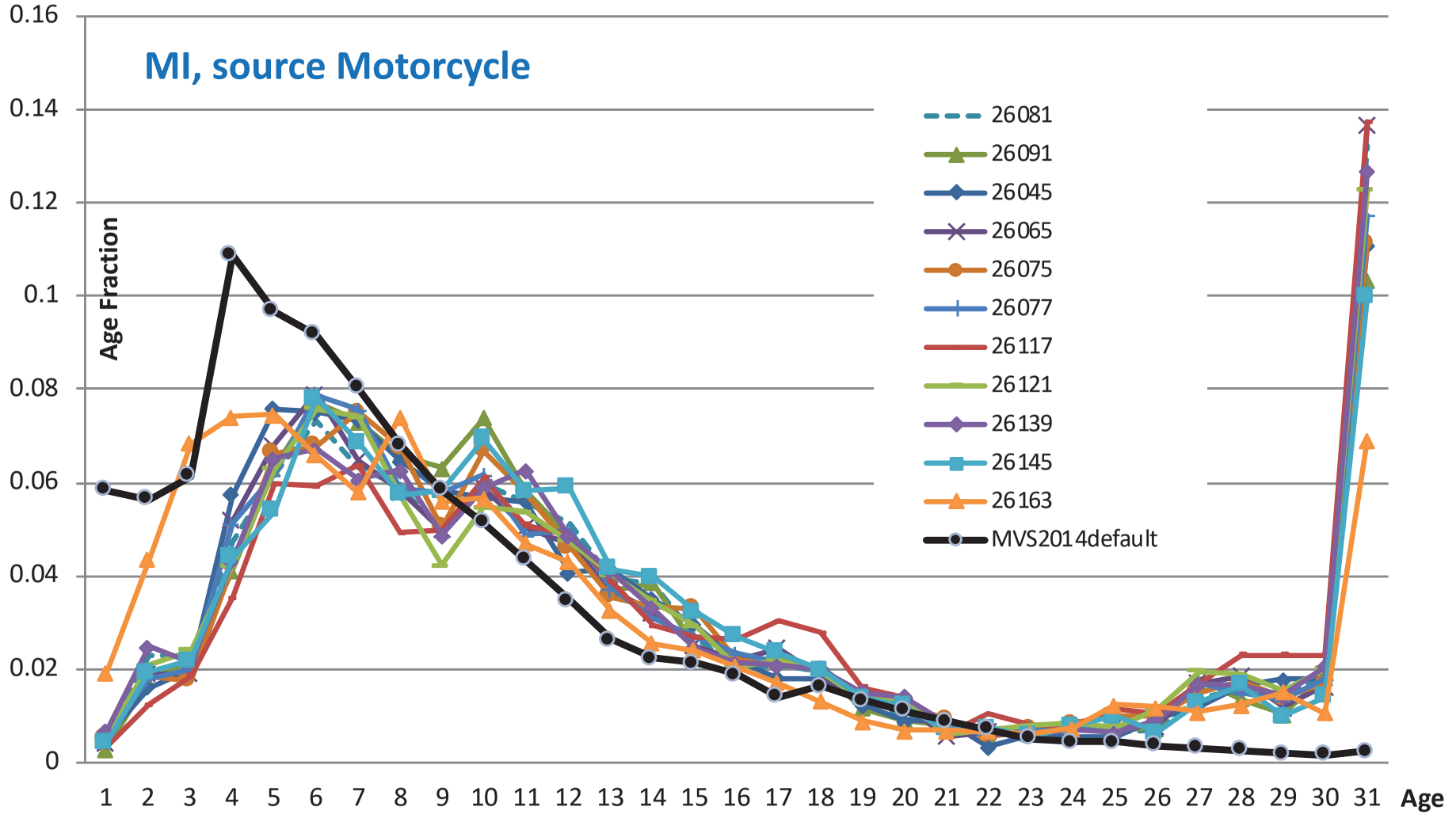
LADCO

LADCO led six projects through CRC to evaluate MOVES inputs

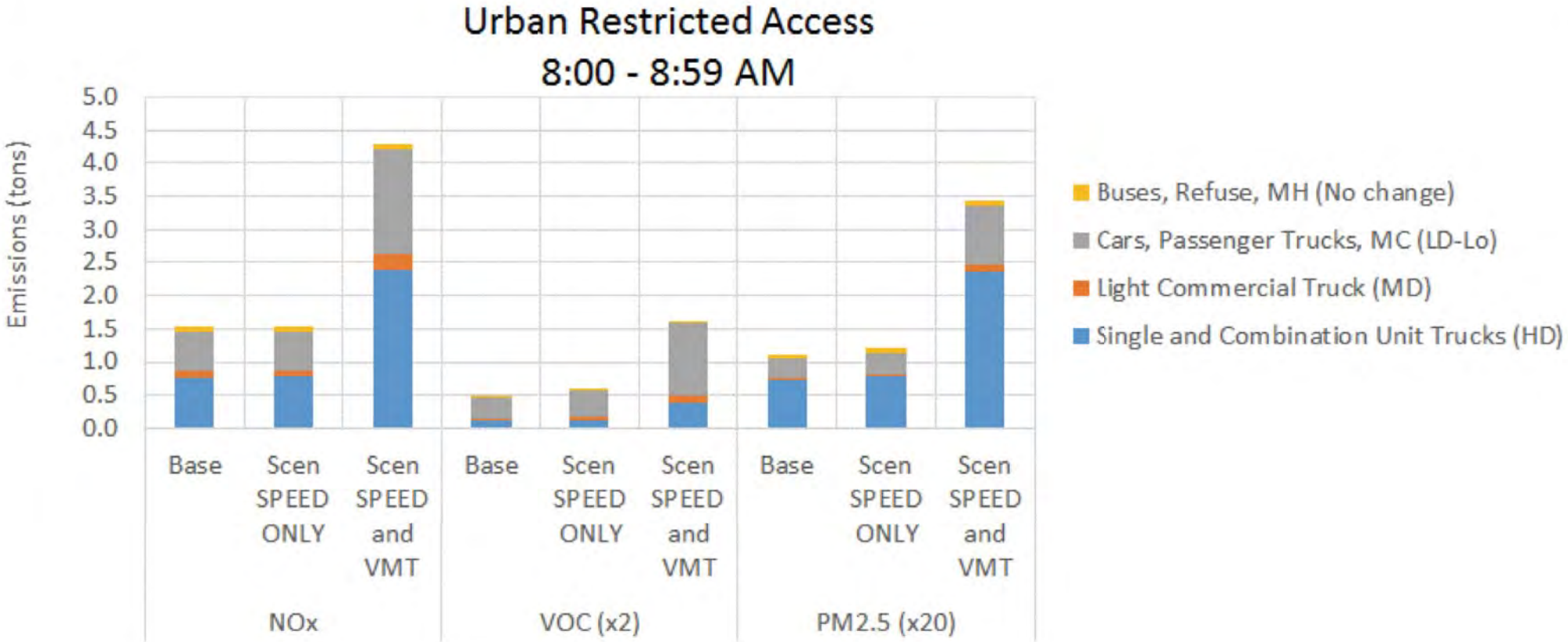
- **A-84**: Identified the input variables most likely to impact modeling and inventories: vehicle age, car/truck mix, vehicle count, speed distribution, road type distribution.
- **A-88:** National VIN decode for 2011, get vehicle counts. Use models to get passenger/heavy truck mix
- **A-100:** Use of telemetry data to improve MOVES inputs: speed distribution, temporal
- **A-103:** Improve cloud based computing
- **A-106:** Improved start information from telemetry
- **A-115:** Re-evaluate VIN decode techniques and update age distributions.
- **A-119:** (Pending) project to build modeling to remote sensing tools.

Age Distribution Example

(from CRC Project A-88)



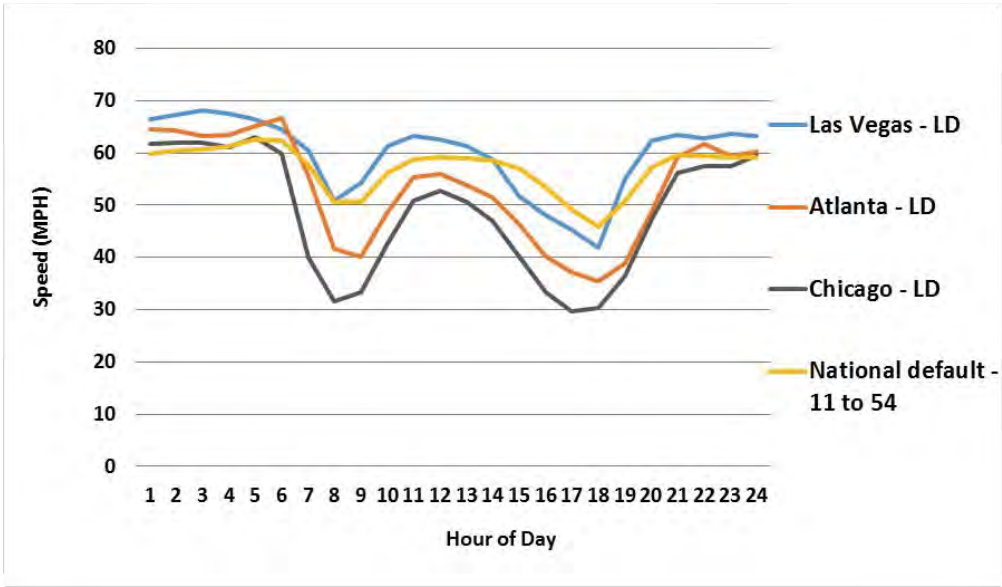
Chicago Telemetry Data



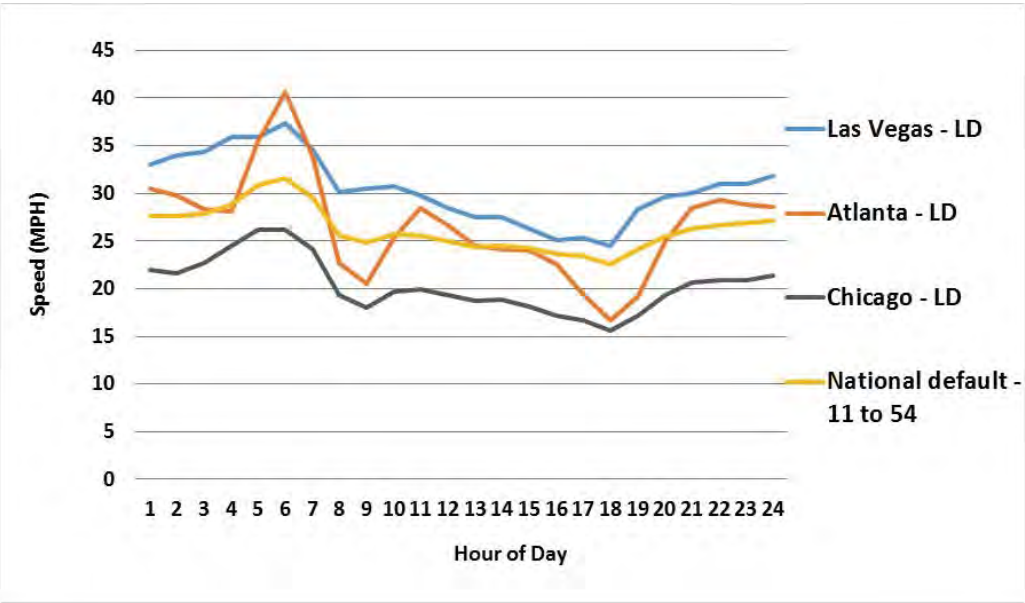
Telemetry Data Improvement

Weekday, LD Passenger Vehicles

Urban Restricted Roads

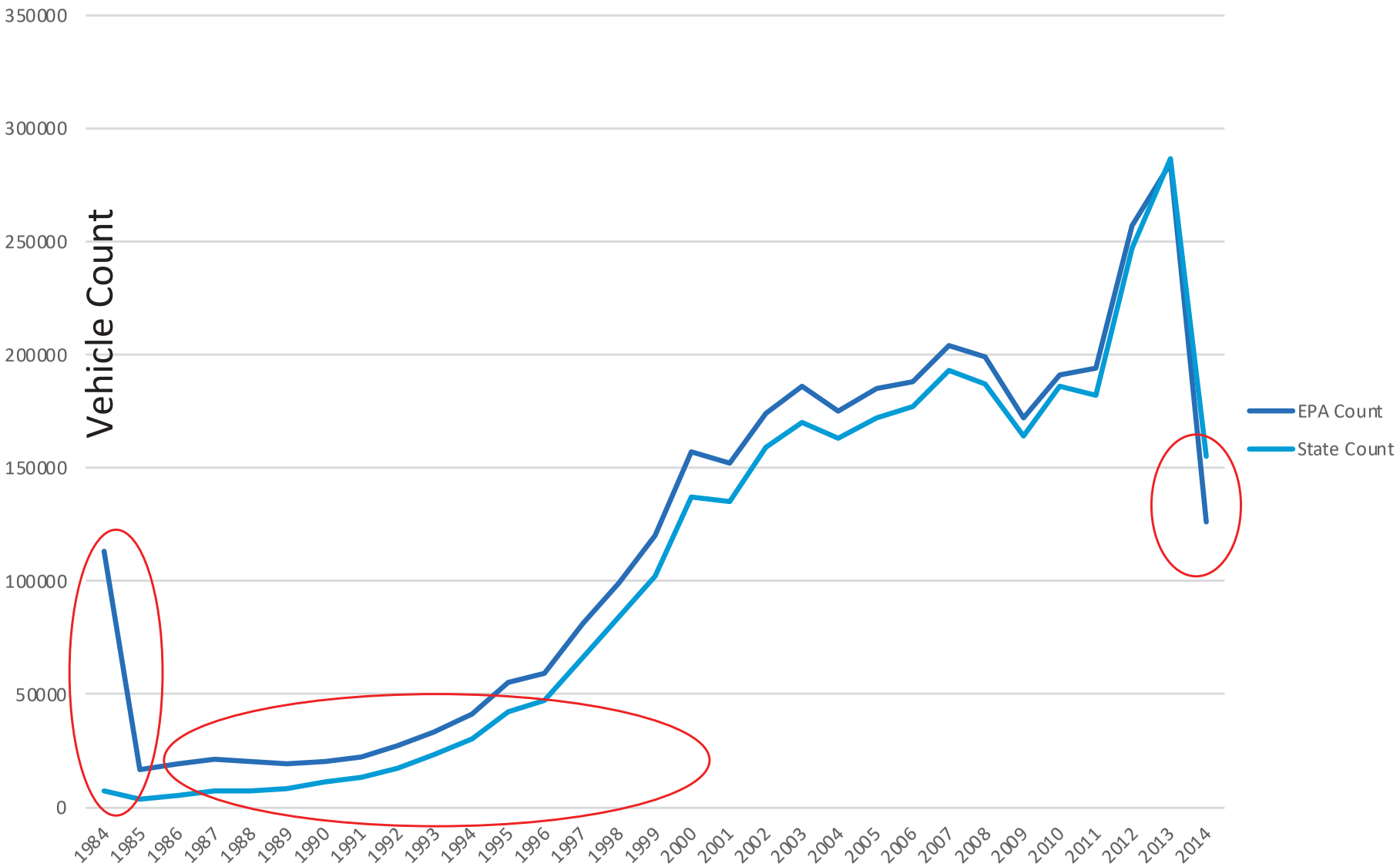


Urban Unrestricted Roads



2014 Vehicle Count by Age

Passenger Cars



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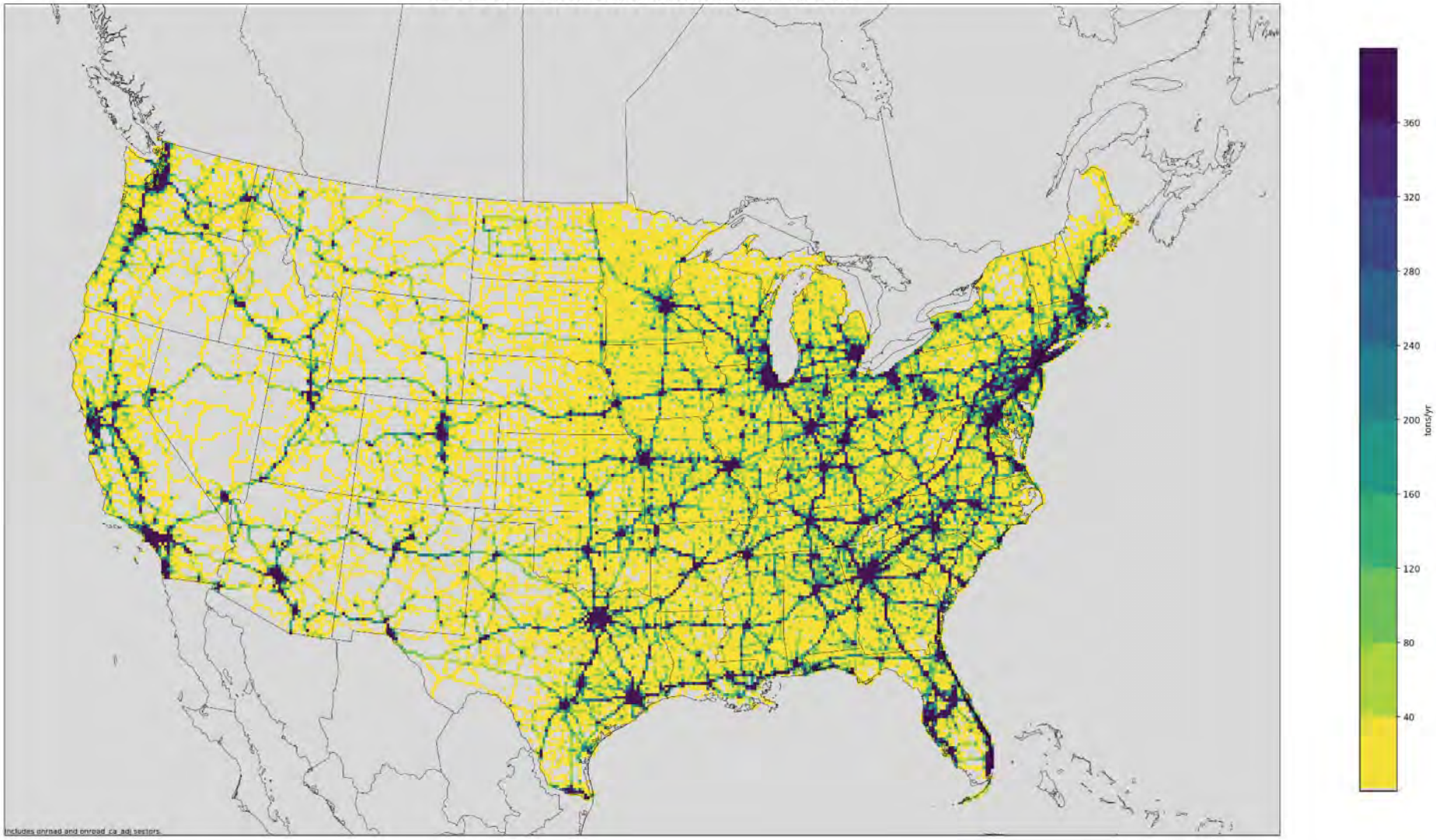
2016ff onroad annual : NOX



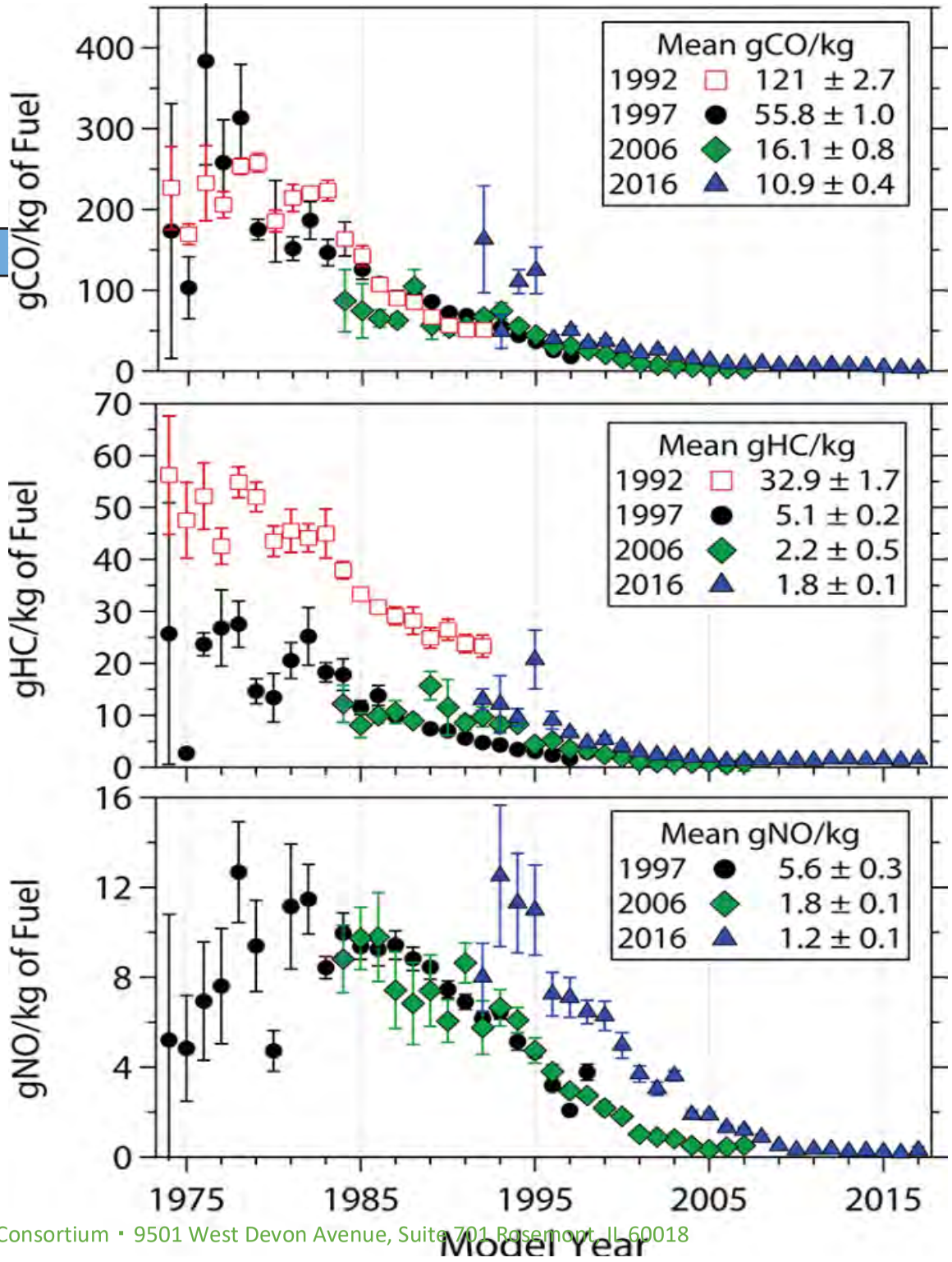
Onroad Mobile Data Strengths

- We continue work with states and cities to improve transportation data. This includes reflecting the differences between states and cities.
- Changes to these inputs do have a measureable effect on emissions.
- Awareness that even if inputs are good, will model produce realistic emissions.
  - Use satellite and remote sensing data to verify inventories and models
  - Compare ground based measurements and Inventories

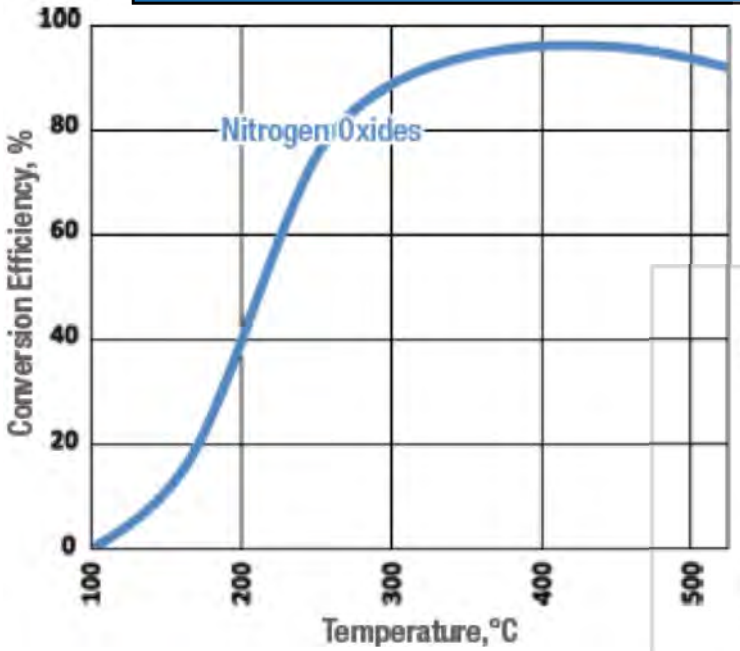
Data Weaknesses

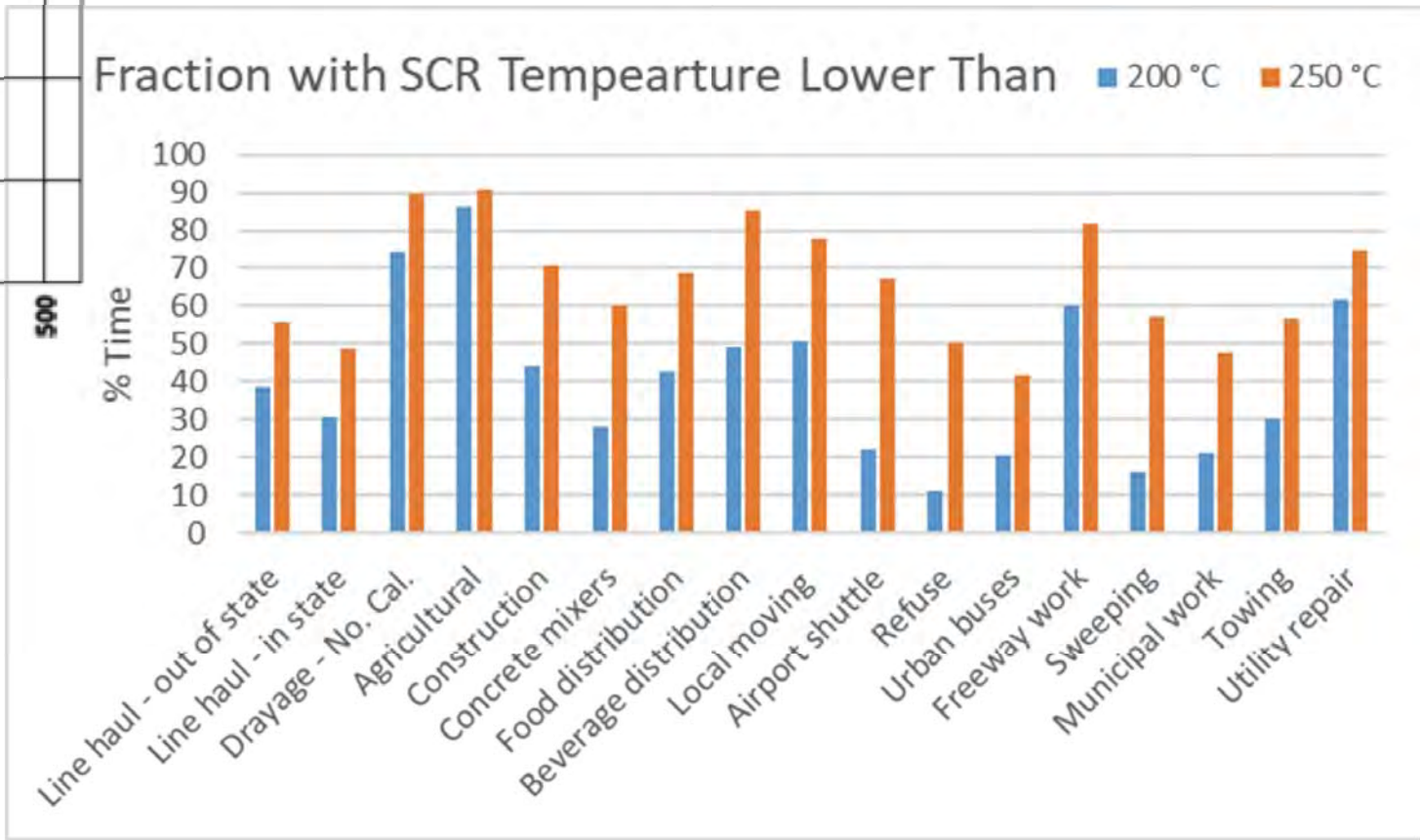
- Cars are becoming so clean that outliers define the category
- How does MOVES compare to real world data?
- MOVES is only as good as the drive cycles that it is based off. If drivers deviate from the predictable then we may see significant impact.

Historic and current emissions rates from in-use measurements



Heavy Duty Off Cycle Idling





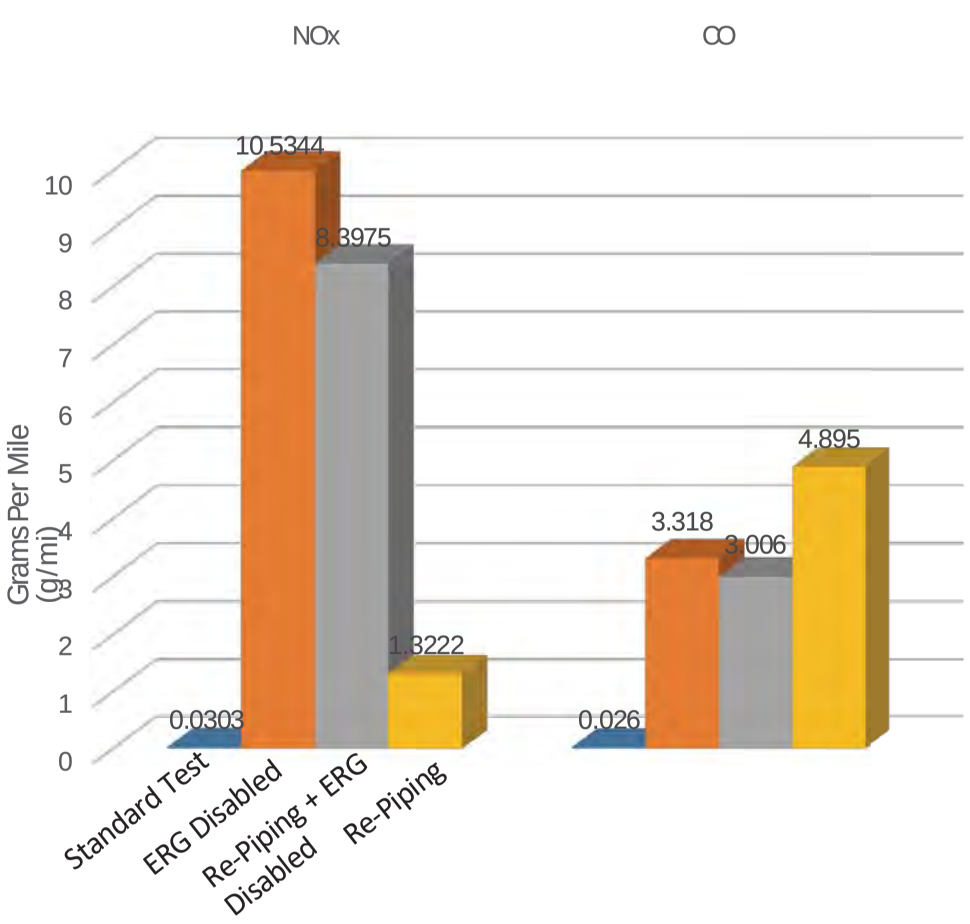
UC –Riverside Presentation to MARAMA Oct 20th, 2017

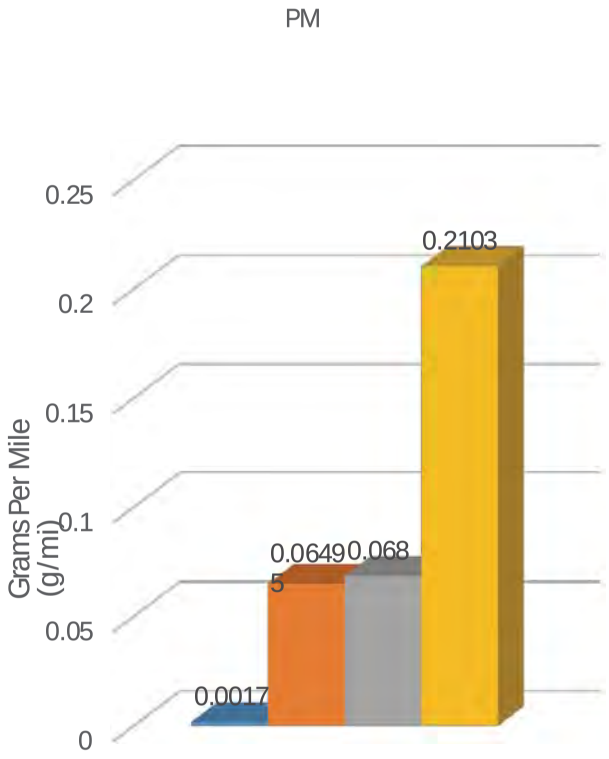
Heavy Duty Defeat Device

- Aftermarket Defeat Device – “any part or component… where a principal effect…is to bypass, defeat, or render inoperative any device or element of design installed on or in a motor vehicle or motor vehicle…”

EPA Tuner Emissions Tests

Stock Calibration Equipment vs. Emissions Equipment wo/ Tuners



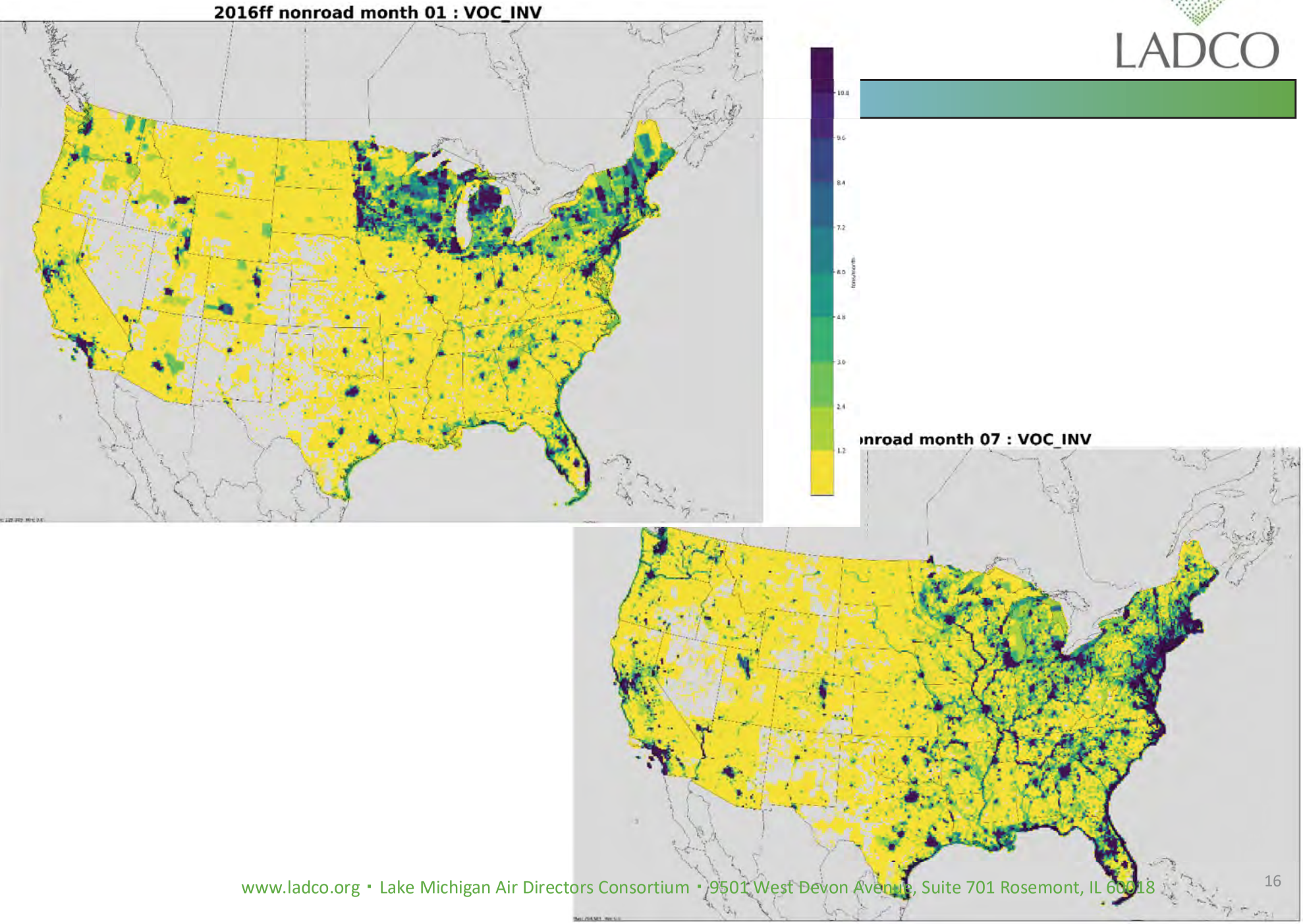


Source: Phil Brooks OECA presentation April 5th 2018

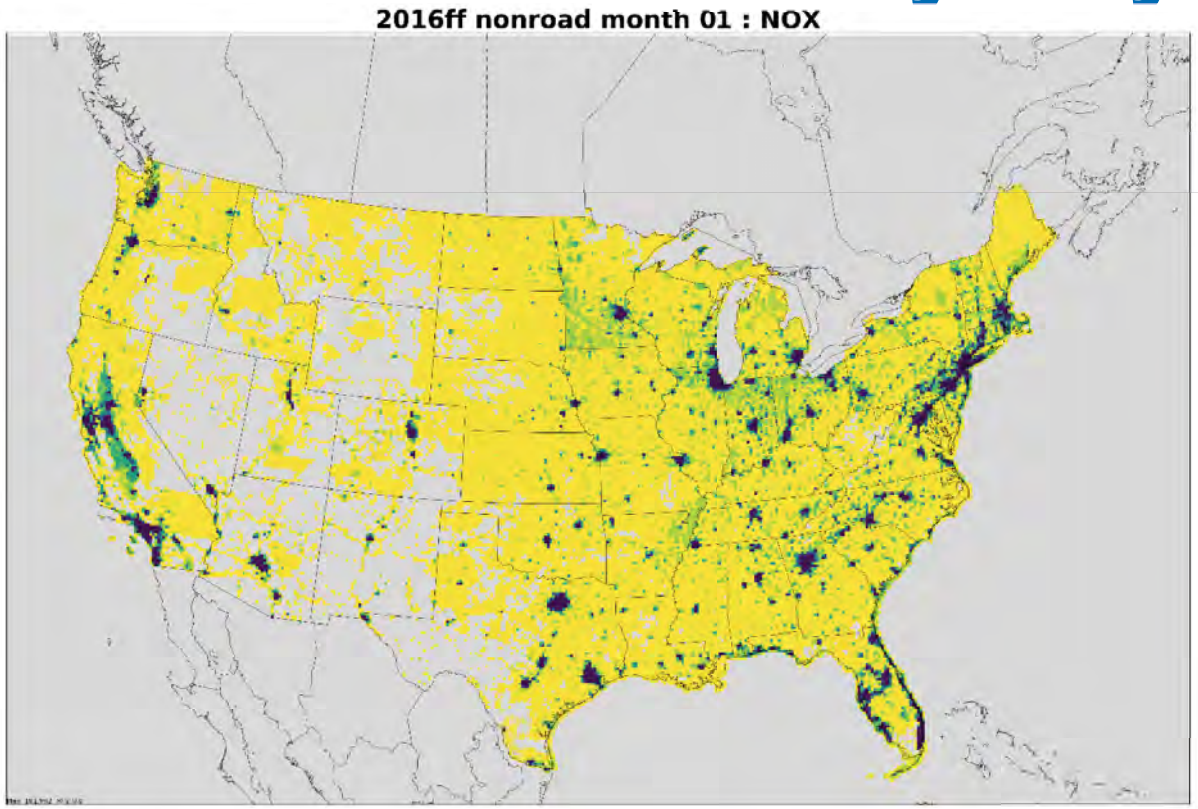
Nonroad Mobile Sources

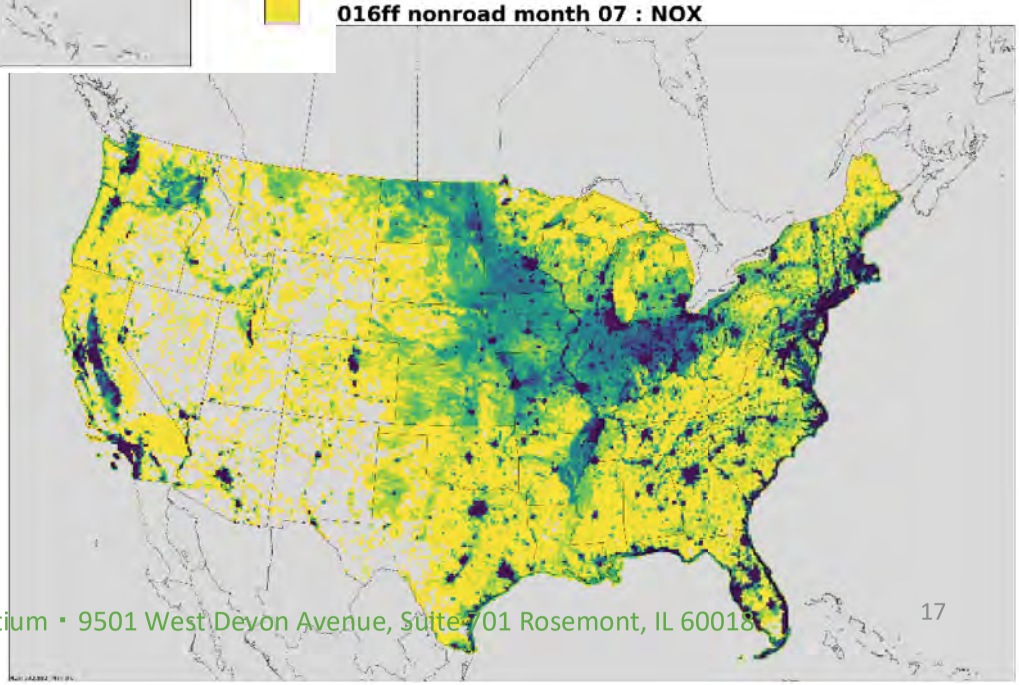
- EPA recently added nonroad source calculation into MOVES. The model creates emissions estimates for engines that do not operate on highways.
- Difficult to inventory because of diversity of sources, across many industries, low individual emissions, large composite emissions.
  - Recreational vehicles, all-terrain vehicles and off-road motorcycles;
  - Logging equipment, chain saws;
  - Agricultural equipment, tractors, combines;
  - Construction equipment, graders and back hoes;
  - Industrial equipment, forklifts and sweepers;
  - Residential and commercial lawn and garden equipment, tillers, leaf and snow blowers
  - Recreational and commercial marine vessels, power boats and oil tankers
  - Railway equipment, train and switching engines
  - Aircraft, Baggage handling equipment, jets and prop planes.

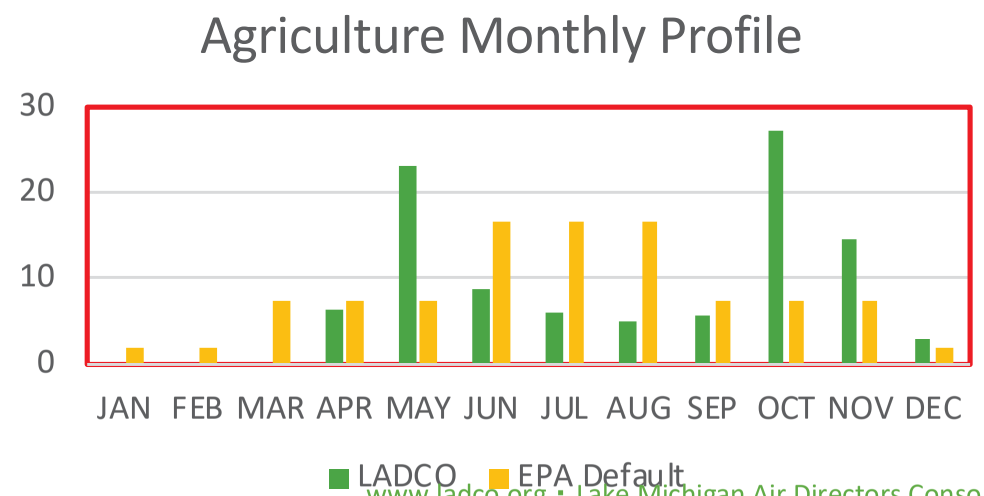
Nonroad January/July VOC

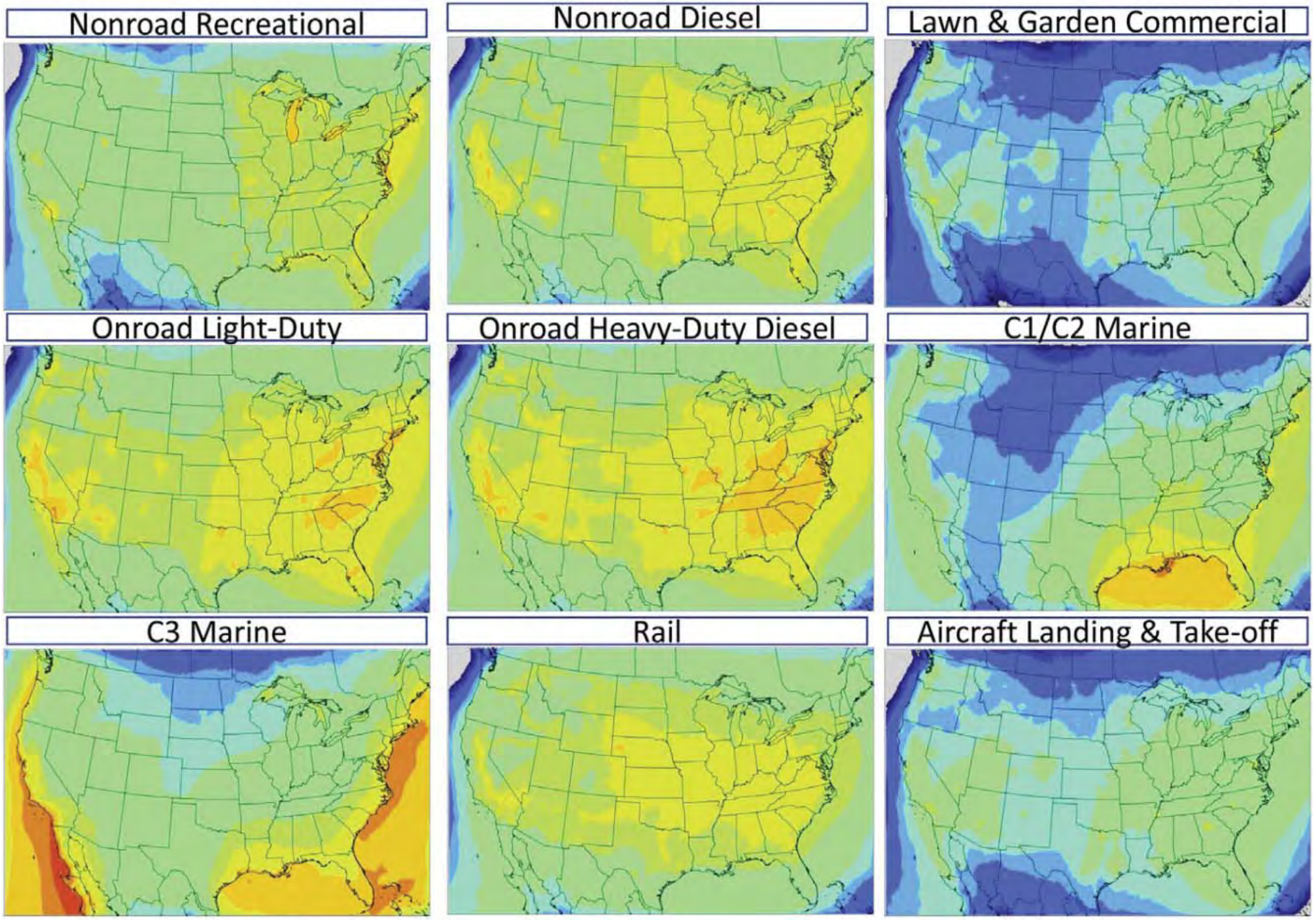


Nonroad January/July NOX







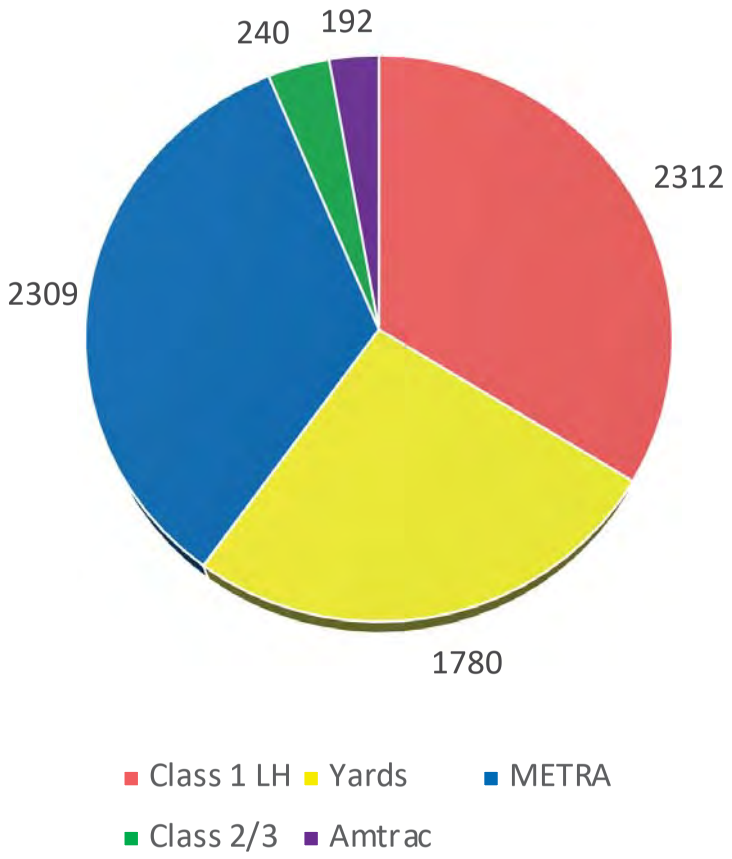


Mobile source contributions to ambient ozone and particulate matter in 2025, Margaret Zawackia,*, Kirk R. Baker, Sharon Phillips, Kenneth Davidson, Philip Wolfe

LADCO Improvements to Rail

- LADCO and IL,MI have put significant effort into rail inventories.
- Included passenger(Amtrak) and commuter(METRA) into inventories into national inventory for the first time ever.
- Use proprietary rail administration database of link level activity and we identified yards as points.
- METRA change is the most significant recent change. Metra's fleet is older, 10% of fleet will upgrade by 2023.

Cook County IL,
NOX Tons/Year



What Can We Do About Vehicle Emissions?

- VW Settlement
- Diesel Emissions Reduction Act (DERA)
- Heavy Duty Diesel Devices/Programming
- Heavy Duty Diesel Device Idling
- Work with EPA on national standards
- LADCO needs to characterize reductions in SIP

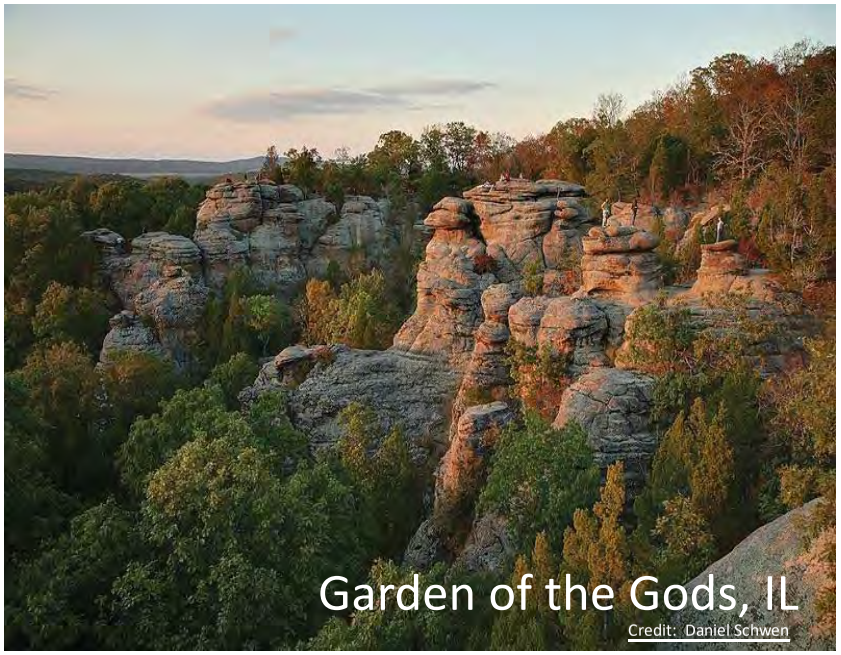
Questions and Contact

Mark Janssen

Emissions Director

janssen@ladco.org


Sleeping Bear Dunes, MI
Credit: NPS


Garden of the Gods, IL
Credit: Daniel Schwen

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Article

Characterizing Elevated Urban Air Pollutant Spatial Patterns with Mobile Monitoring in Houston, Texas

David J. Miller,* Blake Actkinson, Lauren Padilla, Robert J. Griffin, Katie Moore, P. Grace Tee Lewis, Rivkah Gardner-Frolick, Elena Craft, Christopher J. Portier, Steven P. Hamburg, and Ramón A. Alvarez

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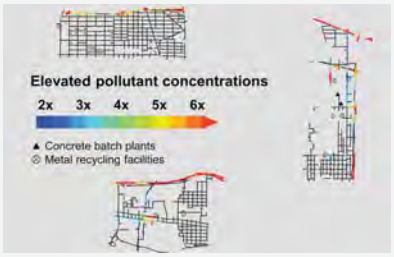
ABSTRACT: Diverse urban air pollution sources contribute to spatially variable atmospheric concentrations, with important public health implications. Mobile monitoring shows promise for understanding spatial pollutant patterns, yet it is unclear whether uncertainties associated with temporally sparse sampling and instrument performance limit our ability to identify locations of elevated pollution. To address this question, we analyze 9 months of repeated weekday daytime on-road mobile measurements of black carbon (BC), particle number (PN), and nitrogen oxide (NO, NO_2) concentrations within 24 census tracts across Houston, Texas. We quantify persistently elevated, intermittent, and extreme concentration behaviors at 50 m road segments on surface streets and 90 m segments on highways relative to median statistics across the entire sampling domain. We find elevated concentrations above uncertainty levels (±40%) within portions of every census tract, with median concentration increases ranging from 2 to 3× for NO_2, and >9× for NO. In contrast, PN exhibits elevated concentrations of 1.5−2× the domain-wide median and distinct spatial patterns relative to other pollutants. Co-located elevated concentrations of primary combustion tracers (BC and NO_x) near 30% of metal recycling and concrete batch plant facilities within our sampled census tracts are comparable to those measured within 200 m of highways. Our results demonstrate how extensive mobile monitoring across multiple census tracts can quantitatively characterize urban air pollution source patterns and are applicable to developing effective source mitigation policies.



1. INTRODUCTION

Urban air pollutant concentrations vary substantially from neighborhood blocks to city-wide scales,[1,2] impacting public health.[3−5] These patterns are driven in part by spatially heterogeneous primary particulate matter and nitrogen oxide (NO_x = NO + NO_2) sources, exposure to both of which is associated with increased mortality.[6] Black carbon (BC) and fine particle number (PN) serve as effective tracers for health-relevant, fresh combustion emissions, and PN is also a tracer for particle nucleation events.[7−9] Primary NO_x and volatile organic compounds (VOCs) also play key roles in photochemistry, serving as precursors to secondary fine particulate matter ($PM_{2.5}$) and ozone (O_3) formation.[10,11]

Approaches to quantify fine-scale (~kilometer) spatial patterns of urban air pollution and their source drivers include chemical transport and dispersion model simulations, satellite retrievals, and ground-based stationary monitoring. Modeling approaches provide attribution of spatial patterns to specific principal source sectors (e.g., Zhang et al.[12]), but have uncertainties at sub-kilometer scales associated with emission inventory spatial allocation and micro-meteorology representation.[13] Recent satellite retrieval approaches have pinpointed and quantified larger industrial and mobile urban sources, though once-daily, typically afternoon overpass observations must be oversampled over several months to achieve a kilometer-scale spatial resolution and could miss sources during other hours of the day.[14] Ground-based stationary sensor networks are capable of characterizing spatial patterns continuously at greater spatial density (sub-kilometer scales) than typical regulatory networks, but they are likely to miss local sources located between monitoring locations.[2,15]

As a complement to fixed sensor networks, ground-based mobile monitoring provides high spatial resolution observations, though temporally sparse at any individual location.[1,16−19] Mobile measurement studies are typically conducted at relatively small scales, including a few city blocks, individual neighborhoods, and major roadway transects.[20−22] These approaches have been valuable to assess the spatial representativeness of stationary monitors,[4,23] characterize temporal variations in spatial patterns along urban transects,[18] and evaluate vehicle emission inventories.[21] Mobile monitoring has also provided observations for inversion modeling using Lagrangian dispersion model footprints[13] and for building land-use regression exposure models.[8,9,24]

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Using multineighborhood, on-road mobile monitoring, Apte et al.[1] found that BC and NO_x concentrations can vary by >5× within individual city blocks and semi-quantitatively identified hotspot locations using the median concentrations from repeated drive passes through 30 m road segments. These temporally aggregated concentration estimates are typically reported with ±10 to ±30% precision, depending on the pollutant and study region,[1,19] yet are summarized using domain-wide average uncertainties and generally do not report instrument drifts over time or between mobile vehicles. Some studies also include an hourly temporal adjustment.[1,19,22,23] However, temporal concentration variability across multiple drive periods may contain valuable information on intermittent or extreme concentrations resulting from local sources and/or meteorological influences that are not captured by the median approach. It remains unclear whether elevated spatial concentration patterns across even larger multineighborhood domains can be quantified and distinguished from potentially large and spatially heterogeneous sampling uncertainties.

In this study, we examine the presence and persistence of elevated concentration patterns using fine-scale, multipollutant mobile monitoring data with large spatial coverage of the Houston, Texas metropolitan area. Houston is a large (~1620 km^2) urban area with diverse emission sources, including on-road and nonroad vehicles, oil refining and petrochemical operations,[25] large industrial[26] and hazardous air pollutant emissions,[27,28] a lack of zoning leading to intermingling of industrial and residential areas, and nonattainment designation with regards to Environmental Protection Agency O_3 standards.[29] Our approach focuses on mobile source tracers because of the close proximity of vehicle traffic to our on-road sampling, though point or area source plumes could also be captured using this approach if they are present at ground level. We apply descriptive statistical methods with comprehensive uncertainty analyses to characterize concentration behaviors of ~10,500 road segments across a nine-month period. We utilize our on-road concentration pattern observations to identify segments with elevated central tendency, high variability, and extreme concentrations relative to typical characteristics across the entire sampling domain, as well as individual census tracts. Then, we investigate their possible source drivers by examining the proximity of identified road segments to known source locations. Specifically, we conduct three transect case studies focusing on the local source and meteorological influences along road transects where multiple pollutants exhibit elevated concentrations. These analyses can help to improve the effectiveness of mobile monitoring strategies to evaluate fine-scale urban emission inventories, identify missing sources, and inform source mitigation policies.

2. MATERIALS AND METHODS

2.1. Instrumentation. We conducted on-road mobile measurements of BC, PN, NO, and NO_2 concentrations with two Google Street View vehicles (2009 Hyundai Santa Fe) equipped with a Global Positioning System (GPS) unit (Garmin 18×). $PM_{2.5}$, O_3, and carbon dioxide (CO_2) were also measured, though we do not use these species in this study as they are the subject of ongoing research. Mobile data were quality-controlled and quality-assured. Data were removed during instrument warm-ups, calibrations, and if conditions were outside an instrument's concentration or relative humidity operating range. Mobile measurements were synchronized to GPS timestamps and time offsets were applied to account for the inlet residence time differences as detailed in Supporting Information Section S1. Instrument specifications and analytical uncertainty estimation are detailed in Supporting Information (Table S1 and Section S2).

We assessed pollutant-specific spatial resolutions of on-road measurements to choose an appropriate fixed road segment spatial scale such that on-road plumes measured at a particular location are assigned to the most likely corresponding road segment. Spatial resolutions were estimated by multiplying the sampling system (inlet and instrument) response time (e-folding time for a step change in concentration) by percentiles of the mobile platform vehicle speed distribution. Pollutant-specific spatial resolutions ranged from 15 to 72 m on surface streets and 62−170 m on highways with higher vehicle speeds (Table S1). To compare across pollutants with the instruments used on our mobile platform, we chose a consistent 50 m road segment spatial scale for surface streets and 90 m road segment spatial scales on highways. The majority of on-road surface street measurements (e.g., >80% for BC and NO) have a spatial resolution finer than 50 m. There is a small probability that for the slowest instrument responses and highest vehicle speeds on surface streets (i.e., longest spatial resolution), a portion of a detected plume will be observed in an adjacent segment. However, for the worst case scenario of a persistent point source plume, wind direction variations will shift the location of an artificially longer plume decay (due to instrument response time) within a specific road segment. In addition, on-road vehicle plumes are intercepted at a variety of locations, and are observed only on a subset of drive periods. Thus, these artifacts are rare and do not significantly impact summary statistics.

The minimum method detection limit (denoted as MDL) was determined for each pollutant via methods detailed in Supporting Information Section S2. Zero concentration time series were collected periodically throughout the study. We aggregated these periods into one time series for each pollutant, a total of ~4.5 h for particles and ~25 min for gases. The mean + 3σ of the aggregated time series is our best estimate of the MDL, and 3σ is the largest term contributing to MDL values.

2.2. Spatiotemporal Coverage. We repeatedly sampled all public roads within 35 representative census tracts in Harris County, Texas containing ~1300 total road kilometers across ~85 km^2 area with a range of emission source influences, adverse health outcome rates,[3] and population characteristics (further details in Supporting Information Section S3). To select census tracts for sampling, we created a unit-less rank score based upon average daily traffic counts, the spatial density of metal recycling, concrete batch plants, and petrochemical facility source locations, and the presence of freeways. These sources were chosen because of their importance for regional stakeholders and recent public scrutiny in Houston, and our ability to probe source plumes from the ground-level, on-road mobile sampling locations. Some selected census tracts contain a regulatory air quality monitoring site.

We analyzed data collected on arterial, local, and service drive (frontage) roads, as well as highways and ramps (only ~5% of sampled road segments). The near-highway road segments within our sampling domain capture concentration patterns immediately adjacent to or below highways associated with on-highway vehicle emissions. VOCs are the primary species of concern that have been observed downwind of

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petrochemical facilities.[30] Because we focused our measurements on mobile combustion source tracers at ground level (rather than atmospheric columns) with no VOC measurements, our mobile measurements were generally not able to detect lofted petrochemical facility source plumes, nor distinguish elevated stack emissions of NO_x or PM from ground-level emission plumes near roadways.

Mobile measurements covered ~51,900 km total driving distance from July 2017 through March 2018. We attempted to drive every public road in a census tract at least once during every drive period. The start times of drive periods in each census tract were distributed across time of day and day of week to minimize systematic temporal biases, in addition to varying the starting location within census tracts. Mobile measurements mainly covered weekday, daytime hours (7:00 to 16:00 local standard time; 5−95 percentile range of timestamps), with limited early morning, late evening, and weekend drive periods (Figure S4). We limited our analyses to the 24 census tracts with 15−44 repeated drive periods to ensure reasonably small uncertainties. The number of drive periods varied across road segments within an individual census tract because practical considerations required additional passes of some roads to obtain full road sampling coverage (Figure S5).

2.3. Geolocation. Measurements were assigned to fixed road segments via methods detailed in Supporting Information Section S1. In brief, GPS coordinates were map-matched to a digitized road network of 10 m spaced points along the roadways. This minimizes road assignment misclassification within the ~8 m GPS location accuracy and preserves the mobile sampling trajectory along the road network length. We defined the boundaries of road segments with a preference to separate segments by road types (Figure S1). Data collected <30 m from a dead-end and >30 m from a 10 m road point were removed because of potential self-sampling of the mobile vehicle's exhaust while stationary or driving in reverse or a GPS error assigning measurements to nonroad locations.

2.4. Statistical Approaches. We created distributions of concentrations measured at each individual road segment during each sampling (drive) period conducted throughout the study (Figure S6). Because census tracts were sometimes sampled during the morning and afternoon on the same day separated by > 4 h, we defined these 4 h periods as distinct drive periods. First, we aggregated multiple 1 s mobile measurements collected during each unique drive-pass of a fixed road segment into a drive-pass mean, consistent with Messier et al.[24] Drive-pass means equal weight passes of variable durations and sample sizes, reducing artifacts associated with oversampling of vehicle exhaust plumes during passes in slow-moving traffic. Because NO and NO_2 measurements were acquired at a 5 s time resolution, we repeated data points across the 1 s timestamps, consistent with Shah et al.,[17] to represent this integrated measurement across road segments. Second, we calculated the median of drive-pass means for each fixed 4 h drive period, termed "drive period median". This second level of aggregation minimizes overweighting repeated drive-passes of a road segment within short (minutes) time intervals. Most drive periods have one drive pass of a single road segment, meaning that drive pass mean = drive period median. Approximately 12% of drive periods have >3 drive-passes per road segment. Third, we built distributions of drive period median concentrations at each road segment and extracted summary statistics of these distributions to characterize near-source plume behaviors. Persistently elevated concentrations are represented by elevation in median and 90th percentile, intermittent elevation or high variability by interquartile range (IQR), and high extreme concentrations by exclusive elevation in 90th percentile or skewness.

Next, we compared summary statistics of each road segment's distribution with the median of its corresponding summary statistic across all road segments within the entire sampling domain of multiple census tracts (hereafter referred to as "domain-wide"). We mainly focus on domain-wide comparisons because of their relevance for city-scale emission inventory evaluation and mitigation strategies for the highest impact urban sources. Furthermore, domain-wide comparisons allow for detecting elevated concentrations even when the majority of road segments in a given census tract exhibit elevated concentrations. We also performed finer scale analyses to detect locally elevated concentrations relative to each census tract domain and nearest-neighbors, the latter via normalization by the inverse distance-weighted median of the road segment summary statistic. Our individual road segment comparison is distinct from traditional hotspot analyses. Those approaches identify statistically significant hotspots as locations that are part of a cluster with other locations of elevated concentration.[31]

2.4.1. Elevation above Uncertainty Levels. We defined an elevated summary statistic at an individual road segment as one with a lower confidence bound that does not overlap with the upper confidence bound of a reference domain median statistic, nor the MDL. Although nonoverlapping confidence bounds is a descriptive measure not directly based on a statistical probability, our approach provides a robust test beyond rejection of the null hypothesis (p clearly less than 0.05) that the drive period median distributions at a given road segment are the same as that of the domain median statistics. We found a weak inverse linear correlation ($r = -0.3$, $p < 0.01$) between median NO_2 or PN concentration confidence intervals and their drive period sample size, which is likely associated with a high number of drive periods on roads used to enter or exit other census tracts (Figure S5). However, the discrete locations we detected on local roads have similar sample sizes as other roads within their census tract.

Sampling precision (95% confidence interval) was calculated separately for each summary statistic based upon distributions at each road segment as 1.96 times the standard error via bootstrap resampling with 2000 draws. If the instrument error (bias and precision summed in quadrature) or median temporal sampling uncertainty (Section 2.4.2) were larger than the sampling precision, the higher of these values defined the 95% confidence interval (Table S2). This assumes that these uncertainties have some dependence on or are somewhat correlated with each other. The same method was applied for the domain median summary statistics. For NO_2 and PN, a small number of drive period medians were below the MDL and we included below MDL values without substitution for bootstrap resampling. For NO_2 and PN standard error estimates, we substituted below MDL values with MDL/2 to compute the domain median statistics. Because the majority of BC and NO data are below the MDL, we estimated confidence intervals for the median and 90th percentiles using the reverse Kaplan−Meier estimator of the cumulative distribution function,[32] a logarithmic variance approach that appropriately bounds the confidence intervals (Supporting Information Section S4).

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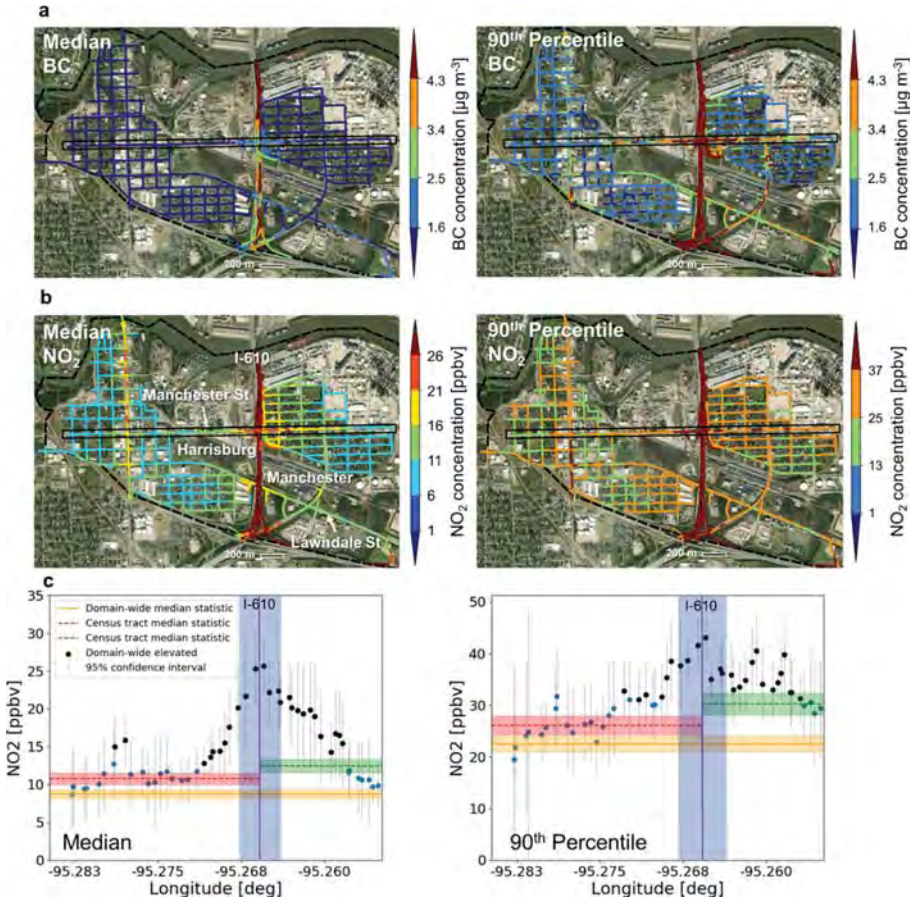

Figure 1. (a) BC and (b) NO_2 median and 90th percentile concentrations at 50 m (surface street) and 90 m (highway) road segments in the Harrisburg and Manchester census tracts, separated by the I-610 interstate highway. The concentration increments represented by the shades of color in the color-bar are based upon the BC instrument precision at 3500 ng m^{-3} and for NO_2 the 95% confidence interval of the highest (90th percentile) summary statistic concentration across the domain (Table S2 and Section 2.4.3). Black rectangles indicate the Manchester St transect, and dashed black lines denote census tract boundaries. The Esri World Imagery Map is shown in the background (sources: Esri, DigitalGlobe, Earthstar Geographics, CNES/Airbus DS, GeoEye, USDA FSA, USGS, Aerogrid, IGN, IGP, and the GIS User Community). (c) Road segment NO_2 median and 90th percentile summary statistics across the Manchester St transect. Locations with elevated concentrations relative to domain-wide median statistics are shown as black points, with the remaining road segments shown as blue points. The I-610 interstate highway location is denoted by the purple line and the blue shaded region is ±200 m horizontal distance from I-610.

2.4.2. Temporal Uncertainties. We considered additional uncertainties associated with aggregating temporally sparse mobile measurements at an individual road segment. First, inter-census tract temporal sampling error can occur when sampling census tracts on different subsets of days with different background concentrations. This uncertainty is important when comparing statistics associated with individual road segments with those across the entire sampling domain. We estimated the potential magnitude of the error associated with inter-drive period atmospheric variability is <±13% for NO and NO_2 based on regulatory monitoring data from the study region (details on other pollutants in Supporting Information Section S5). Second, temporal uncertainty occurs at the intra-census tract level associated with 1−2 h concentration changes associated with boundary layer mixing conditions during an individual drive period (Supporting Information Section S5). We found this uncertainty is relatively small (<9% for pollutants reported here) and can thus be ignored in our analysis (Table S2). Because of the relatively small magnitude of these uncertainties and limited or nonexistent stationary monitoring for some pollutants in some census tracts, we did not perform background adjustments as used in other geographies.[1,23]

2.4.3. Summary of Uncertainties. Overall, the uncertainty of road segment-specific summary statistics was dominated by sampling precision (Table S2, Figure S7). Instrument or temporal sampling uncertainties exceeded sampling precision uncertainty for ≤5 and ≤1% of locations for PN and NO_2, respectively (Figure S7), indicating that mobile data exhibited a larger variability than the stationary data on which the temporal sampling error estimates were based. NO_2 and PN 90th percentiles exhibited similar confidence intervals (±35 to ±39%) as those of the median, and were much smaller than those of the skewness (±64%) (Figure S7). Thus, the 90th percentile is valuable to characterize the upper portion of the distribution, especially for pollutants with a large fraction of measurements below the MDL. Domain median statistic confidence intervals were much smaller than those for individual road segments and were dominated by instrument or temporal sampling errors.

2.5. Meteorological and Temporal Source Variability Analysis. In order to evaluate the influences of temporally

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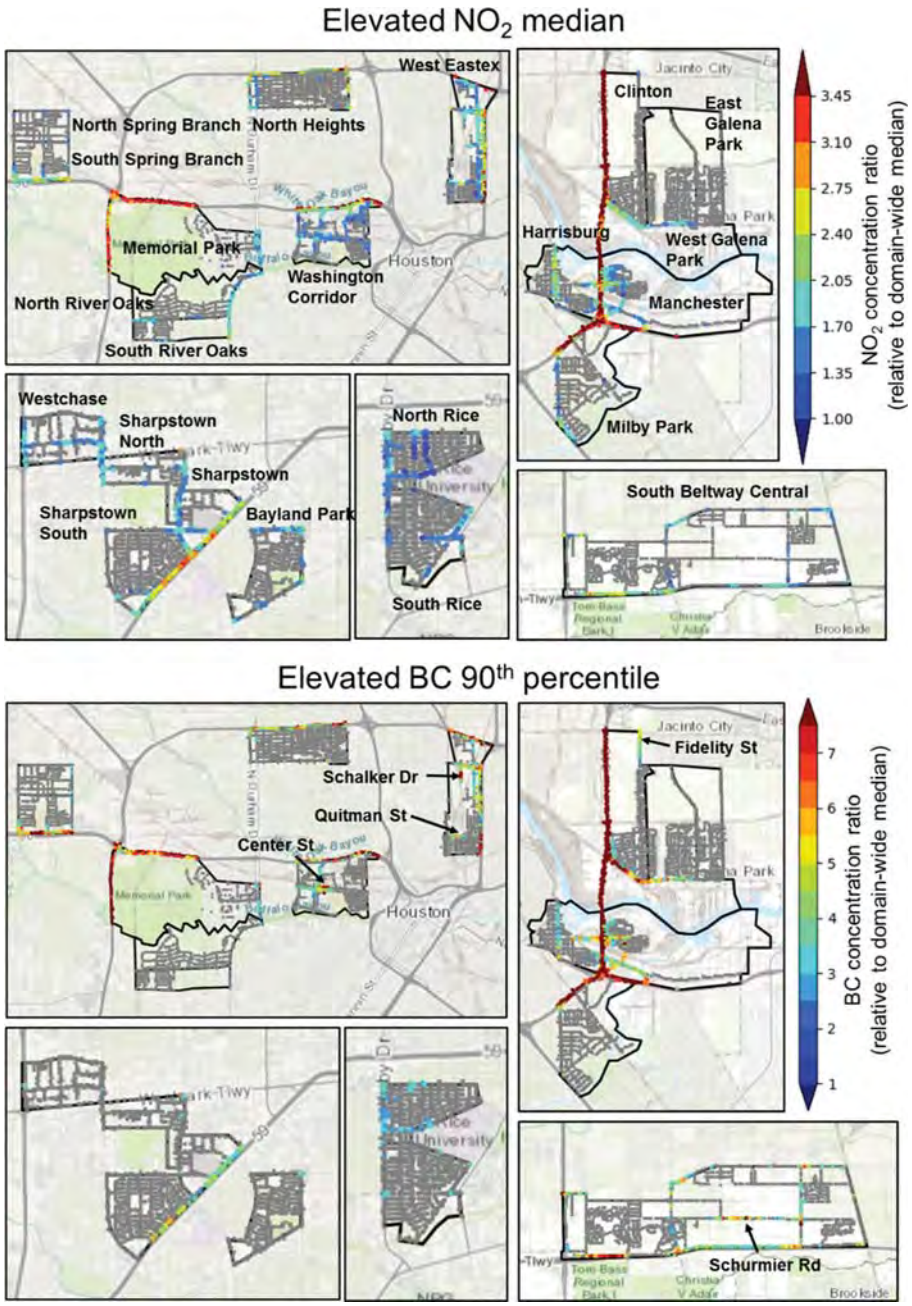

Figure 2. (a) Elevated median NO_2 and (b) elevated 90th percentile BC concentrations normalized by their domain-wide median statistics (9 ppbv NO_2; 1600 ng m^{-3} BC). Census tract names (a) and elevated concentration locations in Table 1 (b) are shown for reference. Grey locations indicate road segments that were measured on more than 15 drive periods, but with a summary statistic confidence interval that overlaps with that of the domain-wide median statistic. The Esri World Topographic Map is shown in the background (sources: Esri, HERE, Garmin, Intermap, INCREMENT P, GEBCO, USGS, FAO, NPS, NRCAN, GeoBase, IGN, Kadaster NL, Ordnance Survey, Esri Japan, METI, Esri China (Hong Kong), OpenStreetMap contributors, GIS User Community).

variable emission sources and meteorological conditions on elevated concentration locations, we examined the patterns of road segment concentrations across the ensemble of individual drive periods and their wind speed and direction characteristics. Because plumes from point sources adjacent to roads are diluted and mixed horizontally and vertically in the turbulent daytime boundary layer before reaching a sampled roadway, it is important to consider the effects of wind conditions and strong signals from local vehicle emission plumes on our platform's ability to detect point source plumes. The closest meteorological station measuring wind speed and direction every 5 min during our study period[33] was identified based on its distance from the census tract centroid. The mean wind vector was computed during each 4 h drive period to represent that period's prevailing wind conditions. This measure was used to evaluate upwind and downwind sampling periods and does not necessarily represent the exact wind conditions during each drive pass because wind conditions were not measured on the mobile platform.

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Table 1. Median Concentration Ratios (Unit-Less, Rounded to Nearest Integer) for Each Pollutant Relative to Respective Domain-Wide Median Statistics at Select Local Road Sections with Known Commercial/Industrial Facilities[a]

census tract(s)	road(s)	potential nearby source influence(s)	BC[c]	NO	NO_2	PN	number of pollutants with elevated median (90th percentile)
West Eastex	Quitman St	metal recycling facility	3	7	3	2	3 (4)
West Eastex	Schalker Dr	metal recycling facility	8	5	2		2 (4)
Washington Corridor	Center St	metal recycling facility	5	3	2		2 (4)
South Beltway Central	Schurmier Rd	2 metal recycling & 3 concrete batch plant facilities	2	8			1 (3)
South Beltway Central	S. Wayside Dr.	concrete batch plant facility	4	7	2	2	3 (2)
Clinton	Fidelity St	metal recycling facility	4	8			2 (2)
all	all within 200 m of highways[b]	near-highway (≤200 m)	3−5	4 to >9	2	2−3	3−4 (3−4)

[a]Blank rows indicate ratios are ≤1.5× or elevation criteria are not met. [b]Within all sampled census tracts. [c]90th percentile concentration ratio.

3. RESULTS AND DISCUSSION

3.1. Identification of Elevated Concentration Locations. We illustrate the process of identifying road segments with elevated pollutant concentrations by focusing on two census tracts in eastern Houston (Manchester and Harrisburg) with populations of ∼1500 and ∼1600, respectively.[34] These census tracts contain a major interstate highway, arterial roads with commercial diesel-fueled vehicle traffic, recycling facilities, refining and petrochemical facilities, and maritime operations along the Houston Ship Channel.[25,27] All pollutants exhibit elevated concentration patterns along Manchester St relative to domain-wide or census tract median statistics (Figures 1 and S8). For BC in this region, we find spatial structure across road segments in the 90th percentile summary statistic (upper tail of drive period median distributions) that is not evident in the median concentrations, which are often below the MDL (Figure 1a).

Our approach to identify elevated concentration locations in the Harrisburg and Manchester census tracts yields consistent results with previously established near-highway concentration decay functions.[1,35] This case study focuses on two roads, Manchester and Lawndale Streets, which are below and perpendicular to the I-610 highway (Figure 1). Near-highway concentration maxima are associated with a combination of on-highway emissions transported to surface streets and the short tunnel's concentrating effect. On Manchester St to the west of I-610, we find similar NO, NO_2, and PN concentration decay (from maximum concentration to 50%) at ∼200 m from the I-610 highway, consistent with Karner et al.[35] Although some highway measurements may be assigned to the same GPS location as below the highway on Manchester St, this does not impact our estimates at adjacent road segments >50 m from the highway. NO, NO_2, and PN concentration decays are asymmetric, with elevated concentrations extending >400 m east of I-610 (Figures 1b,c, and S8a) and additional spatial structure, especially in BC 90th percentile concentration (Figures 1a and S8a). The Manchester St behavior is fundamentally different from the sharp, symmetric decay profiles at Lawndale St and near-highway surface streets north of Manchester St (Figure 1), and the asymmetry is not linked with wind patterns because both westerly and easterly winds were observed at equal frequencies at the nearby Clinton Drive monitoring site.[33] This suggests that in addition to the highway signal, local NO_x and particulate sources influence Manchester St east of I-610.

Across the entire domain sampled, we find elevated concentrations on highways, on major arterial roadways, on roads below and adjacent to highways, and along discrete sections of local roads (Figures 2 and S9). Median pollutant concentrations are often multiple factors above their corresponding median statistics across all roads sampled. Elevated median concentrations on surface streets range from 2× to 3× the domain-wide median for NO_2, 2× to >9× for NO, and elevated BC 90th percentile concentrations are 2× to > 7× the domain-wide median 90th percentile (Figures 2 and S9a). In contrast, PN exhibits surface street elevated median concentrations of ≤2× the domain-wide median (Figure S9b) and large IQR comparable to the median concentration (Figure S7), consistent with large spatiotemporal variability in ultrafine particle counts reported in previous studies.[4,23] Elevated NO_2 concentrations occur at ∼19% of road segments based on the median and ∼13% based on the 90th percentile, but only ∼6% based on the IQR owing to wide IQR confidence intervals (Table S3). The inclusion of highway and service drive road segments reduces the percent of elevated concentration road segments detected on surface streets by ≪30% (Table S4). The spatial patterns of elevated median PN concentrations differ from those of the other pollutants, with elevated concentrations on residential streets in West and East Galena Park at comparable magnitude to those on the nearby arterial road, Clinton Drive (Figure S9b). Finally, the summary statistics describe combinations of persistent, intermittent and/or extreme concentration behaviors (Figure S10), with further descriptions provided in Supporting Information Section S7.

3.1.1. Domain Sensitivity Analyses. We evaluate the sensitivity of identifying elevated concentration road segments to the choice of the reference domain. For this sensitivity analysis, we examine elevated median NO_2 on surface streets in the Harrisburg, Manchester, and West Eastex census tracts, containing ∼12% of all road segments analyzed (Figure S11). In these census tracts, we find that 42% of road segment median concentrations are elevated relative to their domain-wide median, while only 10 and 3% are elevated relative to their census tract or nearest-neighbor medians, respectively. The nearest-neighbors approach provides limited value for our analysis because it is the most stringent comparison that mainly detects a subset of locations identified by the domain-wide comparison, particularly in isolated elevated concentration locations (Figure S11).

Separately, we evaluate the sensitivity of our results to the pool of road types considered. We find that elevated

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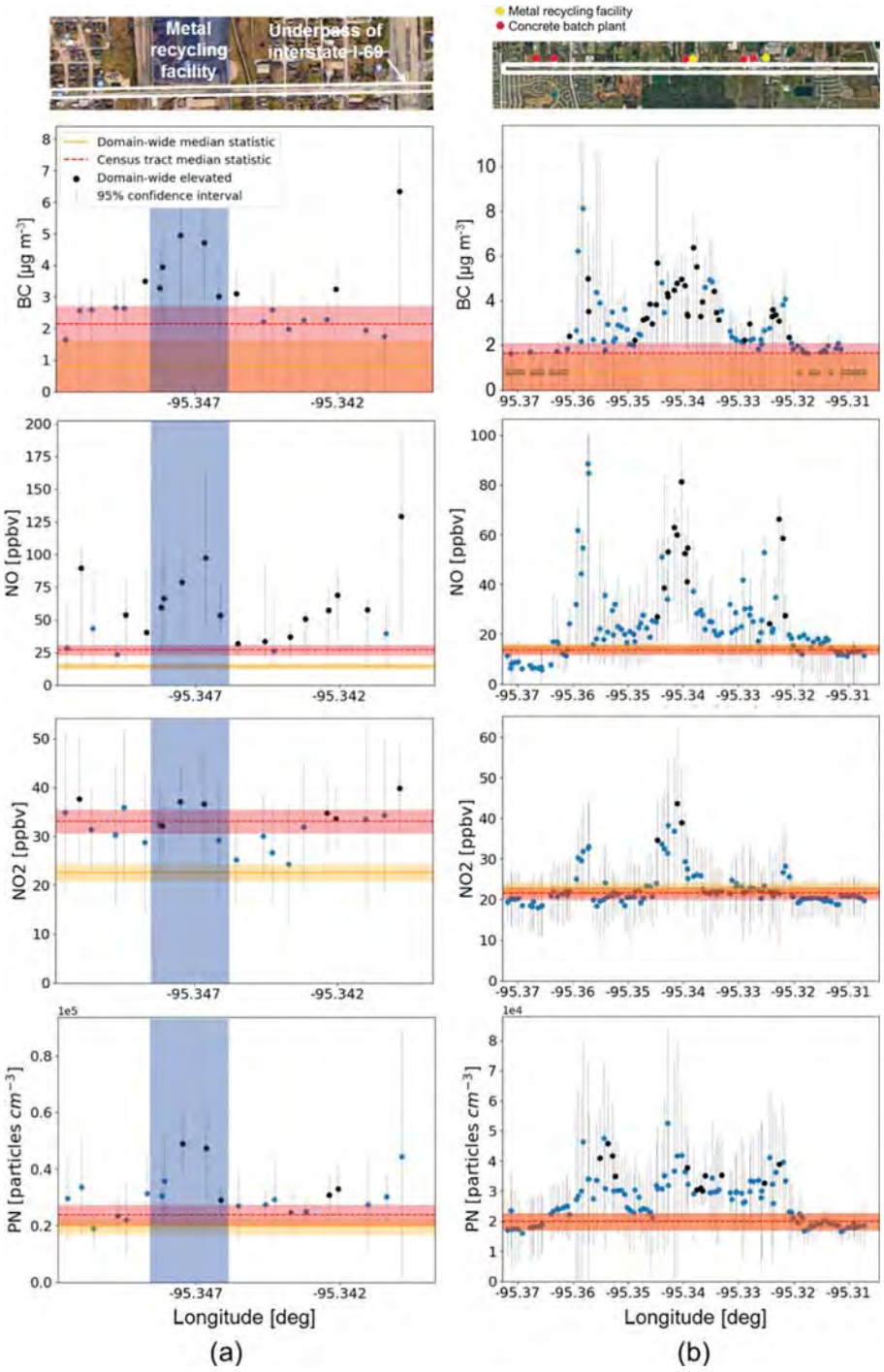

Figure 3. 90th percentile concentrations at (a) Quitman St transect (West Eastex census tract, blue shading represents metal recycling facility boundaries), and (b) Schurmier Rd transect (South Beltway Central census tract). The locations of potential sources influencing the observed patterns are denoted in the Google Earth imagery at top (Imagery 2019 Google, Map data 2019 United States).

concentration detection results are robust to the inclusion of high concentration highway, ramp and service drive road segments, which make up a small (~9%) portion of all sampled road segments. Specifically, we compare median statistic values based on the full population of sampled road segments with those obtained while excluding highway, ramp, and service drive road segments. The former scenario results in ≤8% higher domain-wide median values than the latter scenario, with 12% higher domain-wide median 90th percentile for NO_2 (Table S5).

3.2. Sources of Elevated Concentrations. Our analysis reveals elevated pollutant concentrations along discrete, local road sections (Figures 2 and S9, Table 1). Specifically, we find elevated median concentration in 2−3 pollutants at close proximity to 38% of (6 out of 16) metal recycling facilities and 24% of (4 out of 17) concrete batch plants within our sampled census tracts (Table 1). Elevated concentration magnitudes at

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these facility locations span those observed across the domain (Figures 2 and S9) and are of similar magnitude as those ≤200 m of highways within the same census tracts (Table 1). Near these facilities, BC concentration 90th percentiles reach 2−8× the domain-wide median statistic (Figure 2b, Table 1). These elevated concentration features along local roads are important contributors to overall pollutant spatial patterns. An additional 19% of metal recycling plants and 12% of concrete batch plants are adjacent to road segments exhibiting elevated concentrations on a stretch of major roadway with elevated concentrations along many or all of its road segments. However, apportioning the local contribution of an individual facility relative to the vehicle source along the entire roadway's length is beyond the scope of this study.

Next, we evaluate individual drive periods in elevated concentration locations to examine whether sources and/or meteorology influenced the distributions. We choose two regions with elevated characteristics for illustrative case studies (Figure 3). We find elevated features along a discrete ∼300 m section of Quitman St (West Eastex census tract) along the property boundaries and immediately south of a metal recycling plant (Figures 3a and S12a). The 90th percentiles show the greatest sensitivity for localized elevation. Elevated concentrations at the eastern edge are associated with near or on-highway vehicle emissions, with confidence intervals for segments adjacent to this recycling facility overlapping with those at the near-highway road segment (Figure 3a). The locations of high concentrations between drive passes do not exhibit discernible wind dependence (Figure S13). Locations of higher concentration are relatively narrow (50−100 m) and occur at different locations >50 m apart across individual drive periods (Figure S13). These observations suggest that close proximity mobile sources are the main drivers of these elevated features, rather than a constant point or area source plume shifting with the wind direction. However, more work is needed to rule out effects of on-site operation emissions because only ∼30% of drive periods occurred downwind of the facility (typically mornings with light north winds; Figure S13).

We also observed broad, spatially heterogeneous elevated features for BC and NO along the Schurmier Rd transect (South Beltway Central census tract), which is immediately south of a variety of emission sources including five concrete batch plants, two metal recycling facilities, trucking facilities, and intersections (Figure 3b). The elevated 90th percentile concentrations for BC, NO, and NO_2 in the middle of the transect extend up to ∼1 km, with relatively large confidence intervals owing to large variability across drive periods (Figure S12). However, while PN shows high variability and elevated extreme values, PN medians are not elevated (Figures 3b and S12b). Elevated BC and NO_x concentration features are not adjacent to one particular facility or intersection and also do not exhibit wind dependence (Figure S14), suggesting they are associated with an on-road mobile vehicle source (e.g., traffic backed up from a nearby intersection) or one or more nearby facility's emission source footprints extending beyond their immediate boundaries.

Both case studies suggest that the road transect spatial concentration patterns are associated with mobile on-road or on-site diesel-fueled vehicle sources with spatial scales corresponding to intercepting an on-road diesel vehicle plume for ∼10 s while driving at 5−10 m s^{-1}. Although older gasoline-fueled passenger vehicles with faulty emission control technologies may also contribute to the elevated NO and NO_2 concentrations,[36,37] the coincident elevated BC concentrations are more consistent with a diesel engine source.[38] Although BC and NO_x may be emitted from on-site diesel equipment for recycling processes and particulate matter emissions from such facilities have been reported previously,[39] we did not sample during high speed, north wind periods optimal for capturing on-site emission plumes. In addition, the longer distance from these sources to on-road sampling locations means they are subject to vertical dispersion and dilution to levels below that of much closer on-road source plumes.

4. IMPLICATIONS

Our mobile measurements characterized persistently elevated and extreme concentration behaviors on arterial and local roads. We found heterogeneous, elevated concentration spatial patterns that are ubiquitous across a larger spatial domain than previous studies, ∼85 km^2 compared with ∼30 km^2 in Oakland, California.[1,24] Our approach explicitly quantifies the sampling attributes that determine the sensitivity of a mobile monitoring method to detect elevated concentrations, which can inform routine monitoring and future study designs in other urban areas. We found sampling and instrument uncertainties across 24 noncontiguous census tracts are reasonably small to identify elevated concentrations at least 40% above that of the domain-wide median, even when including near and on-highway road segments. This mobile sampling approach offers unique capabilities to both identify and rank locations by their concentrations at specific road sections that facilitate follow-up surveys and prioritize locations for inspection. Our road transect case studies suggest a diesel-fueled mobile emission source influence with discrete areas (≤100 m) of elevated primary emission tracer (BC, NO_x) concentrations near metal recycling and concrete batch plant facilities with potential cumulative effects of multiple closely clustered facilities. These features are an exemplar of urban emission sources that can be identified with mobile monitoring approaches. Metal recycling facilities are ubiquitous in other urban areas and have also been identified as a source of carcinogenic metals in Houston,[39] and a hotspot for diesel vehicle emissions in Oakland, California.[1] Overall, these results have important implications for understanding and tracking advection of near-facility emission plumes into residential neighborhoods.

Our analyses lay the foundation for future studies focusing on source quantification, source attribution, and health impacts. First, future analyses of individual drive period observations focused on identifying concentration patterns upwind and downwind of known point and area sources and distinguishing their source footprints from on-road vehicle plumes under specific atmospheric stability regimes would provide useful insights. Future mobile monitoring opportunities that incorporate more extensive night time and weekend observations would also be valuable to distinguish the relative influence of emissions from light-duty passenger vehicles versus heavy-duty commercial vehicles operating only during weekdays,[40] as well as night time emission sources, on elevated concentration patterns. Second, emission ratios with respect to CO_2 in elevated concentration locations can be used to quantify on-road vehicle emissions. These results can then be compared with fine-scale mobile source emission inventories (e.g., McDonald et al.[41]) to improve their spatial emission allocation estimates. Third, follow-up mobile monitoring

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should include additional urban source tracers (e.g., carbon monoxide, VOCs) to further fingerprint emission sources. Fourth, mobile monitoring observations can be compared with and complement existing stationary monitoring and satellite remote sensing observations, and provide additional measurements where existing monitors do not exist. For example, mobile monitoring observations could serve as an independent test for upcoming geostationary satellite retrievals at kilometer and hourly scales. Finally, an upcoming Health Impact Assessment study utilizing these mobile measurements will complement previous health studies[3] and further evaluate disparities in pollution-related health impacts. Our study demonstrates the value of using city-scale mobile monitoring to inform policy development related to urban air quality and emission source management strategies, and for communicating patterns of near-road exposure to policy stakeholders and the general public.

■ ASSOCIATED CONTENT

Supporting Information

The Supporting Information is available free of charge at https://pubs.acs.org/doi/10.1021/acs.est.9b05523.

Text, figures, and tables with detailed information on geolocation and uncertainty methodologies, spatiotemporal data set coverage, and supplementary statistical and sensitivity analysis results (PDF)

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Notes

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■ REFERENCES

(1) Apte, J. S.; Messier, K. P.; Gani, S.; Brauer, M.; Kirchstetter, T. W.; Lunden, M. M.; Marshall, J. D.; Portier, C. J.; Vermeulen, R. C. H.; Hamburg, S. P. High-Resolution Air Pollution Mapping with Google Street View Cars: Exploiting Big Data. *Environ. Sci. Technol.* **2017**, *51*, 6999−7008.

(2) Caubel, J. J.; Cados, T. E.; Preble, C. V.; Kirchstetter, T. W. A Distributed Network of 100 Black Carbon Sensors for 100 Days of Air Quality Monitoring in West Oakland, California. *Environ. Sci. Technol.* **2019**, *53*, 7564−7573.

(3) Raun, L. H.; Ensor, K. B.; Persse, D. Using community level strategies to reduce asthma attacks triggered by outdoor air pollution: a case crossover analysis. *Environ. Health* **2014**, *13*, 58.

(4) Simon, M. C.; Hudda, N.; Naumova, E. N.; Levy, J. I.; Brugge, D.; Durant, J. L. Comparisons of traffic-related ultrafine particle number concentrations measured in two urban areas by central, residential, and mobile monitoring. *Atmos. Environ.* **2017**, *169*, 113−127.

(5) Alexeeff, S. E.; Roy, A.; Shan, J.; Liu, X.; Messier, K.; Apte, J. S.; Portier, C.; Sidney, S.; Van Den Eeden, S. K. High-resolution mapping of traffic related air pollution with Google street view cars and incidence of cardiovascular events within neighborhoods in Oakland, CA. *Environ. Health* **2018**, *17*, 38.

(6) Hoek, G.; Krishnan, R. M.; Beelen, R.; Peters, A.; Ostro, B.; Brunekreef, B.; Kaufman, J. D. Long-term air pollution exposure and cardio- respiratory mortality: a review. *Environ. Health* **2013**, *12*, 43.

(7) Janssen, N. A. H.; Gerlofs-Nijland, M. E.; Lanki, T.; Salonen, R. O.; Cassee, F.; Hoek, G. *Health Effects of Black Carbon*; World Health Organization: Copenhagen, Denmark, 2012.

(8) Patton, A. P.; Zamore, W.; Naumova, E. N.; Levy, J. I.; Brugge, D.; Durant, J. L. Transferability and Generalizability of Regression Models of Ultrafine Particles in Urban Neighborhoods in the Boston Area. *Environ. Sci. Technol.* **2015**, *49*, 6051−6060.

(9) Simon, M. C.; Patton, A. P.; Naumova, E. N.; Levy, J. I.; Kumar, P.; Brugge, D.; Durant, J. L. Combining Measurements from Mobile Monitoring and a Reference Site To Develop Models of Ambient Ultrafine Particle Number Concentration at Residences. *Environ. Sci. Technol.* **2018**, *52*, 6985−6995.

(10) Mazzuca, G. M.; Ren, X.; Loughner, C. P.; Estes, M.; Crawford, J. H.; Pickering, K. E.; Weinheimer, A. J.; Dickerson, R. R. Ozone production and its sensitivity to NOx and VOCs: results from the DISCOVER-AQ field experiment, Houston 2013. *Atmos. Chem. Phys.* **2016**, *16*, 14463−14474.
(11) Al-Naiema, I. M.; Hettiyadura, A. P. S.; Wallace, H. W.; Sanchez, N. P.; Madler, C. J.; Cevik, B. K.; Bui, A. A. T.; Kettler, J.; Griffin, R. J.; Stone, E. A. Source apportionment of fine particulate matter in Houston, Texas: insights to secondary organic aerosols. *Atmos. Chem. Phys.* **2018**, *18*, 15601−15622.
(12) Zhang, X.; Craft, E.; Zhang, K. Characterizing spatial variability of air pollution from vehicle traffic around the Houston Ship Channel area. *Atmos. Environ.* **2017**, *161*, 167−175.
(13) Fasoli, B.; Lin, J. C.; Bowling, D. R.; Mitchell, L.; Mendoza, D. Simulating atmospheric tracer concentrations for spatially distributed receptors: updates to the Stochastic Time-Inverted Lagrangian Transport model's R interface (STILT-R version 2). *Geosci. Model Dev.* **2018**, *11*, 2813−2824.
(14) Griffin, D.; Zhao, X.; McLinden, C. A.; Boersma, F.; Bourassa, A.; Dammers, E.; et al. High-resolution mapping of nitrogen dioxide with TROPOMI: First results and validation over the Canadian oil sands. *Geophys. Res. Lett.* **2019**, *46*, 1049−1060.
(15) Kim, J.; Shusterman, A. A.; Lieschke, K. J.; Newman, C.; Cohen, R. C. The BErkeley Atmospheric CO_2 Observation Network: field calibration and evaluation of low-cost air quality sensors. *Atmos. Meas. Tech.* **2018**, *11*, 1937−1946.
(16) Herndon, S. C.; Jayne, J. T.; Zahniser, M. S.; Worsnop, D. R.; Knighton, B.; Alwine, E.; Lamb, B. K.; Zavala, M.; Nelson, D. D.; McManus, J. B.; Shorter, J. H.; Canagaratna, M. R.; Onasch, T. B.; Kolb, C. E. Characterization of urban pollutant emission fluxes and ambient concentration distributions using a mobile laboratory with rapid response instrumentation. *Faraday Discuss.* **2005**, *130*, 327−339.
(17) Shah, R. U.; Robinson, E. S.; Gu, P.; Robinson, A. L.; Apte, J. S.; Presto, A. A. High-spatial-resolution mapping and source apportionment of aerosol composition in Oakland, California, using mobile aerosol mass spectrometry. *Atmos. Chem. Phys.* **2018**, *18*, 16325−16344.
(18) Mitchell, L. E.; Crosman, E. T.; Jacques, A. A.; Fasoli, B.; Leclair-Marzolf, L.; Horel, J.; Bowling, D. R.; Ehleringer, J. R.; Lin, J. C. Monitoring of greenhouse gases and pollutants across an urban area using a light-rail public transit platform. *Atmos. Environ.* **2018**, *187*, 9−23.
(19) Ranasinghe, D. R.; Choi, W.; Winer, A. M.; Paulson, S. E. Developing High Spatial Resolution Concentration Maps Using Mobile Air Quality Measurements. *Aerosol Air Qual. Res.* **2016**, *16*, 1841−1853.
(20) Choi, W.; Ranasinghe, D.; Bunavage, K.; DeShazo, J. R.; Wu, L.; Seguel, R.; Winer, A. M.; Paulson, S. E. The effects of the built environment, traffic patterns, and micrometeorology on street level ultrafine particle concentrations at a block scale: Results from multiple urban sites. *Sci. Total Environ.* **2016**, *553*, 474−485.
(21) Sun, K.; Tao, L.; Miller, D. J.; Pan, D.; Golston, L. M.; Zondlo, M. A.; Griffin, R. J.; Wallace, H. W.; Leong, Y. J.; Yang, M. M.; Zhang, Y.; Mauzerall, D. L.; Zhu, T. Vehicle Emissions as an Important Urban Ammonia Source in the United States and China. *Environ. Sci. Technol.* **2017**, *51*, 2472−2481.
(22) Li, Z.; Fung, J. C. H.; Lau, A. K. H. High spatiotemporal characterization of on-road PM2.5 concentrations in high-density urban areas using mobile monitoring. *Build. Environ.* **2018**, *143*, 196−205.
(23) Li, H. Z.; Gu, P.; Ye, Q.; Zimmerman, N.; Robinson, E. S.; Subramanian, R.; Apte, J. S.; Robinson, A. L.; Presto, A. A. Spatially dense air pollutant sampling: Implications of spatial variability on the representativeness of stationary air pollutant monitors. *Atmos. Environ.* **2019**, *2*, 100012.
(24) Messier, K. P.; Chambliss, S. E.; Gani, S.; Alvarez, R.; Brauer, M.; Choi, J. J.; Hamburg, S. P.; Kerckhoffs, J.; LaFranchi, B.; Lunden, M. M.; Marshall, J. D.; Portier, C. J.; Roy, A.; Szpiro, A. A.; Vermeulen, R. C. H.; Apte, J. S. Mapping Air Pollution with Google Street View Cars: Efficient Approaches with Mobile Monitoring and Land Use Regression. *Environ. Sci. Technol.* **2018**, *52*, 12563−12572.
(25) Wallace, H. W.; Sanchez, N. P.; Flynn, J. H.; Erickson, M. H.; Lefer, B. L.; Griffin, R. J. Source apportionment of particulate matter and trace gases near a major refinery near the Houston Ship Channel. *Atmos. Environ.* **2018**, *173*, 16−29.
(26) Zirogiannis, N.; Hollingsworth, A. J.; Konisky, D. M. Understanding Excess Emissions from Industrial Facilities: Evidence from Texas. *Environ. Sci. Technol.* **2018**, *52*, 2482−2490.
(27) Yacovitch, T. I.; Herndon, S. C.; Roscioli, J. R.; Floerchinger, C.; Knighton, W. B.; Kolb, C. E. Air Pollutant Mapping With a Mobile Laboratory During the BEE-TEX Field Study. *Environ. Health Insights* **2015**, *9*, 7−13.
(28) Linder, S. H.; Marko, D.; Sexton, K. Cumulative Cancer Risk from Air Pollution in Houston: Disparities in Risk Burden and Social Disadvantage. *Environ. Sci. Technol.* **2008**, *42*, 4312−4322.
(29) Houston-Galveston-Brazoria: Current Attainment Status, Compliance of HGB-area counties with the National Ambient Air Quality Standards (NAAQS). https://www.tceq.texas.gov/airquality/sip/hgb/hgb-status (accessed Sep 12, 2019).
(30) Johansson, J. K. E.; Mellqvist, J.; Samuelsson, J.; Offerle, B.; Lefer, B.; Rappenglück, B.; Flynn, J.; Yarwood, G. Emission measurements of alkenes, alkanes, SO2, and NO2 from stationary sources in Southeast Texas over a 5 year period using SOF and mobile DOAS. *J. Geophys. Res.: Atmos.* **2014**, *119*, 1973−1991.
(31) Getis, A.; Ord, J. K. The Analysis of Spatial Association by Use of Distance Statistics. *Geogr. Anal.* **1992**, *24*, 189.
(32) Gillespie, B. W.; Chen, Q.; Reichert, H.; Franzblau, A.; Hedgeman, E.; Lepkowski, J.; Adriaens, P.; Demond, A.; Luksemburg, W.; Garabrant, D. H. Estimating Population Distributions When Some Data Are Below a Limit of Detection by Using a Reverse Kaplan-Meier Estimator. *Epidemiology* **2010**, *21*, S64−S70.
(33) Texas Commission on Environmental Quality (TCEQ). Air Quality Monitoring Site Data. https://www.tceq.texas.gov/airquality/monops/sites (accessed Jan 18, 2019).
(34) United States Census Bureau: Census 2010. https://www.census.gov/programs-surveys/decennial-census/decade.2010.html (accessed April 1, 2017).
(35) Karner, A. A.; Eisinger, D. S.; Niemeier, D. A. Near-Roadway Air Quality: Synthesizing the Findings from Real-World Data. *Environ. Sci. Technol.* **2010**, *44*, 5334−5344.
(36) Bishop, G. A.; Schuchmann, B. G.; Stedman, D. H.; Lawson, D. R. Multispecies remote sensing measurements of vehicle emissions on Sherman Way in Van Nuys, California. *J. Air Waste Manage. Assoc.* **2012**, *62*, 1127−1133.
(37) Bishop, G. A.; Stedman, D. H.; Burgard, D. A.; Atkinson, O. High-mileage light-duty fleet vehicle emissions: Their potentially overlooked importance. *Environ. Sci. Technol.* **2016**, *50*, 5405−5411.
(38) Dallmann, T. R.; Kirchstetter, T. W.; DeMartini, S. J.; Harley, R. A. Quantifying On-Road Emissions from Gasoline-Powered Motor Vehicles: Accounting for the Presence of Medium- and Heavy-Duty Diesel Trucks. *Environ. Sci. Technol.* **2013**, *47*, 13873−13881.
(39) Raun, L.; Pepple, K.; Hoyt, D.; Richner, D.; Blanco, A.; Li, J. Unanticipated potential cancer risk near metal recycling facilities. *Environ. Impact Assess. Rev.* **2013**, *41*, 70−77.
(40) Marr, L. C.; Harley, R. A. Modeling the effect of weekday-weekend differences in motor vehicle emissions on photochemical air pollution in central California. *Environ. Sci. Technol.* **2002**, *36*, 4099−4106.
(41) McDonald, B. C.; Goldstein, A. H.; Harley, R. A. Long-Term Trends in California Mobile Source Emissions and Ambient Concentrations of Black Carbon and Organic Aerosol. *Environ. Sci. Technol.* **2015**, *49*, 5178−5188.

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Local- and regional-scale racial and ethnic disparities in air pollution determined by long-term mobile monitoring

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Disparity in air pollution exposure arises from variation at multiple spatial scales: along urban-to-rural gradients, between individual cities within a metropolitan region, within individual neighborhoods, and between city blocks. Here, we improve on existing capabilities to systematically compare urban variation at several scales, from hyperlocal (<100 m) to regional (>10 km), and to assess consequences for outdoor air pollution experienced by residents of different races and ethnicities, by creating a set of uniquely extensive and high-resolution observations of spatially variable pollutants: NO, NO_2, black carbon (BC), and ultrafine particles (UFP). We conducted full-coverage monitoring of a wide sample of urban and suburban neighborhoods (93 km^2 and 450,000 residents) in four counties of the San Francisco Bay Area using Google Street View cars equipped with the Aclima mobile platform. Comparing scales of variation across the sampled population, greater differences arise from localized pollution gradients for BC and NO (pollutants dominated by primary sources) and from regional gradients for UFP and NO_2 (pollutants dominated by secondary contributions). Median concentrations of UFP, NO, and NO_2 are, for Hispanic and Black populations, 8 to 30% higher than the population average; for White populations, average exposures to these pollutants are 9 to 14% lower than the population average. Systematic racial/ethnic disparities are influenced by regional concentration gradients due to sharp contrasts in demographic composition among cities and urban districts, while within-group extremes arise from local peaks. Our results illustrate how detailed and extensive fine-scale pollution observations can add new insights about differences and disparities in air pollution exposures at the population scale.

air pollution | environmental justice | air quality

Air pollution varies in complex patterns across the urban landscape, arising from the interplay of emissions source locations and atmospheric transport and transformation. Gradients exist at multiple spatial scales, reflecting regional, city-level, and neighborhood-level phenomena, including highly localized peaks near major sources (1–3). The uneven distribution of sources has been shown in the United States to cause systematically higher outdoor concentrations for people of color and communities facing disproportionate socioeconomic and environmental stressors (4–10). Increased air pollution exposure is associated with premature mortality and a multitude of chronic health problems, as well as increased vulnerability to extreme events such as wildfire pollution episodes and COVID-19 (11–13). Measurement and analysis of this disparity in outdoor concentrations are vital for understanding how the causes of air pollution (e.g., city zoning, infrastructure development, emissions sources) affect differential health outcomes. This understanding can aid in designing effective environmental justice measures and tracking the effects of the evolving urban landscape on population-wide and community exposure. Here, we use mobile monitoring (in-motion measurements by vehicle-mounted instruments) to observe highly localized air pollution patterns in a variety of urban settings and consider the implications for the measurement and mitigation of pollution exposure and environmental inequity.

The full complexity of multiscale patterns of air pollution is largely unknown in most urban areas despite great advances in measurement and modeling methods over the past few decades. Regulatory monitoring sites are sparsely distributed and generally do not measure unregulated pollutants of health concern, such as black carbon (BC) and ultrafine particles (UFP). While satellite remote sensing provides nearly global spatial coverage, most conventional products are limited in resolution to 1 to 5 km^2 and do not include all pollutants of interest (14, 15). Mechanistic models predict concentrations over broad domains but are limited by computational constraints and data gaps (16). Recent statistical models provide both high spatial resolution and geographic coverage, but concentration predictions reflect generalized patterns, tend to predict central tendencies better than

Significance

It is known, to researchers and heavily impacted communities, that people of color face a higher average burden of air pollution. It was unknown whether racial/ethnic disparities were caused by spatial heterogeneities at the level of city blocks, neighborhoods, or urban regions. Our approach leverages a unique set of highly local observations, covering every city block of 13 cities and urban districts that are home to 450,000 people. We find that even for pollutants with steep localized gradients, differences in average outdoor concentrations among racial/ethnic groups are driven by regional variability. However, localized peaks indicate opportunities to reduce extremes within groups. The methods and findings of this study can inform strategies to reduce disparities in urban air pollution exposure.

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Competing interest statement: B.L., C.R.U., and M.M.L. are employed by Aclima, Inc., which contributed to the collection of the measurements employed here.

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extremes, and may miss local idiosyncrasies, especially for models with broad (e.g., national) domains (17). Despite the growing sophistication of these technologies, in many cases they are best suited to depict patterns of pollutants that are predominantly secondary, such as ozone and $PM_{2.5}$, which tend to vary more over the regional than neighborhood scale (17). In contrast, fine-scale gradients dominate spatial variability for directly emitted (primary) pollutants like BC and for pollutants with highly localized transformation dynamics like UFP and NO (1, 18–22). Both BC and UFP are suspected to cause distinct health impacts (23–25), but these effects—and the potential racial/ethnic and socioeconomic health disparities—will remain poorly understood until higher-resolution measurements are more widespread (25, 26).

Mobile monitoring and low-cost sensors are increasingly used to detect fine-scale pollution gradients with applications ranging from new risk estimates for cardiovascular disease to the identification of unexpected sources of exposure disparity (24, 27). Low-cost sensors are used to supplement the density of existing regulatory networks, but appropriate sensors are not available for all pollutants and operation is limited by the capacity for consistent calibration and maintenance (28). Mobile monitoring has been used to measure multipollutant gradients in a wide range of urban contexts (29–31). Example applications of mobile monitoring include studies of short-range multipollutant variation attributable to highway traffic (31–35), infrastructure geometry, such as street canyons and near-road barriers (22, 36–38), and a variety of specific local sources (37, 39–41). While mobile monitoring is a flexible method, it is also labor intensive, requiring many repeated visits to collect enough localized measurements to capture the full temporal variation in conditions.

Because of the large resource requirement for long-term mobile monitoring, only a small number of campaigns have measured multiyear, high-resolution patterns over multiple neighborhoods (19, 42). Few have attempted comprehensive coverage of all major and residential streets of several contiguous neighborhoods (21, 41, 43). This study presents the results of 32 mo of mobile monitoring along every street of 13 cities, towns, and urban districts (93 km^2) distributed through four counties of the San Francisco Bay Area, providing over 2,100 h of sampling of four pollutants at ~0.01 km^2 (i.e., ~100 × 100 m^2). We employ two custom-equipped Google Street View cars to repeatedly measure city block air quality, providing estimates of outdoor air pollution for a year-2010 population of ~450,000 individuals and an opportunity to characterize, quantify, and analyze multiscale gradients across the urban landscape. We specifically consider implications for racial and ethnic exposure disparities. We find that dense, urban neighborhoods exhibit peaks that vary by pollutant in both location and magnitude, reflecting complex interactions among diverse emission sources and urban microenvironments. Despite high-magnitude hyperlocal peaks, we find that concentration differences between sampling areas (i.e., among distinct neighborhoods and cities) cause greater average concentrations for people of color. These findings demonstrate the need to consider mitigation policies at multiple urban scales to address environmental inequity.

Results and Discussion

High Variation in Air Pollution Observed from Hyperlocal to Regional Scales. We create empirical maps of four air pollutants in 13 groupings of neighborhoods that constitute urban districts, cities, or towns (total population: ~450,000), ranging in land use, urbanization, traffic density, demographic composition, and historical housing policy (*SI Appendix*, Figs. S1, S2, and S12 and Tables S1–S3). We calculate census block scale concentrations (see *Materials and Methods*) to best approximate conditions at individual residences given the resolution of demographic data (median block population: 36 people). Hereon, we describe outdoor concentration as "exposure" for block residents, terminology consistent with literature linking health effects to census block estimates (24, 44); we acknowledge that individual exposure also depends on multiple other factors (e.g., diurnal activity, indoor infiltration and dynamics, and physiology).

Study area maps (Fig. 1) reveal sharp concentration gradients (~2 to 5×) within groups of contiguous blocks, as well as marked differences in the range of concentrations within different study areas. In discussing spatial variation, we term gradients among neighboring blocks (~100 m) as "hyperlocal," variation within each study area (~1 km) as "local," and variation among study areas (~10 km) as "regional." Among the four pollutants, NO shows the highest-magnitude hyperlocal peaks, with a typical ratio of 10× between a peak and local median (*SI Appendix*, Fig. S4 and Table S5). BC, NO_2, and UFP (peak ratios 3.1×, 2.7×, and 2.6×, respectively) exhibit shallower hyperlocal gradients and more diffuse peaks. Complex hyperlocal patterns reflect the combined influence of traffic and point source emissions (e.g., local industry and restaurants) and dispersion and reaction dynamics within the built environment (e.g., street canyons, green spaces). Local elevation of NO concentrations is strongly linked to highways and major arterials, while the influence of highways on BC, NO_2, and UFP varies by study area, with roadways showing the smallest influence on UFP (*SI Appendix*, Table S6–S9). Differences in hyperlocal patterns among pollutants are reflected in the moderate or low correlations of block-level concentrations in some study areas, especially between BC and UFP (*SI Appendix*, Fig. S5). These differences demonstrate the importance of measuring multiple pollutants. Furthermore, these patterns likely differ from those of other important pollutants like fine particulate matter ($PM_{2.5}$) and air toxics, both in location and degree of local and regional variation.

Exposure variation reflects where residents live relative to observed local and regional gradients, although residents may also experience peaks in unpopulated areas during daily activities. We find that the weighting of population within more homogeneous residential areas mildly diminishes the importance of local variation, but both local and regional gradients contribute to the broad range of exposures across the study domain. Fig. 2 shows the full distribution of exposure levels within each study area. Comparing median exposures between the most- and least-polluted study areas, concentrations varied by a factor of 4, 5, 6, and 28 for BC, NO_2, UFP, and NO, respectively, while within-neighborhood interdecile ranges showed variation up to a factor of 4 for BC, NO_2, UFP, and a factor of 19 for NO. Generally, neighborhoods with higher BC and NO medians also display a wider range of exposures, while NO_2 and UFP ranges remain more consistent across neighborhoods. To partition exposure variability into local and regional components, we decomposed the sum-of-squared deviation from the mean (SSD) of each resident versus the study area mean and of all study areas versus the grand mean (*SI Appendix*, Table S12). We find that local gradients contribute the majority of exposure variation for primary pollutants (NO: 52% and BC: 63% of SSD) but the minority for NO_2 and UFP (37 and 28%, respectively), two pollutants where secondary processes are a key source. A subset of study areas account for a disproportionate share of local variation. For example, the San Francisco (SF) Financial District and East Oakland (24% of study population) account for roughly 50% of local exposure variation for NO and BC and 40% for UFP and NO_2. These study areas represent denser urban settings with a greater mix of land uses. A similar comparison of purely spatial units (SSD among blocks, *SI Appendix*, Tables S10 and S11) finds that local variability is much greater for the primary pollutants NO and BC (76 and 79% of the SSD, respectively) and moderately higher for NO_2 and UFP (NO_2: 54% and UFP: 46%).

Our observations show a substantially expanded range of exposure—both between and within neighborhoods—than census block–level NO_2 predictions from a national-scale predictive

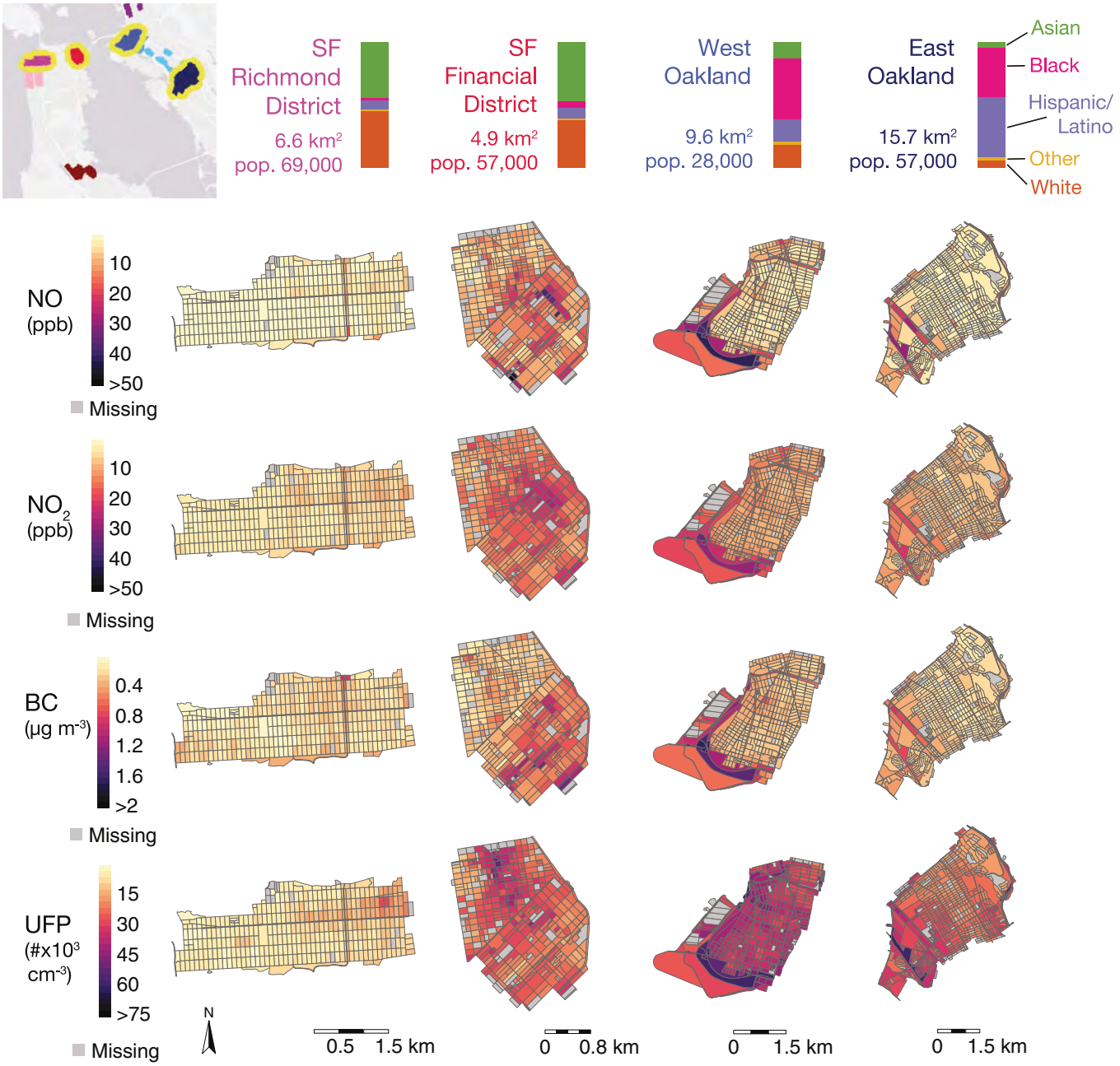

Fig. 1. Daytime median census block pollution levels for four selected study areas, calculated from ~50 visits to each block. For the bar plots indicating race/ethnicity, the White, Asian, Black, and Other race groups only include those identifying as non-Hispanic.

model similar to a land use regression (year 2015 integrated empirical-geographic [IEG] model) (17). The national IEG modeled estimates (Fig. 2, gray overlaid plots) show a ratio of 1.6 between highest and lowest neighborhood median exposure compared to 4.6 from mobile monitoring and a total-population interquartile range (IQR) of 2.2 parts per billion (IEG) compared to 6.1 ppb from mobile monitoring data (*SI Appendix*, Fig. S6). This result suggests that the national IEG model may miss some localized influences and may underestimate total population disparity and, by extension, the potential range of health risks. We find that the IEG model predicts higher median and mean exposure (2.8 ppb [36%] and 2.5 ppb [30%] higher, respectively) across the total population, compared to mobile monitoring data. No NO, BC, or UFP data at the same spatial resolution were publicly available at the time of this analysis (45).

Systematically Higher Concentrations for Black and Hispanic/Latino Groups Driven by Regional Variation. A stratification of pollutant exposure by study area and US Census–reported race and ethnicity (Fig. 3) reveals both a wide range in the measures of air pollution within each racial/ethnic group and systematic differences in exposure among groups. Fig. 3 highlights the role of regional demographic patterns in shaping the distribution of exposure across members of each racial/ethnic group. Study areas with a greater share of a given race/ethnicity are represented by larger areas within the shape of the distribution. For example, exposure for the Asian community within our study areas is dominated by neighborhoods in San Francisco (shades of pink), while exposure for Black residents is almost entirely determined by concentrations in Oakland neighborhoods (shades of blue). These aggregate exposure profiles reveal an overall pattern of racial/ethnic disparities:

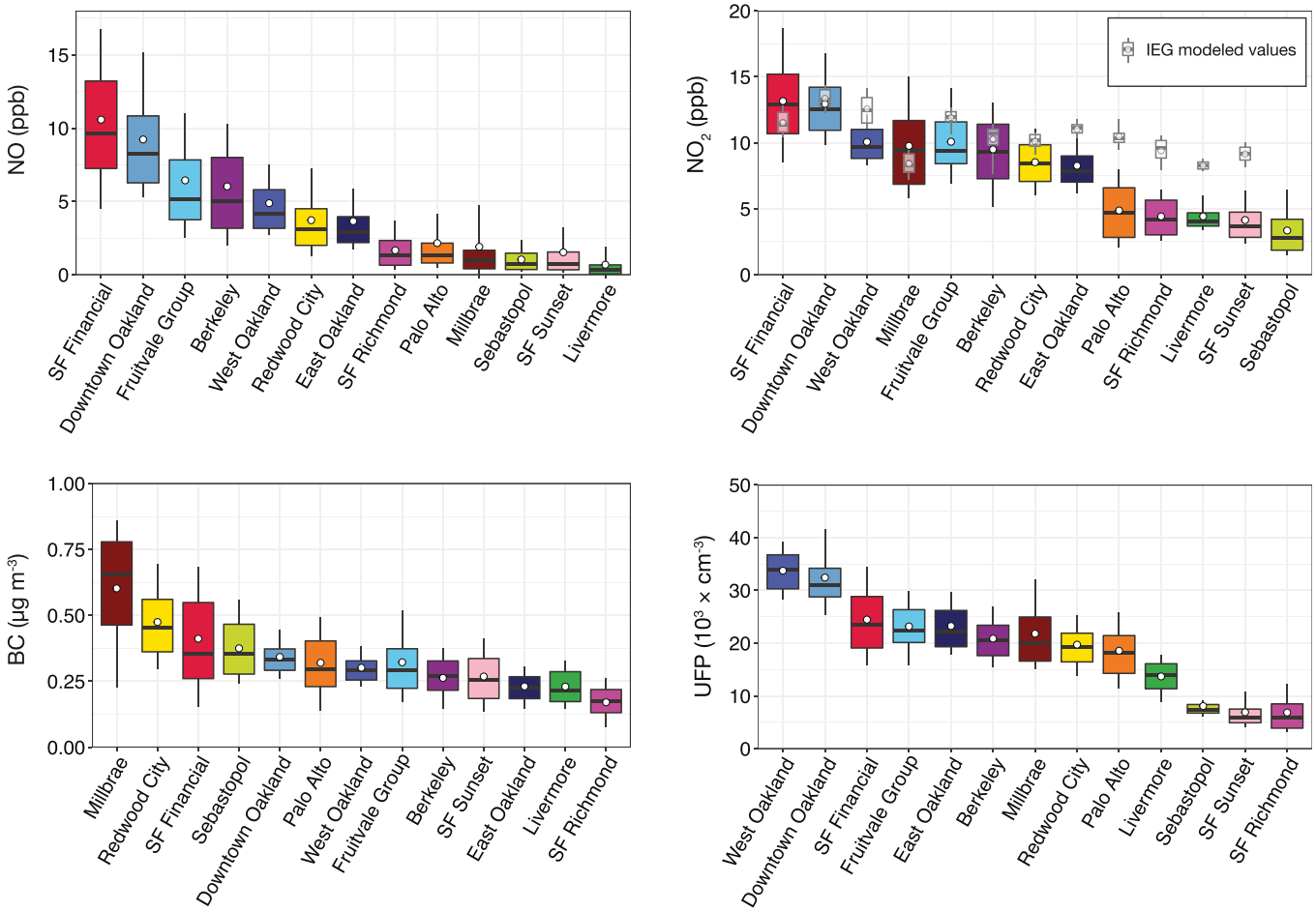

Fig. 2. Exposure distributions by neighborhood, based on daytime, weekday census block concentrations. Areas are shown in order of descending median concentration. Whisker ends represent 10th and 90th percentiles, box boundaries represent upper and lower quartiles, the center bar marks the median, and the circle represents the mean. Gray box plots in the NO_2 panel represent modeled exposure estimates from a national-scale IEG regression model (17).

higher concentration ranges in predominantly Black and Hispanic/Latino neighborhoods result in higher mean exposure for those groups. Notably, many of the neighborhoods with the highest average pollution exposures in our measurement dataset were subjected to overtly racially discriminatory housing policies (i.e., redlining) during the twentieth century (*SI Appendix*, Fig. S12).

At the local level, there is wide overlap among the exposure distributions of each racial/ethnic group (*SI Appendix*, Fig. S7). While study areas do show weak patterns of local disparity—census blocks with majority White residents showed modestly lower concentrations than blocks with majority Black and Hispanic/Latino residents within the same study area—there are not universal trends. In a few instances, a specific community shows exceptionally high exposure, as exemplified by the Asian populations of Downtown Oakland (all pollutants) and the SF Financial District (UFP). Both districts include densely populated corridors occupied by people of Asian descent.

In contrast, we find substantial disparity arising from regional pollution gradients combined with the high intermunicipal racial segregation in the Bay Area (46). On average, the White population is exposed to lower NO, NO_2, and UFP than other groups, with a median exposure 16 to 27% below the total-population median, while medians for the Black and Hispanic/Latino populations are higher by 8 to 30% depending on pollutant (Fig. 4*A* and *SI Appendix*, Table S13). The spatial detail provided by our method reveals nuance in disparity patterns beyond differences in medians. Fig. 4*B* illustrates the weighting of each racial/ethnic group within total-population exposure deciles. Overall, the White population is strongly overrepresented in the lowest deciles of the concentration distributions. The Asian population is overrepresented at the extremes, with the high end driven by the previously mentioned communities in Downtown Oakland and the SF Financial District and the low end driven by the Richmond and Sunset Districts. The Black and Hispanic/Latino populations are strongly underrepresented at the low end and concentrated toward the higher deciles, giving rise to higher average exposures for those groups. Apart from distinctly higher ranges of NO_2 and UFP exposure among Black and Hispanic/Latino populations, the range of exposures within racial/ethnic groups tends to be large compared with the range among groups. This finding holds especially for the Asian population (Fig. 4*A*), which is bimodally distributed between some of the cleanest (coastal) and most polluted (downtown) areas.

Within our study domain, the national IEG model also predicts lowest mean exposure for the White population and highest for the Black population, with moderately higher exposure for the Hispanic/Latino population (*SI Appendix*, Fig. S5). However, because of the narrower band of exposure estimates, IEG-modeled disparity differs from observations. The IEG model predicts a similar magnitude of exposure disparity between the population median and medians for those of Black and Hispanic/Latino descent, but a smaller difference for the White population, and does not show a disproportionate share of people of Asian descent in the highest-exposure categories. Thus, modeled

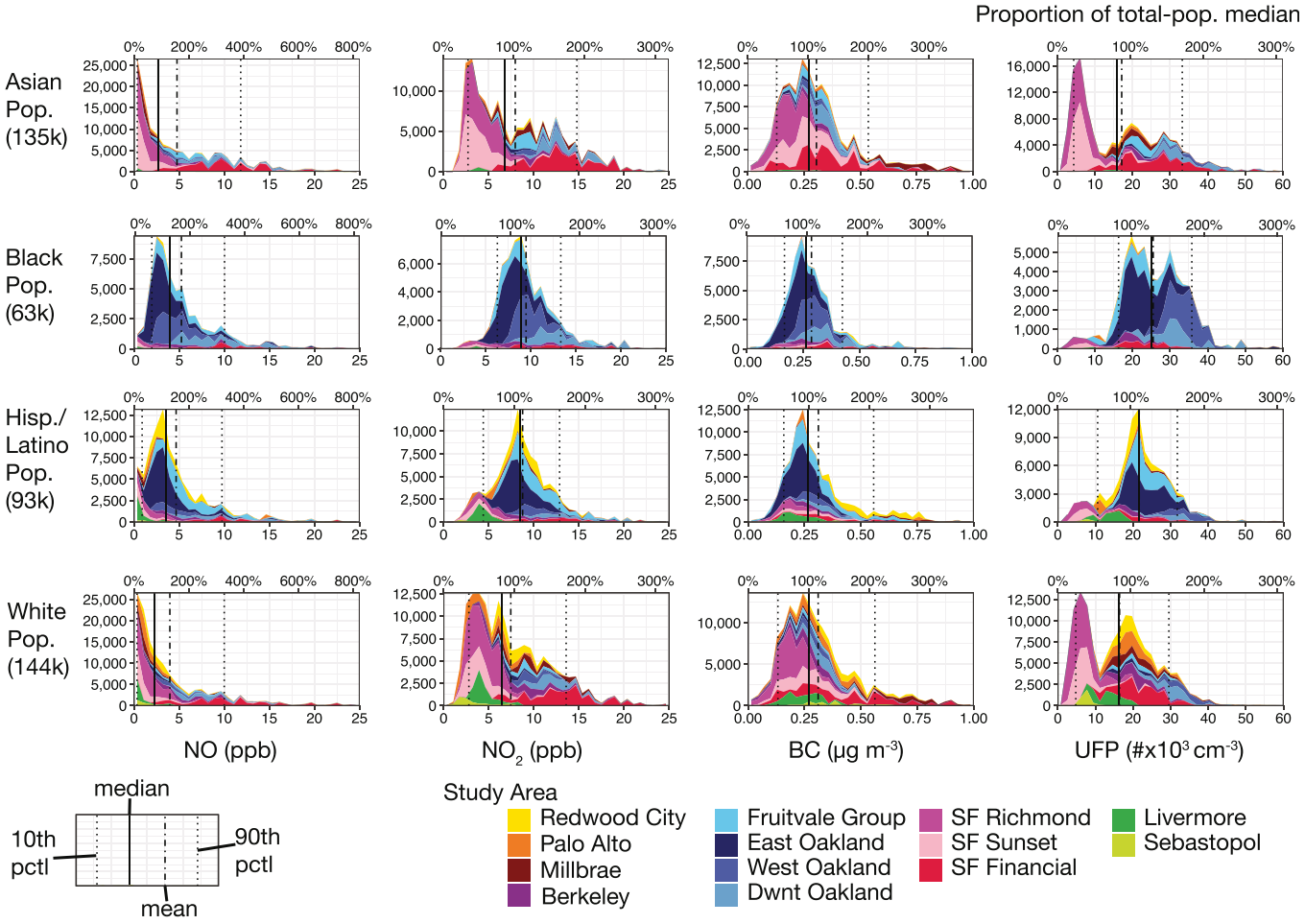

Fig. 3. Distribution of potential exposure stratified by race/ethnicity (not shown is the "Other" racial category, population: 10,000). The White, Asian, Black, and Other race groups only include those identifying as non-Hispanic. Height on the *y*-axis indicates the population at a given concentration level summed over all study areas. Because of significant differences in the racial and ethnic composition of each neighborhood, concentration distributions in some study areas contribute much more to specific race/ethnicities (e.g., East Oakland for the Hispanic/Latino population, West Oakland for the Black population). Vertical lines show the indicated statistics (mean, median, 10th, and 90th percentiles) for each race/ethnicity. Corresponding relative disparity values are provided in *SI Appendix*, Table S13.

disparity may miss an important dimension of racial/ethnic exposure disparity.

Our conclusions come with several caveats. First, mobile monitoring was conducted during daytime, weekday hours. Concentrations and spatial gradients may differ during nighttime and weekend hours due to changes in emission patterns and atmospheric dynamics (28, 33). Second, we assess here the implications of temporally averaged patterns, which are associated with chronic health effects. However, concentrations and spatial patterns show strong diurnal and seasonal variation that may have acute effects. While we believe that we sampled representative conditions (see *Materials and Methods*), temporal pollution dynamics could add another dimension to exposure disparity patterns. For this reason, we recommend that in-depth neighborhood monitoring include continuous fixed-site measurements in addition to mobile measurements (28). Third, we found that our conclusions are likely sensitive to the choice of neighborhoods, so we cannot assert that the disparity patterns we found are fully representative of conditions throughout the Bay Area (i.e., in unsampled neighborhoods). Finally, disparities in personal exposure to pollutants of outdoor origin depend not only on fine-scale patterns of outdoor air pollution—as measured here—but also personal mobility and the penetration of pollutants into the indoor environment. Future developments in measurement and modeling could usefully refine understanding of these dimensions of disparity. However, this study affirms that assessments of racial/ethnic air pollution disparities in the United States should include a broad range of representative neighborhoods that capture the varied experiences of those within each racial/ethnic group, considering the context of historic and present day municipal-level segregation.

Implications for Future Exposure Assessments and Environmental Justice Research. The problem of disparate exposures to environmental pollution has become the focus of renewed policy interest in the United States. Full-coverage mobile monitoring of air pollution can support policy advances in several ways. High-intensity mobile monitoring can reveal sources of localized hotspots (e.g., warehouses, metals processing, or restaurant clusters) that may be the subject of community concern but are not always detected by conventional monitoring (47). More broadly, the range and spatial variability of exposures revealed by mobile monitoring—not generally provided by conventional modeling and regulatory measurement networks—support the development of community-focused plans to improve air quality.

Our findings highlight the importance of identifying both highly localized patterns and intraurban disparity for addressing environmental injustice. We performed a set of stylized sensitivity analyses, described in *SI Appendix*, to characterize the degree to

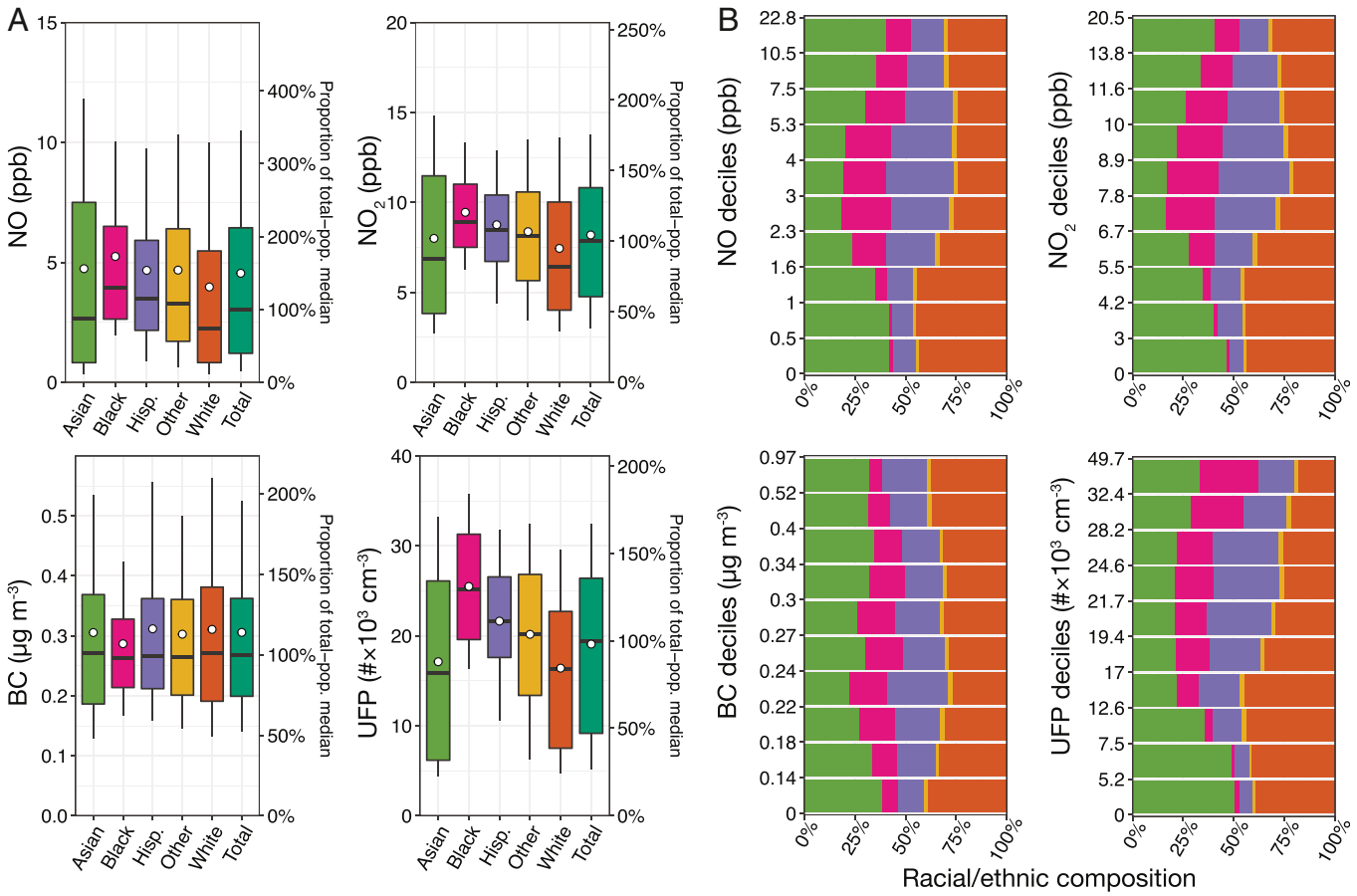

Fig. 4. Variation in total exposure distributions by race/ethnicity group. The distributions shown in the box-and-whisker plots (*A*) include the median (central bar), mean (white circle), upper and lower quartiles (box boundaries), and upper and lower 90th and 10th percentiles (whiskers). The division of the total-population exposure distribution into concentration deciles (*B*) shows the division of the population within the decile by race/ethnicity, with decile boundary concentrations indicated on the *y*-axis and the racial/ethnic color key provided by the box-and-whisker plots. The White, Asian, Black, and Other race groups only include those identifying as non-Hispanic.

which hyperlocal, local, and regional air pollution control interventions might help reduce racial/ethnic exposure disparities within the San Francisco Bay Area. Consistent with our findings that between-neighborhood patterns are the principal driver of racial/ethnic disparities, interventions that only eliminate within-neighborhood hotspots may not substantially reduce systematic disparity, although such interventions address the concentration extremes often experienced disproportionately by people of color. In contrast, interventions that reduce the differences in background concentrations among neighborhoods have higher potential to address population-wide racial/ethnic exposure disparities (*SI Appendix*). Higher-pollution areas often contain a variety of features that individually cause hyperlocal peaks and as a group contribute to elevated local background concentrations. Some examples of these features include greater heavy-duty vehicle traffic, adjacent high-emissions areas like ports and railyards, a high density of industrial operations, and a lack of green space. A community-centered approach to policymaking can account for issues at both hyperlocal and local scales by first identifying a suite of neighborhood features that contribute to higher pollution and then seeking local, regional, or state support for targeted interventions. Such interventions may include incentives and infrastructure to support zero-emission vehicles, accelerated emission control retrofits for point sources, and city planning to restrict new industrial facilities in disproportionately burdened communities. The differences in disparity patterns among pollutants further underscores the importance of multisource control policies. This community-centered approach is exemplified by the West Oakland "Owning Our Air" plan coauthored by the Bay Area Air Quality Management District, the West Oakland Environmental Indicators project, and a community-based steering committee and the broader Community Air Protection Program (AB617) in California, which includes West Oakland among 15 areas chosen for targeted interventions (47, 48).

Continued monitoring is necessary for evaluating policy effectiveness. Regional monitoring strategies may benefit from the multiscale approach of this study, as we identified some areas for which a single "background" monitor would represent neighborhood exposure within reasonable bounds, especially for secondary pollutants. Further investigation of the association of peaks and regional variation with specific emission sources and land uses is a useful direction for future research. We hypothesize that expanding this approach to other urban or rural settings would likely reveal different but similarly complex patterns and advance our understanding of the interaction between demographic patterns, pollution sources, and the built environment.

Environmental policy efforts in the coming years will increase the emphasis on addressing systemic racial/ethnic disparities in air pollution exposure. We illustrate here how environmental justice concerns arise from the confluence of multiscale patterns of emissions, concentrations, and populations within urban landscapes. Future efforts to address disparities will benefit from recent advances in both extensive fine-scale measurement—as we have demonstrated here—as well as source-oriented models

that identify the specific causes and drivers of exposure disparities. Broader solutions to environmental injustice will also involve greater partnership between scientists and community organizations, new approaches to synthesizing and communicating high-resolution data, and innovations in inclusive policy making.

Materials and Methods

Measurement Methods. Mobile monitoring was conducted with two Google Street View vehicles equipped with the Aclima mobile platform, which used research-grade instrumentation to provide 1-Hz measurements of BC, NO, NO_2, and UFP (*SI Appendix*, Table S14), and Global Positioning System coordinates with a nominal precision of ∼1 m. Detailed description of measurement and calibration protocol is contained in Apte et al. (21). The inlet and sampling manifold, mounted on the roof of each mobile laboratory, was designed to minimize self-sampling (measurements of the vehicle's own exhaust) as well as particle- and gas-phase sampling losses, with an estimated particle sampling loss of <5%. Raw measurement data were reviewed and processed to exclude periods of poor instrument performance (49). Data collection occurred over 32 mo from May 2015 through December 2017. The average nonhighway driving speed of 9 $m \cdot s^{-1}$ provides a nominal spatial resolution of ∼10 m.

Study areas were repeatedly visited on a rotating schedule designed to assess long-term average concentrations indicative of typical weekday, daytime conditions. During a visit, the driver would follow a Google Street View–based driving protocol to visit every road segment within the neighborhood at least once, driving with the normal flow of traffic. For large study areas, a subunit would be assigned for full coverage in a single day's driving, with full sampling occurring over multiple days. Visits to each area were distributed over different times of day and different seasons. Incidental temporal sampling bias was assessed using fixed-site regulatory monitors located in five of the sampling areas (*SI Appendix*)

Study Areas and Demographic Data. In total, 13 urban districts, cities, and towns ranging in size from 2.4 to 15.7 km^2 (total: 93 km^2) were selected as study areas to provide a range of land uses (e.g., industrial, commercial, dense residential, and light residential), share of open or green space, traffic density, demographic composition, and historical housing policy (*SI Appendix*, Figs. S1, S2, S12, and S13 and Tables S1–S4). Study areas were distributed within the counties of San Francisco, Alameda, San Mateo, and Santa Clara in the San Francisco Bay Area, with one background location in Sonoma County (*SI Appendix*, Fig. S5 and Table S6). Sampling areas include between 95 and 930 census blocks (median: 447), depending on population density and spatial extent (*SI Appendix*, Table S4).

Population data from the US Census Bureau were retrieved from the IPUMS National Historical Geographic Information System for the year 2010, the most recent year for which block-level data are available (50). Using the racial/ethnic designations provided by the US Census, we categorize all individuals identifying as Latino and/or Hispanic in one group, and then categorize non-Hispanic individuals by race: Asian, Black, White, and "Other," including those of Native American, Pacific Islander, multiracial, or other racial identity. The racial composition of our study population is broadly representative of the Bay Area as a whole, although it includes more neighborhoods with a high proportion of Black residents (*SI Appendix*, Fig. S2).

Data Processing. Concentrations for each block are calculated as the median of surrounding roads, typically located within 50 to 100 m from the block center point (*SI Appendix*, Fig. S3). The geographic assignment of on-road measurements to census blocks involved a two-step process. First, concentrations were calculated for road segments: the raw longitude and latitude values associated with each 1-Hz measurement were "snapped" to 30 m road segment points, as described by Apte et al. (21). To give equal weight to each pass of a road segment and minimize the influence of transient extreme events (e.g., vehicle plumes), measurements at each road segment were averaged per drive pass: all consecutive measurements made at a single road segment were summarized as a single mean value, following methods described in Messier et al. (43). Long-term concentrations at each road segment were calculated as the median of those drive pass means, with the median statistic chosen to further reduce the influence of outlier events.

Second, census block concentrations were calculated as the median of concentrations at every adjacent or intersecting 30-m road segment, using 10-m buffer to capture road segments a small distance from the census block edge. This dataset of census block concentrations and demographic data has been posted to an online repository (51). Blocks vary in size and shape but typically cover an area of ∼0.01 km^2 and are surrounded by roads, with a median perimeter of 447 m. Accordingly, census block estimates integrated measurements from 15 to 20 road segments (*SI Appendix*, Fig. S3). In some cases—near highways and strong point sources—pollution gradients may vary over finer spatial scales than those captured by census block spatial units (∼110 m). However, the integration of multiple road segments provides an increase in total number of visits and total sampling time per spatial unit, which reduces sampling error and measurement uncertainty (52). While on-road measurements are not a perfect approximation of concentrations throughout a census block, a previous comparison of on- and near-road measurements in West Oakland showed no evidence of bias in on-road concentrations due to increased proximity to on-road emissions (52). The general spatial representativeness of on-road measurements holds especially in low-traffic residential neighborhoods with mixed wind conditions (34).

Uncertainty and Sensitivity Analyses. Because of the temporal sparsity of mobile measurements relative to fixed-site monitoring, it is critical to consider both the uncertainty inherent in a small sample size and the potential for temporal bias based on the days and times of day sampled. To mitigate temporal bias from lower sampling rates, we imposed the following standards for all data used in this analysis: Road segments were included if valid measurements were made on at least 3 d, and census blocks were only included if they integrated measurements from three or more road segments and included a total of 100 s or more of valid sampling data. The median cumulative sampling time of each block was 19 min, collected during a median of 47 unique visits over 20 d.

A bootstrap resampling technique, described in *SI Appendix*, was used to estimate uncertainty at different sampling intensities. The characteristic sampling intensity for each study area is provided in *SI Appendix*, Table S4. A Monte Carlo subsampling analysis of an exceptionally highly sampled area (West Oakland) shows that with moderate sampling intensity (each block visited 10 to 20 d) provides a 90% CI around the population-weighted median concentration of approximately ± 20% for NO_2 and UFP, ± 30% for BC, and ± 40% for NO, with CIs that are positively skewed (*SI Appendix*, Table S15). At higher sampling intensity (20 to 40 d), the 90% CI is ± 20% for the medians of all pollutants. Considering the influence of sampling intensity on estimates of within-neighborhood variability, we find that even at high sampling intensity the interdecile range is overestimated in 95% of simulations, affecting both estimates of local variance and the total-population variance. Despite this effect, we find that study areas that accounted for large portions of SSD for NO, NO_2, and UFP included some of the most-sampled (Downtown Oakland and Berkeley) as well as moderately sampled areas (SF Financial and Fruitvale Group; *SI Appendix*, Table S4). Nevertheless, this finding serves as a caution against overinterpreting variation in less-sampled areas.

Diurnal sampling bias metrics show that sampling of NO_2, BC, and UFP was broadly representative (± 15%) of daytime conditions, while the distribution of hourly measurements resulted in a systematic low bias in NO concentrations (*SI Appendix*, Table S16). Error tended to be in the same direction across all study areas (e.g., 12 out of 13 are biased ∼10% lower for BC) and thus had a small effect on the relative rankings of concentrations among study areas (*SI Appendix*, Fig. S8). Annual representativeness ratios show less consistent patterns across study areas, and the application annual adjustment factor results in minor changes in the relative rankings of NO_2 and BC among study areas and increases the ranges of NO and UFP exposure distributions among high concentration study areas. Overall, these findings are robust against the effects of temporal bias.

Data Availability. A dataset including concentrations and demographics by census block has been deposited in figshare (10.6084/m9.figshare.15070314) (51). All other study data are included in the article and/or supporting information.

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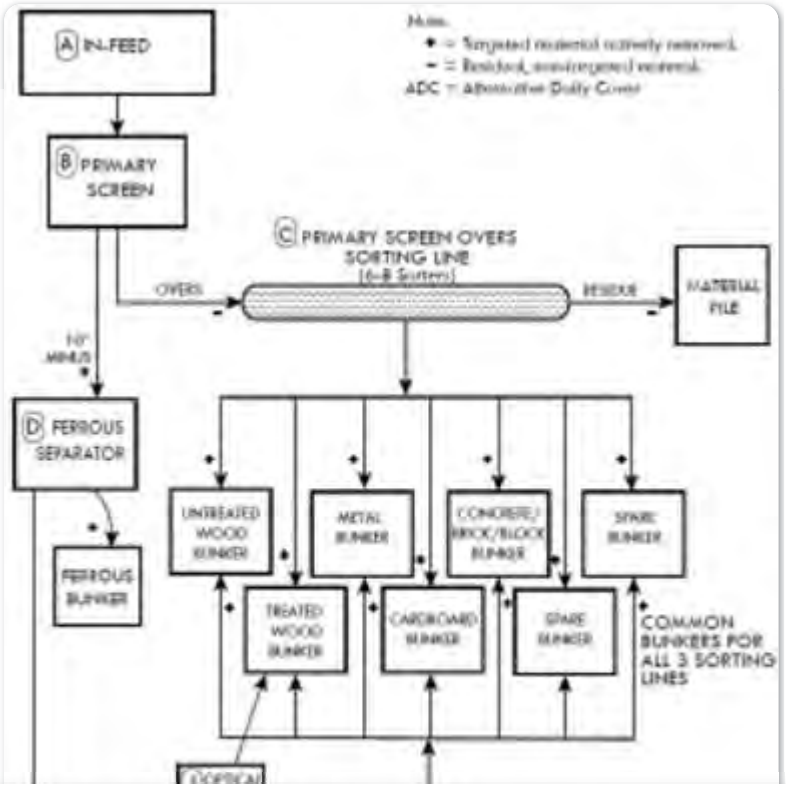
1. A. A. Karner, D. S. Eisinger, D. A. Niemeier, Near-roadway air quality: Synthesizing the findings from real-world data. *Environ. Sci. Technol.* **44**, 5334–5344 (2010).
2. P. Gu *et al*., Intracity variability of particulate matter exposure is driven by carbonaceous sources and correlated with land-use variables. *Environ. Sci. Technol.* **52**, 11545–11554 (2018).
3. Y. Wang *et al*., Spatial decomposition analysis of NO2 and PM2.5 air pollution in the United States. *Atmos. Environ.* **241**, 117470 (2020).
4. J. G. Su, M. Jerrett, R. Morello-Frosch, B. M. Jesdale, A. D. Kyle, Inequalities in cumulative environmental burdens among three urbanized counties in California. *Environ. Int.* **40**, 79–87 (2012).
5. D. Houston, W. Li, J. Wu, Disparities in exposure to automobile and truck traffic and vehicle emissions near the Los Angeles-Long Beach port complex. *Am. J. Public Health* **104**, 156–164 (2014).
6. M. R. Jones *et al*., Race/ethnicity, residential segregation, and exposure to ambient air pollution: The Multi-Ethnic Study of Atherosclerosis (MESA). *Am. J. Public Health* **104**, 2130–2137 (2014).
7. R. Morello-Frosch, B. M. Jesdale, Separate and unequal: Residential segregation and estimated cancer risks associated with ambient air toxics in U.S. metropolitan areas. *Environ. Health Perspect.* **114**, 386–393 (2006).
8. N. P. Nguyen, J. D. Marshall, Impact, efficiency, inequality, and injustice of urban air pollution: Variability by emission location. *Environ. Res. Lett.* **13**, 024002 (2018).
9. C. W. Tessum *et al*., Inequity in consumption of goods and services adds to racial-ethnic disparities in air pollution exposure. *Proc. Natl. Acad. Sci. U.S.A.* **116**, 6001–6006 (2019).
10. C. W. Tessum *et al*., $PM_{2.5}$ polluters disproportionately and systemically affect people of color in the United States. *Sci. Adv.* **7**, eabf4491 (2021).
11. A. J. Cohen *et al*., Estimates and 25-year trends of the global burden of disease attributable to ambient air pollution: An analysis of data from the Global Burden of Diseases Study 2015. *Lancet* **389**, 1907–1918 (2017).
12. D. E. Schraufnagel *et al*., Air pollution and noncommunicable diseases: A review by the forum of international respiratory societies' environmental committee, part 2: Air pollution and organ systems. *Chest* **155**, 417–426 (2019).
13. X. Wu, R. C. Nethery, M. B. Sabath, D. Braun, F. Dominici, Air pollution and COVID-19 mortality in the United States: Strengths and limitations of an ecological regression analysis. *Sci. Adv.* **6**, eabd4049 (2020).
14. A. van Donkelaar, R. V. Martin, C. Li, R. T. Burnett, Regional estimates of chemical composition of fine particulate matter using a combined geoscience-statistical method with information from satellites, models, and monitors. *Environ. Sci. Technol.* **53**, 2595–2611 (2019).
15. M. S. Hammer *et al*., Global estimates and long-term trends of fine particulate matter concentrations (1998-2018). *Environ. Sci. Technol.* **54**, 7879–7890 (2020).
16. G. Hoek, Methods for assessing long-term exposures to outdoor air pollutants. *Curr. Environ. Health Rep.* **4**, 450–462 (2017).
17. S.-Y. Kim *et al*., Concentrations of criteria pollutants in the contiguous U.S., 1979 - 2015: Role of prediction model parsimony in integrated empirical geographic regression. *PLoS One* **15**, e0228535 (2020).
18. J. Peters, J. Theunis, M. V. Poppel, P. Berghmans, Monitoring PM_{10} and ultrafine particles in urban environments using mobile measurements. *Aerosol Air Qual. Res.* **13**, 509–522 (2013).
19. A. P. Patton *et al*., Spatial and temporal differences in traffic-related air pollution in three urban neighborhoods near an interstate highway. *Atmos. Environ.* **99**, 309–321 (2014).
20. N. Baldwin *et al*., Factors affecting pollutant concentrations in the near-road environment. *Atmos. Environ.* **115**, 223–235 (2015).
21. J. S. Apte *et al*., High-resolution air pollution mapping with Google Street View cars: Exploiting big data. *Environ. Sci. Technol.* **51**, 6999–7008 (2017).
22. P. Deshmukh *et al*., Identifying air pollution source impacts in urban communities using mobile monitoring. *Sci. Total Environ.* **715**, 136979 (2020).
23. G. Oberdörster, A. Elder, A. Rinderknecht, Nanoparticles and the brain: Cause for concern? *J. Nanosci. Nanotechnol.* **9**, 4996–5007 (2009).
24. S. E. Alexeeff *et al*., High-resolution mapping of traffic related air pollution with Google street view cars and incidence of cardiovascular events within neighborhoods in Oakland, CA. *Environ. Health* **17**, 38 (2018).
25. E. F. Kirrane *et al*., A systematic review of cardiovascular responses associated with ambient black carbon and fine particulate matter. *Environ. Int.* **127**, 305–316 (2019).
26. HEI Review Panel on Ultrafine Particles, "Understanding the health effects of ultrafine particles" (2013) (4 October 2020).
27. R. U. Shah *et al*., Socio-economic disparities in exposure to urban restaurant emissions are larger than for traffic. *Environ. Res. Lett.* **15**, 114039 (2020).
28. J. J. Caubel, T. E. Cados, C. V. Preble, T. W. Kirchstetter, A distributed network of 100 black carbon sensors for 100 days of air quality monitoring in West Oakland, California. *Environ. Sci. Technol.* **53**, 7564–7573 (2019).
29. R. M. Brice, J. F. Roesler, The exposure to carbon monoxide of occupants of vehicle moving in heavy traffic. *J. Air Pollut. Control Assoc.* **16**, 597–600 (1966).
30. G. A. Petersen, R. H. Sabersky, Measurements of pollutants inside an automobile. *J. Air Pollut. Control Assoc.* **25**, 1028–1032 (1975).
31. D. Westerdahl, S. Fruin, T. Sax, P. M. Fine, C. Sioutas, Mobile platform measurements of ultrafine particles and associated pollutant concentrations on freeways and residential streets in Los Angeles. *Atmos. Environ.* **39**, 3597–3610 (2005).
32. K. H. Kozawa, S. A. Fruin, A. M. Winer, Near-road air pollution impacts of goods movement in communities adjacent to the Ports of Los Angeles and Long Beach. *Atmos. Environ.* **43**, 2960–2970 (2009).
33. S. Hu *et al*., Observation of elevated air pollutant concentrations in a residential neighborhood of Los Angeles California using a mobile platform. *Atmos Environ (1994)* **51**, 311–319 (2012).
34. I. Levy *et al*., Elucidating multipollutant exposure across a complex metropolitan area by systematic deployment of a mobile laboratory. *Atmos. Chem. Phys.* **14**, 7173–7193 (2014).
35. S. Hankey, J. D. Marshall, On-bicycle exposure to particulate air pollution: Particle number, black carbon, PM2.5, and particle size. *Atmos. Environ.* **122**, 65–73 (2015).
36. L. M. Zwack, C. J. Paciorek, J. D. Spengler, J. I. Levy, Characterizing local traffic contributions to particulate air pollution in street canyons using mobile monitoring techniques. *Atmos. Environ.* **45**, 2507–2514 (2011).
37. G. S. W. Hagler *et al*., Field investigation of roadside vegetative and structural barrier impact on near-road ultrafine particle concentrations under a variety of wind conditions. *Sci. Total Environ.* **419**, 7–15 (2012).
38. R. W. Baldauf *et al*., Air quality variability near a highway in a complex urban environment. *Atmos. Environ.* **64**, 169–178 (2013).
39. J. Steffens *et al*., Near-port air quality assessment utilizing a mobile measurement approach. *Atmos. Pollut. Res.* **8**, 1023–1030 (2017).
40. H. L. Brantley *et al*., Characterization of spatial air pollution patterns near a large railyard area in Atlanta, Georgia. *Int. J. Environ. Res. Public Health* **16**, 535 (2019).
41. R. U. Shah *et al*., High spatial resolution mapping of aerosol composition and sources in Oakland, California using mobile aerosol mass spectrometry. *Atmos. Chem. Phys.* 10.5194/acp-2018-703 (2018).
42. Z. Li, J. C. H. Fung, A. K. H. Lau, High spatiotemporal characterization of on-road $PM_{2.5}$ concentrations in high-density urban areas using mobile monitoring. *Build. Environ.* **143**, 196–205 (2018).
43. K. P. Messier *et al*., Mapping air pollution with Google Street View cars: Efficient approaches with mobile monitoring and land use regression. *Environ. Sci. Technol.* **52**, 12563–12572 (2018).
44. S. Weichenthal *et al*., Within-city spatial variations in ambient ultrafine particle concentrations and incident brain tumors in adults. *Epidemiology* **31**, 177–183 (2020).
45. V. A. Southerland *et al*., Assessing the distribution of air pollution health risks within cities: A neighborhood-scale analysis leveraging high-resolution data sets in the Bay Area, California. *Environ. Health Perspect.* **129**, 37006 (2021).
46. Othering & Belonging Institute, Racial Segregation in the San Francisco Bay Area (2018). https://belonging.berkeley.edu/segregationinthebay. Accessed 20 August 2020.
47. Bay Area Air Quality Management District, West Oakland Environmental Indicators Project, Owning Our Air: The West Oakland Community Action Plan (2019). https://www.baaqmd.gov/~/media/files/ab617-community-health/west-oakland/100219-files/owning-our-air-plan-summary-pdf.pdf. Accessed 22 August 2021.
48. CARB, *California Air Resources Board* (Community Air Protection Blueprint, 2018).
49. M. Lunden, B. LaFranchi, Metadata for Aclima-Google California AQ mapping dataset" (2017). https://docs.google.com/document/d/1qOR8j4ORSV9VhaAUVuLDnf9nfLh_RGbd7rCycXh3U3s. Accessed 9 November 2020.
50. S. Manson, J. Schroeder, D. Van Riper, S. Ruggles, *IPUMS National Historical Geographic Information System: Version 14.0 [Database]* (IPUMS, 2019).
51. S. E. Chambliss *et al*., Bay Area mobile monitoring multi-pollutant block Medians. *Figshare*. https://figshare.com/articles/dataset/Bay_Area_Mobile_Monitoring_Multi-pollutant_Block_Medians/15070314. Deposited 30 July 2021.
52. S. E. Chambliss *et al*., Comparison of mobile and fixed-site black carbon measurements for high-resolution urban pollution mapping. *Environ. Sci. Technol.* **54**, 7848–7857 (2020).

BMPS

New Life for C&D

Marc J. Rogoff and Bruce J. Clark

Aug. 23, 2013



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In recent years, materials recovery facilities designed for construction-and demolition-debris (C&D) have rapidly emerged as a major trend by communities to increase their overall landfill diversion rates.

C&D is the material generated from the demolition of buildings and other similar civil works infrastructure, the vegetation from land clearing, including rocks and soil, and the residual material from the construction of a structure. Thus, specific types of materials could include:

- ***Metals***-reinforcing steel, steel shapes, wire, caps, conduit, etc.

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- Green Infrastructure
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- Sediment Control

wire, cans, conduit, etc.

- ***Concrete***, brick, stone, tile, rocks, and soil
- ***Plastics***-bottles, buckets, containers, drums, wrapping, etc.
- ***Rope***
- ***Paper*** and cardboard
- ***Wood*** and dimensional lumber, pallets
- ***Ceramics***
- ***Gypsum drywall***
- ***Plaster***
- ***Asphalt***
- ***Rubber***
- ***Foam***, insulation, packing, etc.
- ***Adhesive***, glues, paint, etc.
- ***Vegetation***

Managing municipal solid waste is more than landfilling: publicity, education, engineering, long-term planning, and landfill gas waste-to-energy are specialties needed in today's complex environment. We've created a handy infographic featuring 6 tips to improve landfill management and achieve excellence in operations. 6 Tips for Excellence in Landfill Operations. Download it now!

C&D waste presents unique recycling challenges because of its nature. It is inherently a relatively dry waste that, when agitated and processed, produces a lot of dust. Materials can be

⊕ Slope Stabilization

⊕ Vegetation Management

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combined and intertwined in a way that makes separating them difficult. Most of the material is bulky, heavy, and/or abrasive. It can cause extensive wear and tear on machinery and can also be a safety hazard to plant workers if proper controls are not observed.

Recycling of parts of demolished buildings has been conducted since the turn of the 20th century. The reuse of steel beams and columns in automobile bodies is one example, as is the reuse of bricks. However, several factors that emerged in the 1980s have stimulated the growth of C&D recycling into a large and varied sector of industry encompassing many other materials. These factors include:

- Sustained cycles of significant new construction in many major cities
- The development of relatively inexpensive C&D landfills to take residual material left over from the recycling process
- The emergence of other markets for recycled materials, a result of the increased use of alternative feedstock for energy production
- The development and refinement of automated machinery to process C&D derived from municipal wastestreams
- The development of formal “green” building development initiatives that advocate recycling of municipal waste

advocate recycling of municipal waste

Managing municipal solid waste is more than landfilling: publicity, education, engineering, long-term planning, and

landfill gas waste-to-energy are specialties needed in today's complex environment. We've created a handy infographic featuring 6 tips to improve landfill management and achieve excellence in operations. 6 Tips for Excellence in Landfill Operations. Download it now!

Most C&D recycling operations in the past relied mostly on manual labor to sort materials. The current generation of these recycling facilities can consistently remove as much as 90%, or more, of the recyclable material from the wastestream, using mostly automated machinery, resulting in a very low production of residual material left over for landfilling. For specific tasks where the benefit/cost for replacing a worker with a machine can be as effective, manual labor is not economically attractive.

Waste Processing and Recycling Operations

Most of the automated machinery used in current C&D plants has been derived from the basic materials mining industry and modified for the commercial market. In some cases, new

the commercial market. In some cases, new equipment has been designed specifically for municipal C&D materials. The planning and design of a recycling plant is based on many factors that include the following:

- Volume of C&D material projected to be managed over the next five to 10 years
- The reuse markets that exist or can be developed for recycled materials and the revenue potential
- Location, size, and availability of property relative to the market being served
- Development and operating budget (this includes the use of manual labor and automated equipment)

The primary goal in a C&D recycling facility is to be highly efficient, meaning extract as much of the valuable material as possible while producing the lowest amount of residual material. For a privately owned recycling business, this is strictly an economic issue. When the total cost of processing C&D waste to remove a specific material exceeds the revenue generated from the sale of that material to the reuse market, then the material is not valuable from a recycling standpoint. Accordingly, the disposal of residual material in the landfill is a cost to the plant operator, so reducing residual disposal costs enhances the net revenue from the

disposal costs enhances the net revenue from the sale of recyclable materials.

The economic "equation" for a municipally owned/operated facility is a little different, in that this would be a public service, and demonstrating a healthy "profit" is not the prime focus. However, the process efficiency goal is still the same whether the facility is private or publicly owned.

What is determined to be a valuable material can change over time as the market and, more specifically, the prices paid change for various materials. Thus, the recycling plant must have some flexibility built into the separation process and the overall "floor plan layout" in order to be able to process a variety of valuable material and be responsive to changing markets.

Processing Schemes

Design of a processing system for C&D waste follows an approach that, in general, encompasses the following basic principles listed below, starting at the waste tipping area, progressing through mechanized and manual separation then to the temporary storage areas for each targeted material. Some or all of these may be incorporated depending on the factors

previously mentioned. Keep in mind there are many different technologies and equipment that can accomplish your goals and we could not

cover them all. We have only described herein a few proven techniques.

A schematic illustrating the basic material separation and flow processes is provided in Figure 1. Following along with that exhibit, the descriptions below correspond to the letters designating certain features on the exhibit:

- Initial agitation to start to separate the mix of material to improve productivity of downstream processing operations (B)
- Separation of smaller material from larger material. This split usually is around an approximate diameter of 8 inches to 12 inches (20 to 30 cm) (B).
- Separation of ferrous metal to prevent damage to downstream equipment (D)
- Separation of small stones, glass pieces, sand, and grit to reduce wear and tear on downstream equipment (E)
- Separation of light, “two-dimensional” materials (e.g., paper, cardboard, and plastic film products) from smaller and/or denser “three-dimensional” materials (e.g., small containers, bottles, other objects, pieces of nonferrous metal, sticks of wood) (E)

wood) (F)

- Separation of wood/lumber, either with optical machinery (J) or water bath

- Separation of nonferrous metal, other wood, concrete pieces, asphalt, drywall, etc. This is usually done manually on a “picking line” conveyor system (described below) (C, G, and I).

Bulk Material Recycling

The recycling of large pieces of concrete, steel-reinforced concrete, and asphalt is a process that requires specially designed, heavy-duty machines. Concrete and asphalt recycling is either done at a permanent site, or a mobile processing system can travel to the specific site if there is an extensive volume of material and bulky and heavy pieces that would be cost prohibitive to transport to the recycler. These machines can remove and recycle the steel reinforcing bars and crush the remaining concrete and aggregate to specific sizes. A bulk operation may be separate from an operation receiving a mixed wastestream with smaller pieces because of the land area required for the storage piles at the former operations.

A somewhat unique activity that is common in south Florida is the filling of rock-pit lakes with certain C&D material considered to be “clean debris.” Clean debris is clean concrete, brick

debris. Clean debris is clean concrete, brick, soil, and rocks. Rock-pit lakes are very common in the region and are excavated to obtain the limestone, which is a popular and inexpensive source for concrete aggregate and for use in road subgrade. Later, after the quarry operation moves away, some of the lakes are back-filled with clean debris to make buildable ground.

Screening Operations

Finger screen-C&D waste is often a clump of different sizes and types of materials intertwined together. These clumps must be separated at the front end of the process in order to make the separation and removal of the specific material as efficient as possible in downstream operations and to reduce the potential for downstream machinery to become clogged.

A popular and effective device known as a *finger screen* is often used for this purpose. A finger screen is a heavily built, mechanized conveyor that uses an articulated, oscillating floor to break up the waste clumps. The waste material is fed into one end and the combination of oscillating motion, and the downward slope of the conveyor induces the waste to move forward and at the same time start to separate (Figure 2).

Credit: Keith Howard
Figure 2. Vibrating finger screen

At the downstream end of the screen, another feature can provide a rough separation of waste by size and density. If this division is desired, the conveyor floor has an opening in which smaller and denser materials (e.g., stones, wood, and metal pieces, etc.) fall through the opening and are dropped onto another conveyor that takes them to another sorting process. Lighter, flatter, and/or bulkier materials (e.g., cardboard, paper, plywood, plastic film) are carried off the end of the screen conveyor and into the next piece of sorting equipment.

Trommel screen-A trommel screen can be used to accept the material from a finger screen. If no size and density separation occurred on the finger screen, then the trommel can be used for this purpose. A trommel is a device that contains a hollow perforated tube through which the waste moves. The trommel tube is perforated along its length and its circumference with holes of a specific diameter. The tube is angled similarly to the finger screen and is slowly turned on rollers by a motor. As the waste moves through the tube, the rotating action and downward slope moves the materials forward where they encounter the perforations. Pieces of waste smaller than the perforations (known as *unders*) will fall through the holes and onto a conveyor that will take that material to another

sorting process. The larger pieces of material (known as the *overs*) will move through the trommel and onto the next sorting process.

Star gear screen-Another mechanized device used to separate waste into two size fractions is the star gear screen. A star gear is a conveyor that contains multiple rows of gears affixed to rotating axles. Each axle contains multiple gears spaced at precise intervals along the axle. The spacing of the gears is designed to control the separation of material. As waste material moves into the upstream end of the screen, it is propelled forward by the rotating gear assemblies. As the material moves over the gears, pieces that are more dense and that are smaller than the opening between the gears fall though opening. These may include stones, pieces of wood, metal, plastic, brick, and similar material. Those pieces (the unders) are collected by another conveyor and taken for further sorting or disposal. Lighter pieces that are larger than the opening (the overs) continue to move along the conveyor, essentially floating over the gears to the downstream end, where they flow into the next process. Overs may include cardboard, other papers, pieces of lumber, drywall, plastic film, and similar materials.

Ferrous Metal Separation and Removal

If large amounts of heavy ferrous metal objects are anticipated in the wastestream, then a device known as an *overhead magnet* is used to remove them from the other wastes. This device is essentially a large industrial magnet surrounded by a continuous looped conveyor belt. The entire device is suspended on a structural frame positioned perpendicular (at a 90-degree angle) above a conventional conveyor belt. Ferrous objects on the conventional conveyor passing under the energized magnet are pulled from the belt onto the magnet's moving belt and are swept away from the lower conveyor. As the looped belt passes out of the magnetized zone, the object is released and falls into a container for removal.

Other configurations of magnets are available for removal of such smaller and lighter types of ferrous waste as metal cans. These magnets, known as *pulley head magnets*, can be located within the end rotating pulley system of a conventional conveyor belt system. These magnets operate on a similar principle to the overhead magnet. Cans on the conveyor belt passing over the magnet are retained on the belt as it travels around the pulley to a point almost directly under the belt where the magnetic field ends. At that point the can falls off the belt and into a container. Nonferrous material will flow off the end of the belt and with the help of the

belts momentum, will take a trajectory away from the conveyor into a separate container.

Heavy and Light Material Separation

De-stoner/air knife-Separation of such light, "two-dimensional" materials as paper, textile plastic film, aluminum, and pieces of cardboard from heavier, bulkier materials can be accomplished in an automated device called a de-stoner/air knife. Figure 3 shows a schematic cross-section through a unit with two air knifes. This unit uses directional air currents and a vibratory motion to stratify and separate lighter material from heavier material. The unit is mounted on heavy coil springs to reduce the transmission of vibration to the ground.

Figure 3. Crosssection of Destoner/airknife

Following along on this exhibit, the waste is fed in from a star gear screen and immediately encounters a gap in the air knife through which high-velocity but low-pressure air is flowing. The airflow is provided by a standard centrifugal blower. This flow of air blows the lighter and two-dimensional materials up and toward the downstream conveyor. The heavier and bulkier materials, including small stones and pieces of glass and metal, are unaffected by the air current, and fall through the gap onto a take

current, and fall through the gap onto a take-away conveyor. Thus, the designation as a *de-stoner*.

As the lighter materials are conveyed to the end of the air knife, there is another high-velocity, low-pressure air stream directed through the conveyor. This final current of air separates the very light material (mostly plastics and light paper) from other denser bulkier material. The lighter material, with a relatively large cross-sectional area, is carried to the far end of the collection bin, while the denser, more compact material does not travel as far and drops into the bin directly at the end of the conveyor. All air is exhausted out the top of the air box. If a hood is not used over the final air discharge to capture the light product, then a grate or similar screen is used to deflect light material into the air box end bin.

Exhaust air can be captured and rerouted back to the blowers to increase efficiency and reduce discharge of dust to the environment to near zero. This feature is recommended when the unit is used inside a building and outside where migrating dust could pose a nuisance to other operations or adjacent businesses.

Optical sorter-The use of automated machinery employing “electronic eyes” to assist in separating materials has become

assist in separating materials has become increasingly popular. In many cases the increased efficiency and higher purity of the targeted material provides a positive

benefit/cost ratio as compared with traditional manual methods. The function of the equipment is based on the principle that all solid materials have a unique surface “signature” that reflects and absorbs light rays in varying amounts.

Figure 4. Cross-section of optical sorter

Figure 4 shows a general cross-section of an optical sorter than can be used to separate out large wood pieces.

Following along on this exhibit, as the waste material enters from the conveyor and passes under the control unit, a bright light illuminates the materials. A sophisticated instrument called a spectrometer imbedded in these machines “reads” the reflected light from the materials and, through a computerized interface tuned to see wood product, actuates a compressed-air device that sends a blast of air channeled by the computer program to specific multiple ports positioned across the end of the conveyor belt. As the materials pass over the ports, the ports that have been activated will discharge a blast of air under the material. The air ejects larger wood pieces to a receiving hopper on the far end of the machine, while non-wood material simply rolls

off the end of the conveyor belt into a separate hopper.

A *vibrating pan feed conveyor is often used to feed the optical sorter.* This type of conveyor will increase the effectiveness of the optical sorter by flattening out and separating the materials before they enter the electronic eye detection zone of the sorter.

Picking (sorting) line-Some separation of materials is still best achieved with manual labor. The *picking line*, or sorting line, is a mostly manual work station that is very common in C&D recycling facilities. Picking lines are used for removal of many potentially valuable recyclable materials, including but not limited to flat pieces of wood, nonferrous metals, asphalt and concrete pieces, and cardboard. A picking line is typically a long and narrow steel work platform elevated at least 10 feet or more above the normal working floor (Figure 5). Under the platform, steel or concrete walls form side-by-side bunkers and are used to provide additional platform support. Each bunker is dedicated to temporary storage of a specific material.

Credit: Keith Howard
Figure 5. Semi-automated picking/sorting line

One or more conveyors are mounted on the platform, parallel to its length, and workers stand along the length of the conveyor, usually

on both sides. The conveyor is set at a fixed height so that workers have a comfortable arm reach to remove material from the moving belt. Conveyor belt widths can vary from 30 inches to approximately 72 inches in width, the maximum practical that allows a comfortable reach from either side. Workers are typically spaced about 6 to 8 feet apart. Located between each worker is a steel chute whose bottom is open through the platform floor.

Each worker is assigned to remove (i.e., pick) a specific material. As the conveyor moves material along, the workers pick their targeted material off the belt and drop it through the chute, where the material falls into a storage bunker. This is termed a *positive sort*. Any materials that are not picked from the conveyor are assumed to be of no recycling value or not practical to segregate and are allowed to flow off the end of the belt into an end bunker. This is termed the *negative sort* material, or alternately where the contaminant material is selected to be removed from the targeted material.

Periodically the bunkers will be unloaded and the materials taken away for final recycling/reuse or disposal. Bunkers can have

recycling/reuse or disposal. Bunkers can have mechanized unloading systems or be unloaded with a front-end loader.

Dust Control

The processing of C&D waste typically generates a significant amount of dust. Most of the materials in C&D are inherently dry, and when they are agitated in the various processing operations, dust particles (actually the minute pieces of some of the materials) are released into the air. Dust particles from a C&D operation can range in size from around 100 microns (concrete dust) up to 1,000 microns (textile dust).

C&D processing operations outdoors and inside a building can create a nuisance to neighbors with fugitive dust generation if it is not controlled properly. Constant -uncontrolled dust can also be a health -hazard to workers.

The primary sources of dust can include the following:

- Tipping floor
- Finger screen
- Air knife
- Storage piles (especially for residual "fine" materials)
- Recycled material load-out

Dust control is mostly science (and partly an art)

and too extensive a topic to cover completely here. However, fugitive dust is probably the most prevalent problem with C&D processing. Dust controls can take many forms, from basic static screening material to misting devices and mechanized filtering systems. The most common system for waste operations, the misting system, is discussed herein.

Wet systems-Dust is commonly controlled using wet spray systems where enclosing the material area is impractical. Wet systems can be applied for prevention of dust and suppression of dust. Prevention is done by applying a wet mist directly to the material to reduce liberation of dust. Suppression is applying a mist to the air around the material once it is agitated and dust is released. The type of misting system depends on the situation. Many C&D recycling operations may require both. Recycling facilities are particularly sensitive to the correct dust control. If too much water is applied, the materials may stick together and significantly reduce the effectiveness of downstream separation activities.

Effective dust control depends on many factors, including dust particle size, wind effects, freezing temperatures, nozzle types, nozzle spray patterns, spray locations, available water pressure, use of compressed air, and use of surface wetting agents (surfactants), to name a

surface wetting agents (surfactants), to name a few. The droplet size produced from the system must be compatible with the dust particle size, or removal is reduced. Dust control is best left to companies that are experts in the design and operation of these systems in waste handling operations.

Vibration-A brief discussion on vibration is in order. The type of waste processing systems described herein contain several pieces of heavy oscillating equipment, as well as a significant amount of rotating machinery in general. These systems produce steady-state (continuous) vibration, and impact (isolated) vibration. Steady-state vibration occurs during normal running operation. Impact vibration can occur when the equipment is started and stopped or a unusually heavy material is dropped into the machinery. The location of such a system should consider the potential for this equipment to transmit these vibrations to the ground, despite the vibration-dampening systems designed into most of the equipment. Vibration also results from waste-tipping operations and load-out of recyclable materials. A significant buffer zone is generally needed between the equipment and the property line to allow the ground vibration to dissipate to a background level.

Many municipalities have a standard for industrial zones that sets the maximum limit on

industrial zones that sets the maximum limit on steady-state and impact vibration. Without an adequate buffer width, vibration can be transmitted to adjacent properties where it can become a significant annoyance. The distance and strength over which vibration can be transmitted is contingent on many factors, including soil type, moisture content, frequency of vibration (i.e., number of oscillations per unit time), and others. It is recommended you consult the equipment manufacturer for guidance on a buffer distance. A buffer distance of 200 feet would be considered a good starting minimum with competent soil, a larger distance is preferred if available.

Recycling Byproducts

The processing of C&D waste with a certain combination of equipment can result in a residual by-product known as *recovered screen material* (RSM). RSM looks similar to soil and is mostly the combined residual of some actual soil and minute pieces of friable waste that may include, drywall, paint, plaster, asphalt shingle grit, pieces of grout, cement, brick, glass, and plastic.

Some recycling companies have experimented with developing this product for use as a substitute where lighter, load-bearing, natural soil fill is acceptable. For example, in shaping contours on a golf course, filling residential lots

contours on a golf course, filling residential lots up to flood criteria, and similar uses where natural soil is typically used. Although some regulatory agencies have approved its use in certain conditions after an extensive chemical testing program, and it has been marketed with some limited success, extra caution is warranted. Some RSM has been found to be contaminated with chemicals that, although they are native to the virgin materials, are of environmental concern because of their potential to leach out of the RSM. These chemicals include polynuclear aromatic hydrocarbons (PAHs), lead, and arsenic.

RSM is ground-up residual from many materials. As a result, the surface area of the individual particles is increased, which can then increase their solubility when wetted. The increased solubility may result in leaching of the chemical, which can result in contamination of the natural soil and groundwater. Common materials containing these contaminants include asphalt roofing shingles (PAHs), paint (lead), and treated wood (arsenic). Removal of the offending materials from the wastestream before they go through the separation process so they do not end up in the RSM is a time-consuming and expensive process that requires constant vigilance on the part of the operator. It is recommended to steer clear of RSM.

Recovering the Organics

Randy's Environmental Services of Delano, MN, had a challenge on its hands as it prepared to equip a MRF that would be accepting a wide variety of materials. The company has provided trash and recycling hauling services in the Twin Cities metropolitan area for over 30 years. Randy's is a recognized leader in recycling and source-separated organics collection. Randy's customers have expressed a growing interest in being able to recycle the organics fraction of their wastestreams, and by 2011 the company was ready to ensure it could provide this service.The Blue Bag Organics program was a key consideration in the overall design of the new MRF. In the Blue Bag program, customers are able to recycle foodwaste and yardwaste organics by putting the material into compostable blue bags that they can then place in their garbage bins. The MRF operator had to be able to pull those bags and limit the potential for the machinery to damage those bags prior to recovering them.

> Credit: Machinex
> Randy's Environmental Services had a challenge on its hands as it prepared to equip a MRF that would be accepting a wide variety of materials.

Jim Wallschlager and Mark Stoltman from

Jim Wollschlager and Mark Stoltman from Randy's Environmental Services conducted their research, and one vendor began to emerge as their favorite. "We toured several MRFs around the country and felt the Machinex installations were well laid out and were able to move the material without jams and bottlenecks," says Stoltman.

Randy's gave Machinex the challenge to team up with the company in designing a system that would meet its needs and allow for flexibility in the future. The system provider also worked in partnership with Randy's to design a system that allowed recovering 100% of those bags.

The Blue Bag Organics program was just one component of a wider stream that the Randy's MRF would have to handle.

The primary focus at Randy's was to be able to process both residential single-stream material as well as municipal solid waste (MSW) because of the Organics Blue Bag program.

A variety of screens would be necessary, although Machinex had an alternative technology that turned out to be a key component. Conventional screening technologies appeared more labor and maintenance intensive, specifically because of the film and bags. The Mach Ballistic answered

the client concerns and proved to be a very efficient means to separate the three fractions of fiber, containers, and fines. This versatile equipment ensures that different material types can be processed within a single system. Also, it has the advantage that the plastic film doesn't wrap around the disks.

Chris Hawn, Machinex North American sales manager, says the Machinex team took on several challenges to offer solutions for the MRF. Mach Ballistic was the right solution for several reasons. "No conventional system, whether for MSW or single-stream, was really efficient for this application, so Machinex designed a hybrid system that includes a trommel, two Mach Ballistic separators, and mechanical separation for containers."

The combination of sorting technologies creates a great system for efficient sorting of both types of material.

After operating and observing the system in place for nearly a year, Wollschlager agrees. "The system has worked well and proved to be very dependable," he states.

Customizing a MRF

MRF: Randy's Environmental Services, Delano, MN.

Capacity: 25 tons per hour of MSW; 12 tons per hour of residential single-stream recyclables

Equipment: Two Mach Ballistic separators, trommel screen, additional mechanical separation equipment

Process: In addition to MSW and single-stream recyclables, the MRF accepts organics in compostable blue bags that residents place in

Zero-Emission Off-Road Strategies

The Governor's Executive Order N-79-20[1] directs the California Air Resources Board (CARB) to develop and propose strategies to achieve 100% zero-emission from off-road vehicles and equipment operations in the State by 2035 where feasible. While CARB will work out details on how to achieve this goal during the public outreach process, many of CARB's existing programs and ongoing work focus on advancing and increasing adoption of zero-emission technologies in off-road applications. CARB, with our stakeholder partners, has developed a range of programs that focus on rapid adoption of the cleanest available commercial technologies to innovative demonstration projects supporting commercialization. These highly effective programs support early cost-effective actions that have consistently delivered air quality and climate benefits in an achievable timeframe, creating economic opportunities for individuals and businesses.

Incentives

Incentives are critical for supporting the advancement and wide-scale deployment of zero-emission technologies while simultaneously providing immediate emission reductions to help meet our air quality and climate goals. Traditional, monetary incentives from federal, state, and local sources may be used to demonstrate and assess feasibility of zero-emission technologies in various applications or to increase adoption of those technologies before required. Additionally, regulatory programs can provide strong incentives for businesses to develop or adopt zero-emission technologies in order to generate credits, which may be used for compliance or have monetary value (e.g., the Low Carbon Fuel Standard). Below are examples of traditional incentive programs that provide funding for zero-emission technologies:

- **Carl Moyer Program:** Once commercialized, the Carl Moyer Program increases the deployment of clean technologies for early fleet and equipment turnover by providing incentives for replacing existing vehicles and equipment with the cleanest available.
- **Volkswagen Environmental Mitigation Trust:** The Volkswagen Environmental Mitigation Trust provides funding for zero-emission freight and marine projects in California.
- **Funding Agricultural Replacement Measures for Emission Reductions (FARMER) Program:** The FARMER Program has opportunities for zero-emission demonstration projects in agricultural applications and provides funding for zero-emission equipment used in agriculture.
- **Low Carbon Transportation Program:** The Low Carbon Transportation Program focuses on advancing technologies through off-road zero-emission demonstration projects and supports early deployments of off-road zero-emission technologies.

Many of CARB's off-road demonstration projects have focused on advancing zero-emission technology in freight applications and continued support is necessary for commercialization. Additional demonstrations will be necessary to advance and assess the feasibility of zero-emission technology in other sectors, such as construction and agriculture. Once commercially available, incentives will play a critical role in increasing deployment of zero-emission technology in construction and farm equipment.

Regulations

Regulations can require manufacturers to develop and commercialize zero-emission technologies as well as increase or accelerate user adoption of those technologies. CARB sets certification standards through new engine emission control regulations, and the development of zero-emission certification standards will be critical for the widespread deployment of zero-emission technologies through

[1] Executive Order N-79-20: https://www.gov.ca.gov/wp-content/uploads/2020/09/9.23.20-EO-N-79-20-Climate.pdf

regulations and incentive programs. When developing new or amending existing regulations, CARB collaborates closely with stakeholders to ensure regulations are cost-effective, feasible, and provide the near-term reductions needed to meet federal air quality standards, while paving the way for long-term air quality and climate benefits. Below are some of the regulations that CARB currently has in place or under development to accelerate the deployment and adoption of zero-emission technologies in off-road applications:

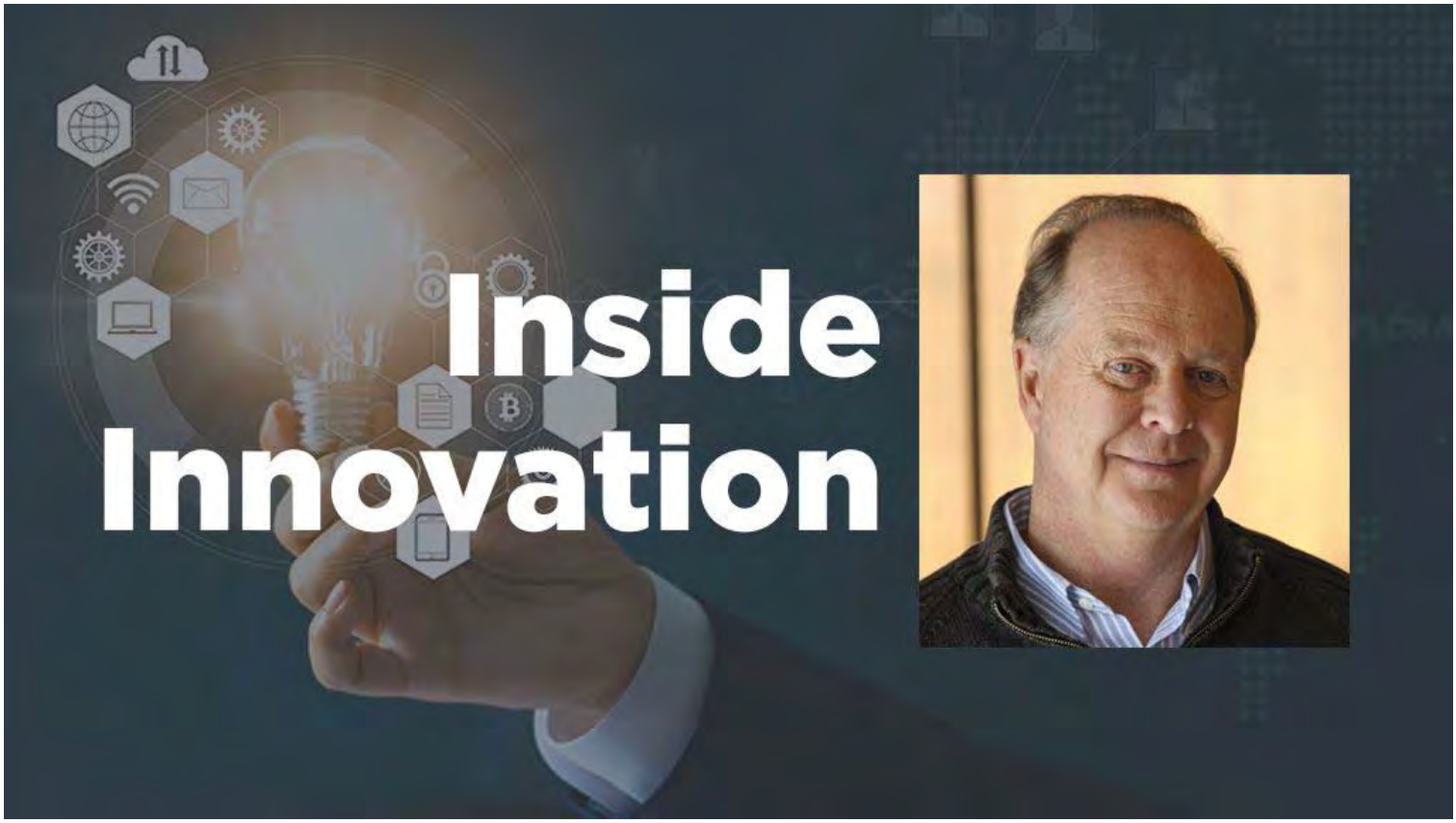
- **Vessels At Berth:** The At Berth regulation requires the control of exhaust emissions from vessels while at berth using shore power, a zero-emission technology, or other CARB approved control strategies. The At Berth regulation was recently amended to include additional vessel types, ports, terminals, and visits and provide additional oxides of nitrogen (NOx), particulate matter (PM), and greenhouse gas (GHG) emissions reductions.
- **Transport Refrigeration Units (TRU):** TRUs are refrigeration systems powered primarily by internal combustion engines and designed to refrigerate or heat perishable products transported in various containers. To reduce emissions from facilities with TRU activities, CARB is developed concepts to transition truck TRUs to zero-emissions, trailer TRUs to zero-emission operation, and usage of refrigerant with lower global warming potential. A TRU regulation is in development for Board consideration in 2021.
- **Commercial Harbor Craft:** Commercial harbor craft includes fishing vessels, ferries, excursion vessels, tug boats, tow boats, crew and supply boats, barges, dredges, and other vessel types. Amendments to existing rules will include zero-emission opportunities for all vessels while at dock, some ferries operating over shorter routes, and new excursion vessels, and are expected to go to the Board for consideration in mid-2021 with full implementation of zero-emission requirements by 2028.
- **Locomotives:** Locomotives are rail transport vehicles that provide the motive power for trains carrying both passengers and/or freight. In the absence of federal action, CARB staff are developing concepts to reduce criteria pollutants, toxic air contaminants, and GHG emissions for locomotives in-use, idling, and maintenance activities through the accelerated usage of cleaner locomotive engines, and zero-emission operations where feasible. CARB is in the process of developing a regulation for Board consideration in 2022.
- **Cargo Handling Equipment:** Cargo handling equipment is any motorized vehicle used to handle cargo or perform routine maintenance activities at California's ports and intermodal rail yards and includes yard trucks (hostlers), rubber-tired gantry cranes, container handlers, and forklifts. CARB is considering amendments to existing rules to include the transition to 100% zero-emission operations starting in 2026.
- **Zero-Emission Forklifts:** Forklifts are used in many different industrial sectors, but are most prevalent in manufacturing and at freight facilities, such as warehouse, distribution centers, and ports. CARB is in the process of developing a regulation for Board consideration in 2022 to increase zero-emission forklift deployment throughout the State.
- **In-Use Off-Road Diesel-Fueled Fleets (Off-Road) Regulation:** The Off-Road Regulation reduces NOx and PM emissions from diesel-fueled off-road fleets operating in California and zero-emission technology may be used to comply. Future amendments under consideration may ban older, high-emitting vehicles from fleets and include additional opportunities to encourage and incentivize zero-emission adoption where feasible.
- **Off-Road New Compression-Ignition Engines:** CARB staff currently plan to propose more stringent emission standards (i.e., Tier 5 standards) for off-road, land-based diesel engines around 2024. The Tier 5 standards would apply to engines used in farming, construction, and industrial applications in equipment like tractors, excavators, dozers, scrapers, portable generators, and

irrigation pumps; and they could be implemented as early as 2028 for non-federally preempt equipment. Staff plan to consider efficiency and zero-emission-transitional strategies within the Tier 5 standards and potentially in related rulemakings shortly thereafter.

TECHNOLOGY, US NEWS

Inside Innovation: Electrification makes worksites both ‘green’ and not heard

John Bleasby March 25, 2021



New electrified construction machinery has the obvious benefit of reducing GHGs and fuel exhaust emissions. Yet there is another important benefit: reduced noise.

The continuous high decibels on a jobsite created by fossil-fuel engines pose an ongoing threat to the health and safety of workers.

“Occupational hearing loss, primarily caused by high noise exposure, is the most common U.S. work-related illness…the third most common chronic physical condition in the United States,” say the Centers for Disease Control and Prevention (CDC).

The risks are recognized in Canada too. For example, Ontario’s Ministry of Labour, Training and Skills Development mandates that, “every employer shall ensure that no worker is exposed to a sound level greater than a time-weighted average exposure limit of 85dBA, measured over an eight-hour work day.”

WorkSafeBC has established various protocols for construction employers, including site signage, annual hearing tests, and “Record of Hearing Test” cards to show when hearing was last tested.

Noise also impacts those in adjacent buildings, particularly in today’s tight urban settings. Builders can be expected to demonstrate increased social responsibility by making efforts to mitigate noise as much as possible on their projects.

Combined with the reduction in GHGs and exhaust, electrification appears to be a way to make worksites both “green” and not heard. That’s because electrically-powered equipment and vehicles are, of course, far quieter than fossil-fuelled equivalents.

Aside from vehicles and earth-moving equipment, gas or diesel-powered generators are often overlooked as a source of noise and GHGs on worksites. Noise levels from portable 50kW generators might be around 85dBA, but larger units can run over 100dBA. Typically, generators run all day powering all kinds of smaller equipment. They can also operate through the night running security lighting around the site, emitting noise and exhaust constantly.

Taken together, the issues of emissions and noise should encourage operators to look at alternatives, like the onsite energy stations developed by Portable Electric based in Vancouver B.C.

The company’s range of VOLTstack power stations are, of course, nearly silent and emit no exhausts of any sort. In fact, their range of units from 5kW to 20kW can be both solar-powered or plugged in for recharging. Units are currently sold in several countries around the world and are widely accepted in the entertainment and movie industries.

However, VOLTstack power stations are more expensive to purchase or rent than fossil-fuelled generators.

Given the construction industry’s longstanding resistance to new methods and sensitivity to costs, some hesitancy is to be expected. Cost premiums need to be understood and justified, both in terms of ongoing operation and long-term investment.

To address the issues of operational efficiency and costs, and to quantify the environmental benefits, Portable Electric undertook a sustainability demonstration during a construction project in Aldergrove, B.C. in 2020.

One part of the study involved replacing two portable gas-powered generators with two 5 kW VOLTstack units. In another part, a 20kW VOLTstack unit was “hybridized” with an onsite diesel generator. The results were dramatic.

Over the course of two weeks, the two 5kW VOLTstack units alone eliminated 303kg of C02 emissions. In fact, during one week, fossil fuel costs were totally eliminated through the use of PV panels.

Meanwhile, it was found that the VOLTstack 20kW was able to support the load of multiple office trailers for a full five-day working week. When necessary, the VOLTstack 20kW was easily recharged using the onsite diesel generator. Diesel generator runtime was thereby reduced by as much as 93 per cent, reducing fuel costs and CO2 emissions by as much as 76 per cent. Longer term, the reduced maintenance costs for the VOLTstack power stations versus those for typical gas/diesel generators is another factor to be considered.

The construction industry needs to maintain an open and inquisitive mind if it is to meet the noise and emission reductions expected by 2030 demanded by new regulations and government policies. Going electric onsite is one means to that end.

John Bleasby is a Coldwater, Ont.-based freelance writer. Send comments and Inside Innovation column ideas to editor@dailycommercialnews.com

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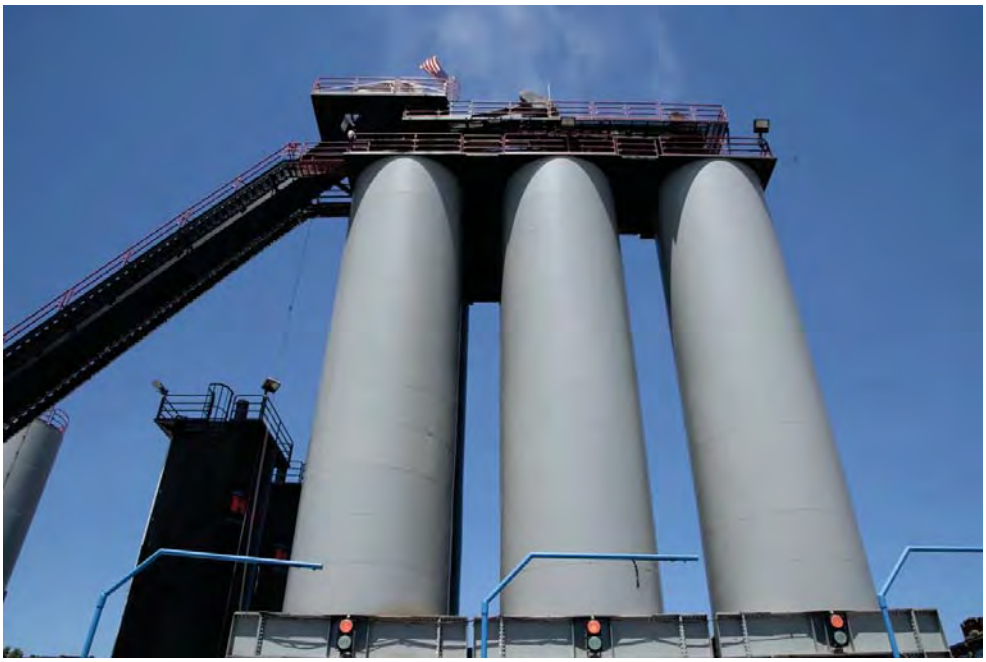
ENVIRONMENT NEWS POLITICS

Feds jump into McKinley Park asphalt plant fracas

Senators Durbin and Duckworth asked President Biden's EPA to oversee air pollution testing and oversight of MAT Asphalt, which prompted more than 100 odor complaints from neighbors in three years.

By Brett Chase | Jul 28, 2021, 6:00pm CDT



Neighbors have filed more than 100 complaints about odor from MAT Asphalt over the past three years. | Brian Rich/Sun-Times

At the request of Illinois' two U.S. senators, federal officials are turning up the heat on a controversial asphalt plant in McKinley Park.

The U.S. Environmental Protection Agency said it is working to make sure the state sets strict rules within an operating permit for the MAT Asphalt operation on Pershing Road across from McKinley Park, according to a letter sent to Sen. Dick Durbin from an EPA official in Chicago last week.

The letter from acting EPA regional administrator Cheryl Newton also notes that Latino-majority McKinley Park is a "community with environmental justice concerns," a reference to the burden of cumulative pollution in a low-income community of color. Under President Joe Biden, the EPA is stressing the need to reduce pollution in environmental justice communities. In May, Mayor Lori Lightfoot halted the permit process for the relocation of the General Iron car-shredding operation to the city's Southeast Side at the request of Biden's top environmental chief — an action that community organizers had been pleading with the mayor to do for many months.

"EPA is working closely with the state agency to ensure that this facility's operating permit includes all applicable requirements and that public concerns — including concerns about health impacts and environmental justice — are adequately evaluated and addressed," Newton wrote in her July 20 letter to Durbin.

RELATED

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City explores moving controversial McKinley Park asphalt plant

MAT has been operating under a state-issued construction permit since 2017 but will eventually need to obtain a full-time operating permit, a process that has been slowed in part due to the COVID-19 pandemic.

Durbin, Sen. Tammy Duckworth and Reps. Jesus “Chuy” Garcia and Marie Newman, all Democrats, sent a letter to Biden’s EPA Administrator Michael Regan in late June, asking the agency to help provide oversight of the asphalt plant, including ordering “rigorous health impact analyses and air monitoring.” They note more than 100 odor complaints to the city from residents in the past three years.

Michael Tadin Jr., a city contractor and co-owner of the asphalt facility, has said the odor complaints are exaggerated and touts that he provides an essential service as an asphalt producer.

The legislators noted the plant is directly adjacent to two schools and across from a park in an area that “was already overburdened by pollution.” EPA data show McKinley Park, also home to a large Asian population, ranks high for the amount of cumulative air pollution and health risks.

“This cumulative pollution burden damages the health of residents and impacts their ability to safely recreate and enjoy their homes and community spaces,” the lawmakers wrote in their June 21 letter. “The operation of additional industrial facilities in this neighborhood, such as MAT Asphalt, places further stress on this overburdened community.”

In her letter to Durbin, Newton said the EPA conducted an inspection at MAT in 2019 and asked state officials to conduct air emissions tests “based on the concerns EPA staff found.” Those tests were supposed to be completed last year but the Pritzker Administration postponed them, citing the pandemic. The testing took place this week.

Tadin said he welcomes the EPA testing requirements.

“The agency asked us to do testing and we obliged,” Tadin said.

MAT Asphalt co-owner and city contractor Michael Tadin Jr. said he welcomes the air testing of his facility, which he says provides an essential service. | Brian Rich/Sun-Times

City Hall's mixed messages

City Hall has been split on MAT. While city planners have lauded the plant's central location for providing asphalt, a top adviser, Lightfoot's housing commissioner Marisa Novara, held up a plan last year to put low-income housing next to the asphalt maker, which is also close to schools and the neighborhood's namesake park. In May, the City Council approved the housing project, which found alternative funding. Separately, federal housing officials are investigating a civil rights complaint related to the Southeast Side car-shredder fight.

Novara also made public last year that the city was exploring whether MAT could be moved.

Late last year, a consultant for MAT provided the city with a plan to move just hundreds of feet at a cost of almost $32 million, according to the report obtained through the state's open records law.

City planners "determined there is no public funding available to help pay the costs identified," a city spokesman said. "Staff also determined there are no known alternative locations where the plant could lawfully operate in the city without generating similar community concerns."

Brett Chase's reporting on the environment and public health is made possible by a grant from The Chicago Community Trust.

Ozinga Ready Mix, Inc.

EPA reached an agreement with Ozinga Ready Mix, Inc., in June 2016 to reduce particulate matter or dust from its operations in the Chicago area and to buy new, cleaner diesel trucks. Ozinga will also pay a penalty of $37,689 for alleged violations of the Clean Air Act.
Ozinga allegedly failed to control dust to required levels at facilities at 1818 East 103rd Street as well as 2255 South Lumber Street in Chicago. EPA is requiring Ozinga to update dust control programs at its facilities.

Under the agreement with EPA, Ozinga will also purchase 30 new diesel trucks to replace older ones at its Chicago facilities at an estimated cost of $5 million. The diesel truck replacement project will reduce emissions of carbon dioxide, particulate matter, volatile organic compounds, nitrogen oxides and carbon monoxide, making the air cleaner in Southeast Chicago.

Contact Information

Nathan Frank
(frank.nathan@epa.gov)
312-886-3850 or 800-621-8431, ext. 63850

Environmental Issues in Southeast Chicago

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REGION 5
77 WEST JACKSON BOULEVARD
CHICAGO, IL 60604-3590

APR 20 2018

REPLY TO THE ATTENTION OF:

CERTIFIED MAIL
RETURN RECEIPT REQUESTED

Austin Zenere
Chicago Rail and Port, LLC
3245 East 103rd Street
Chicago, Illinois 60617

Re: Notice of Violation
Chicago Rail and Port, LLC

Dear Mr. Zenere:

The U.S. Environmental Protection Agency is issuing the enclosed Notice of Violation (NOV) to Chicago Rail and Port, LLC (you) under Section 113(a)(1) of the Clean Air Act, 42 U.S.C. § 7413(a)(1). EPA finds that you are violating the Illinois State Implementation Plan at your Chicago, Illinois facility.

Section 113 of the Clean Air Act gives us several enforcement options. These options include issuing an administrative compliance order, issuing an administrative penalty order and bringing a judicial civil or criminal action.

We are offering you an opportunity to confer with us about the violations alleged in the NOV. The conference will give you an opportunity to present information on the specific findings of violation, any efforts you have taken to comply and the steps you will take to prevent future violations. In addition, in order to make the conference more productive, we encourage you to submit to us information responsive to the NOV prior to the conference date.

Please plan for your facility's technical and management personnel to attend the conference to discuss compliance measures and commitments. You may have an attorney represent you at this conference.

The EPA contact in this matter is Molly Smith, (312) 353-8773, or Patrick Miller, (312) 886-4044. You may call at either to request a conference. You should make the request within 10 calendar days following receipt of this letter. We should hold any conference within 30 calendar days following receipt of this letter.

Sincerely,

Edward Nam
Director
Air and Radiation Division

Enclosure

cc: Julie Armitage, Chief, Bureau of Air, Illinois Environmental Protection Agency, Julie.Armitage@Illinois.gov

UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
REGION 5

IN THE MATTER OF:	)	
	)	
Chicago Rail and Port, LLC	)	**NOTICE OF VIOLATION**
Chicago, Illinois	)	
	)	**EPA-5-18-IL-10**
Proceedings Pursuant to	)	
Section 113(a)(1) of the	)	
Clean Air Act, 42 U.S.C.	)	
§ 7413(a)(1)	)	

NOTICE OF VIOLATION

The U.S. Environmental Protection Agency is issuing this Notice of Violation under Section 113(a)(1) of the Clean Air Act (CAA), 42 U.S.C. § 7413(a)(1). EPA finds that Chicago Rail and Port, LLC (CRP) is violating the Illinois State Implementation Plan (SIP), as follows:

Statutory and Regulatory Background

1. Pursuant to Sections 108 and 109 of the CAA, 42 U.S.C. §§ 7408 and 7409, EPA reaffirmed the National Ambient Air Quality Standards (NAAQS) for particulate matter equal to or less than 10 microns in diameter (PM_{10}) on October 17, 2006. 71 *Fed. Reg.* 61224 (2006).

2. The revised national primary and secondary ambient air quality standard for PM_{10} is 150 micrograms per cubic meter ($\mu g/m^3$), 24-hour average concentration.

3. Appendix K to 40 C.F.R. Part 50 explains the computations necessary for analyzing particulate matter data to determine attainment of the 24-hour standards specified in 40 C.F.R. § 50.6.

Illinois SIP

4. On May 31, 1972, EPA approved Illinois Pollution Control Board (IPCB) Rules 101 and 102 as part of the federally enforceable SIP for the State of Illinois. 37 *Fed. Reg.* 10842. IPCB Rule 101 has been recodified at 35 Illinois Administrative Code (Ill. Admin. Code) § 201.102. IPCB Rule 102 has been recodified at 35 Ill. Admin. Code § 201.141.

5. On February 21, 1980, EPA approved the IPCB Rule 203(f)(1) as part of the federally enforceable SIP for the State of Illinois. 45 *Fed. Reg.* 11493 (February 21, 1980). IPCB Rule 203(f)(1) has been recodified at 35. Ill. Admin. Code § 212.301.

6. The Illinois SIP at 35 Ill. Admin. Code § 201.141 provides, in pertinent part, that no person shall cause or threaten or allow the discharge or emission of any contaminant into the environment in any State so as, either alone or in combination with contaminants from

other sources, to cause or tend to cause air pollution in Illinois or so as to prevent the attainment or maintenance of any applicable ambient air quality standard.

7. The Illinois SIP at 35 Ill. Admin. Code § 201.102 defines "Ambient Air Quality Standard" as those standards promulgated from time to time by the IPCB pursuant to authority contained in the Illinois Environmental Protection Act and found at 35 Ill. Adm. Code § 243, or by the EPA pursuant to authority contained in 42 U.S.C. § 7401 et seq. as amended.

8. The Illinois SIP at 35 Ill. Admin. Code § 201.102 defines "Air Pollution" as the presence in the atmosphere of one or more air contaminants in sufficient quantities, characteristics and duration so as to be injurious to human, plant, or animal life, to health, or to property, or to unreasonably interfere with the enjoyment of life or property.

9. The Illinois SIP at 35 Ill. Admin. Code § 243.120 incorporated the 24-hour NAAQS for PM_{10} as 150 $\mu g/m^3$, 24-hour average concentration. The primary and secondary NAAQS for PM_{10} are attained when the expected number of days per calendar year with a 24-hour average concentration above 150 $\mu g/m^3$ is equal to or less than one.

10. The Illinois SIP at 35 Ill. Admin. Code § 243.102 defines "PM_{10}" as particulate matter that has an aerodynamic diameter less than or equal to a nominal 10 micrometers (μm).

11. The Illinois SIP at 35 Ill. Admin. Code § 243.120 incorporates by reference Appendix K to 40 C.F.R. Part 50 (2013) (Interpretation of the Primary and Secondary National Ambient Air Quality Standards for Particulate Matter).

Factual Background

12. CRP owns and operates a bulk solid storage and handling facility 3245 East 103rd Street, Chicago, Illinois. The facility operates storage piles, roadways, material handling operations, and loading and unloading operations that emit particulate matter, including PM_{10}.

13. On March 1, 2017, S.H. Bell Company, Chicago, Illinois (S.H. Bell), began operating a meteorological station and a PM_{10} ambient air monitoring network at 10218 South Avenue O, Chicago, Illinois (Monitoring Network). The Monitoring Network includes four continuous Federal Equivalence Method (FEM) PM_{10} monitors and one filter-based Federal Reference Method (FRM) monitor. The monitors are located at the following locations:

    a. FEM, S1, 41.708239, -87.544058;
    b. FEM, S2, 41.710553, -87.539204;
    c. FEM, S3, 41.710552, -87.542043;
    d. FEM and FRM, S4, 41.711541, -87.539607; and
    e. Meteorological station, 41.709841, -873540376.

14. On January 16, 2018, S.H. Bell submitted meteorological data and Monitoring Network data for the month of December 2017 from the four FEM monitors. The data submissions included the southernmost monitor, S1.

15. The December 2017 Monitoring Network data showed that on December 4, 2017, the 24-hour PM_{10} average concentration was 179 $\mu g/m^3$ at monitor S1.

16. The December 4, 2017, Monitoring Network meteorological data shows the wind was from the south and averaged over 15 miles per hour (mph).

17. EPA inspected the CRP facility on February 1, 2018, and confirmed bulk solid limestone storage piles in the northern portion of the CRP site, near S1 of the Monitoring Network.

18. During the February 1, 2018, inspection, CRP provided copies of the Reference Method 22 Daily Observation Checklist. The checklist for December 4, 2017, indicated 25 loads of limestone were shipped during wind gusts of 20 – 38 mph.

Violations

19. CRP caused the emission of PM_{10} into the air, so as, either alone or in combination with contaminants from other sources, to cause or tend to cause, air pollution in Illinois and/or to prevent the maintenance of the revised NAAQS for PM_{10} in violation of the Illinois SIP at 35 Ill. Admin. Code § 201.141

Environmental Impact of Violations

20. These violations have caused or can cause excess emissions of particulate matter: Particulate matter, especially fine particulates contains microscopic solids or liquid droplets, which can get deep into the lungs and cause serious health problems. Particulate matter exposure contributes to:
    - irritation of the airways, coughing, and difficulty breathing;
    - decreased lung function;
    - aggravated asthma;
    - chronic bronchitis;
    - irregular heartbeat;
    - nonfatal heart attacks; and
    - premature death in people with heart or lung disease.

4/20/18
Date

Edward Nam
Director
Air and Radiation Division

EPA United States Environmental Protection Agency

Office of Enforcement and Compliance Assurance
EPA-300-B-17-001 June 2017

U.S. EPA Small Business Resources Information Sheet

The United States Environmental Protection Agency provides an array of resources to help small businesses understand and comply with federal and state environmental laws. In addition to helping small businesses understand their environmental obligations and improve compliance, these resources will also help such businesses find cost-effective ways to comply through pollution prevention techniques and innovative technologies.

Office of Small and Disadvantaged Business Utilization (OSDBU)
www.epa.gov/aboutepa/about-office-small-and-disadvantaged-business-utilization-osdbu

EPA's OSBBU advocates and advances business, regulatory, and environmental compliance concerns of small and socio-economically disadvantaged businesses.

EPA's Asbestos Small Business Ombudsman (ASBO)
www.epa.gov/resources-small-businesses/asbestos-small-business-ombudsman or 1-800-368-5888

The EPA ASBO serves as a conduit for small businesses to access EPA and facilitates communications between the small business community and the Agency.

Small Business Environmental Assistance Program
https://nationalsbeap.org

This program provides a "one-stop shop" for small businesses and assistance providers seeking information on a wide range of environmental topics and state-specific environmental compliance assistance resources.

EPA's Compliance Assistance Homepage
www.epa.gov/compliance

This page is a gateway to industry and statute-specific environmental resources, from extensive web-based information to hotlines and compliance assistance specialists.

Compliance Assistance Centers
www.complianceassistance.net

EPA sponsored Compliance Assistance Centers provide information targeted to industries with many small businesses. They were developed in partnership with industry, universities and other federal and state agencies.

Agriculture
www.epa.gov/agriculture

Automotive Recycling
www.ecarcenter.org

Automotive Service and Repair
www.ccar-greenlink.org or 1-888-GRN-LINK

Chemical Manufacturing
www.chemalliance.org

Construction
www.cicacenter.org

Education
www.campuserc.org

Food Processing
www.fpeac.org

Healthcare
www.hercenter.org

Local Government
www.lgean.org

Surface Finishing
http://www.sterc.org

Paints and Coatings
www.paintcenter.org

Printing
www.pneac.org

Ports
www.portcompliance.org

Transportation
www.tercenter.org

U.S. Border Compliance and Import/Export Issues
www.bordercenter.org

EPA Hotlines and Clearinghouses
www.epa.gov/home/epa-hotlines

EPA sponsors many free hotlines and clearinghouses that provide convenient assistance regarding environmental requirements. Examples include:

Clean Air Technology Center (CATC) Info-line
www.epa.gov/catc or 1-919-541-0800

Superfund, TRI, EPCRA, RMP, and Oil Information Center
1-800-424-9346

EPA Imported Vehicles and Engines Public Helpline
www.epa.gov/otaq/imports or 1-734-214-4100

National Pesticide Information Center
www.npic.orst.edu or 1-800-858-7378

National Response Center Hotline to report oil and hazardous substance spills - http://nrc.uscg.mil or 1-800-424-8802

Pollution Prevention Information Clearinghouse (PPIC) - www.epa.gov/p2/pollution-prevention-resources#ppic or 1-202-566-0799

Safe Drinking Water Hotline - www.epa.gov/ground-water-and-drinking-water/safe-drinking-water-hotline or 1-800-426-4791

Toxic Substances Control Act (TSCA) Hotline
tsca-hotline@epa.gov or 1-202-554-1404

U.S. Small Business Resources

Small Entity Compliance Guides
https://www.epa.gov/reg-flex/small-entity-compliance-guides

EPA publishes a Small Entity Compliance Guide (SECG) for every rule for which the Agency has prepared a final regulatory flexibility analysis, in accordance with Section 604 of the Regulatory Flexibility Act (RFA).

Regional Small Business Liaisons
www.epa.gov/resources-small-businesses/epa-regional-office-small-business-liaisons

The U.S. Environmental Protection Agency (EPA) Regional Small Business Liaison (RSBL) is the primary regional contact and often the expert on small business assistance, advocacy, and outreach. The RSBL is the regional voice for the EPA Asbestos and Small Business Ombudsman (ASBO).

State Resource Locators
www.envcap.org/statetools

The Locators provide state-specific contacts, regulations and resources covering the major environmental laws.

State Small Business Environmental Assistance Programs (SBEAPs)
https://nationalsbeap.org/states/list

State SBEAPs help small businesses and assistance providers understand environmental requirements and sustainable business practices through workshops, trainings and site visits.

EPA's Tribal Portal
www.epa.gov/tribalportal

The Portal helps users locate tribal-related information within EPA and other federal agencies.

EPA Compliance Incentives
EPA provides incentives for environmental compliance. By participating in compliance assistance programs or voluntarily disclosing and promptly correcting violations before an enforcement action has been initiated, businesses may be eligible for penalty waivers or reductions. EPA has two such policies that may apply to small businesses:

EPA's Small Business Compliance Policy
www.epa.gov/enforcement/small-businesses-and-enforcement

EPA's Audit Policy
www.epa.gov/compliance/epas-audit-policy

Commenting on Federal Enforcement Actions and Compliance Activities

The Small Business Regulatory Enforcement Fairness Act (SBREFA) established a SBREFA Ombudsman and 10 Regional Fairness Boards to receive comments from small businesses about federal agency enforcement actions. If you believe that you fall within the Small Business Administration's definition of a small business (based on your North American Industry Classification System designation, number of employees or annual receipts, as defined at 13 C.F.R. 121.201; in most cases, this means a business with 500 or fewer employees), and wish to comment on federal enforcement and compliance activities, call the SBREFA Ombudsman's toll-free number at 1-888-REG-FAIR (1-888-734-3247).

Every small business that is the subject of an enforcement or compliance action is entitled to comment on the Agency's actions without fear of retaliation. EPA employees are prohibited from using enforcement or any other means of retaliation against any member of the regulated community in response to comments made under SBREFA.

Your Duty to Comply

If you receive compliance assistance or submit a comment to the SBREFA Ombudsman or Regional Fairness Boards, you still have the duty to comply with the law, including providing timely responses to EPA information requests, administrative or civil complaints, other enforcement actions or communications. The assistance information and comment processes do not give you any new rights or defenses in any enforcement action. These processes also do not affect EPA's obligation to protect public health or the environment under any of the environmental statutes it enforces, including the right to take emergency remedial or emergency response actions when appropriate. Those decisions will be based on the facts in each situation. The SBREFA Ombudsman and Fairness Boards do not participate in resolving EPA's enforcement actions. Also, remember that to preserve your rights, you need to comply with all rules governing the enforcement process.

EPA is disseminating this information to you without making a determination that your business or organization is a small business as defined by Section 222 of the Small Business Regulatory Enforcement Fairness Act or related provisions.

CERTIFICATE OF MAILING

I certify that I sent a Notice of Violation, No. EPA-5-18-IL-10 by Certified Mail, Return Receipt Requested, to:

Austin Zenere
Chicago Rail and Port, LLC
3245 East 103rd Street
Chicago, Illinois 60617

I also certify that I sent copies of the Notice of Violation to:

Julie Armitage
Chief
Bureau of Air
Illinois Environmental Protection
Agency
1021 North Grand Avenue East
Springfield, Illinois 62794
Julie.Armitage@Illinois.gov

Mort Ames
City of Chicago Law Department
Assistant Corporation Counsel Supervisor
30 N. La Salle Street, Suite 1400
Chicago, Illinois 60602
Mort.Ames@cityofchicago.org

Dave Graham
Chicago Department of Public Health
Assistant Commissioner
333 S. State Street, Room 200
Chicago, Illinois 60604
Dave.Graham@cityofchicago.org

On the 23rd day of April 2018.

Kathy Jones

Kathy Jones
Program Technician
AECAB, PAS

CERTIFIED MAIL RECEIPT NUMBER: 7009 1680 0000 7641 3428

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THE INVERT

FAQ

HAS THIS BEEN DONE BEFORE?

Yes! Our favorite example is the Subtropolis development in Kansas City which houses more than 50 businesses with over 90% occupancy rate. In fact, the Kansas City market has about 30 million SF of total subsurface space with 10 million more under development. Subsurface facilities can be found throughout the world and our construction methodology will be very similar to Chicagoland facilities such as CTA stations, the Pedway, and underground parking garages. You can learn more about Subtropolis by watching episode 4 of the documentary series “Off the Cuff” available on Amazon Prime or by visiting their website at https://huntmidwest.com/industrial-space-for-lease/.

WHO OWNS THIS PROJECT?

The Invert Chicago, LLC is a privately held real estate development company. The lead investor is Ozinga Ventures, an independent investment group established by the Ozinga family to support innovation and entrepreneurship that makes a positive impact on individuals, their families, and the community for generations. However, The Invert Chicago is not affiliated with Ozinga’s traditional construction materials business.

DO YOU REALLY JUST WANT TO SELL THE MATERIAL THAT WOULD BE REMOVED FROM THE SITE?

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subsurface construction. However, subsurface construction techniques do allow us to work around the environmental challenges that otherwise prevent the property from being developed. Subsurface construction is a proven technique used in Subtropolis in Kansas City and throughout Chicago in the form of the Pedway, CTA subways, underground parking garages and more.

HOW MUCH ADDITIONAL TRAFFIC WILL THIS BRING INTO THE COMMUNITY?

CMAP's Go To 2040 plan estimates an increase in traffic in the area. The type of tenants The Invert will target and attract and has already received interest from typically require far fewer trucks than those currently anticipated by planners. Our current estimates point to an additional 3200 vehicle trips per day starting in 2040 which is about 20% less than CMAP's projections without The Invert. Furthermore, our current estimates project that 80% of the expected traffic increase will be from passenger vehicles.

The Invert also would move trucks away from residences on Avenue O by extending Burley Ave. The Invert will encourage the adoption of electric vehicles by supporting electric vehicle charging infrastructure. It is also our plan to minimize vehicle emissions through the strategic placement of hundreds of trees, stringent vehicle idling policies, and the use of other innovative technologies.

HOW MANY JOBS WILL THIS BRING HERE? WILL THEY BE FILLED BY LOCAL PEOPLE?

The property currently generates just eight jobs. We expect over 2,500 permanent jobs will be created once both levels are fully leased. This dramatic

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ISN'T THIS GOING TO TAKE A LONG TIME TO BUILD?

Our current timeline expects the first block of leasable space to be available within 3-5 years of start of construction , followed by ongoing phased openings of leasable space, and completion between years 12-15, subject to market conditions. This is typical of large projects. For reference, the highly successful Pullman developments have seen timelines of 10+ years for 1.75M SF of development compared to our expected 15 years for 6M SF of space.

ISN'T THE CONTAMINATED PROPERTY HARMING OR GOING TO HARM THE COMMUNITY IF YOU BUILD ON IT?

The Invert is designed to not disturb any environmental hotspots on the surface by developing underneath it. The Phase I and II environmental survey assessments revealed areas with significant contamination buried up to 30 feet below the surface, which is currently not a harm to the local community since it is buried in gravel, slag and other crushed material. The Federal EPA has determined there is no further surface remediation needed since the historical contaminated materials are sufficiently contained. Our proposed development will not disturb contaminated areas monitored by the EPA. For example, the vertical shafts are located away from these hotspots, and our dual tunnel system is located on the east side of the property where the EPA has determined no environmental threats.

WHAT CONSTRUCTION TECHNIQUES WILL BE USED? WHAT WILL I SEE, HEAR, AND FEEL?

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below the surface, electronically-controlled demolition will be utilized. This is a carefully controlled procedure that will not be seen or heard from the surface. Because we're very deep in the ground, the greatest vibration experienced on the surface will be negligible.

WHAT IF I HAVE A QUESTION OR AN IDEA FOR THE PROJECT?

We strongly value community engagement and participation in the project. You can learn more online at www.TheInvertChicago.com. The Invert also operates a Community Engagement Center at 10548 S. Ewing Avenue where we invite local residents, businesses, nonprofits and community groups to come learn about the project and provide feedback. We also host virtual meetings so that all can safely attend. To request a meeting, you can email us at info@TheInvertChicago.com

WHAT WILL HAPPEN TO THE SOIL THAT IS EXCAVATED FROM THE SITE, SINCE SO MUCH IS CONTAMINATED?

Our plan is not to disturb the contaminated portions of the property by developing around and beneath them to create uniquely marketable subsurface space. This avoids additional health risks as well as the need for special excavation, transportation, and landfill costs. However, we are interested in pursuing emerging technologies or partnerships that will allow for more cost-efficient remediation. We are also considering alternative best uses for the contaminated part of the surface such as an even larger solar panel field.

WHY SHOULD I SUPPORT THE INVERT? WHAT'S IN IT FOR ME AND MY COMMUNITY?

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we are committed to providing them in the final design. Examples of the ideas we have heard thus far include: additional community gathering spaces, recreational space, local partnerships, community reinvestment, and more!