Title
Effects of Diesel Emission Control Measures and Truck Routing on Air Quality, Environmental Equity and Justice

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Effects of Diesel Emission Control Measures and Truck Routing on Air Quality, Environmental Equity and Justice

by

Regan F. Patterson

A dissertation submitted in partial satisfaction of the

requirements for the degree of

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in

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of the

University of California, Berkeley

Committee in charge:

Professor Robert A. Harley, Chair
Professor Ashok Gadgil
Professor Rachel Morello-Frosch

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Abstract

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In goods movement corridors, some freeways and local arterial roadways are heavily trafficked by heavy-duty diesel trucks. Heavy-duty diesel trucks are major sources of air pollution. Localized impacts of diesel exhaust have environmental justice implications for communities near port and goods movement truck activity. These communities are disproportionately burdened by exposures and associated health risks. Predominately nonwhite and low-income communities are concentrated in goods movement corridors due to sociospatial processes, such as residential segregation. Policies that mitigate heavy-duty diesel truck exposures have the potential to address disproportionate exposure burdens. Two recently implemented policies that reduce exposures are the use of diesel truck emission controls and truck rerouting. In this dissertation, I examine the impact of these policies on near-roadway concentrations of diesel-related air pollution. This dissertation provides new policy-relevant insights by quantifying the environmental justice and equity outcomes of air pollution mitigation strategies for heavy-duty diesel trucks. This analysis is significant because it illustrates a systematic approach for incorporating quantitative measures of inequality and injustice within analyses of diesel truck-related policies. This formal process can be used to inform the future design and implementation of policies and initiatives, particularly in highly impacted areas.

The pollutant dispersion model RLINE was used to predict near-roadway concentrations of nitrogen oxides (NO_x) and black carbon (BC). Model predictions were compared with continuous, yearlong records of measurements from two near-roadway sites in the San Francisco Bay Area. Heavy-duty diesel trucks were a significant source of NO_x and BC at both sites. Characterization of temporal variations in heavy-duty truck activity on diurnal, weekly, and seasonal scales was included in this analysis; truck traffic and emissions are not well-correlated with passenger vehicle or total traffic volumes. For both pollutants, more than 90% of predicted 24-hour average concentrations were within a factor of two of observations at both near-roadway monitoring sites. The model responds appropriately to seasonal variations in meteorology and day-of-week variations in emissions. Model performance for NO_x was better overall than for BC. Reducing uncertainties in emission factors would help to improve model performance for BC.

The air quality and environmental equity benefits that result from accelerated use of diesel particle filter (DPF) and selective catalytic reduction (SCR) systems on heavy-duty diesel trucks are assessed in this dissertation. My research focuses on communities in Oakland, California, adjacent to two major freeways: Brookfield Village and Sobrante Park along I-880, which are heavily affected by truck traffic; and Sequoyah along I-580 where heavy-duty trucks are prohibited. Brookfield-Sobrante has a higher proportion of nonwhite, low-income residents than Sequoyah (97% versus 76%). I modeled concentrations of nitrogen oxides (NO_x), diesel particulate matter (PM), and black carbon (BC) prior to widespread use of diesel emission controls (2009), after universal adoption of DPFs to control PM emissions (2018), and after universal adoption of SCR to control NO_x emissions (2023). Reductions in mean near-roadway pollutant concentrations in Brookfield-Sobrante were 63±3% for NO_x and 48-49±4% for diesel PM and BC by 2018. In Sequoyah, reductions in mean NO_x concentrations were smaller (52±3%), and mean diesel PM and BC concentrations increased by 19±7% and 15±6%, respectively. While estimated NO_x concentrations remain higher in Brookfield-Sobrante compared to Sequoyah, diesel PM and BC concentrations will be similar in both neighborhoods by 2023. Reductions in diesel emissions also led to improvements in equity when quantified by the difference in mean intakes for Brookfield-Sobrante versus Sequoyah. Exposure inequities decreased more for diesel PM and BC than NO_x. Maintaining these air quality and equity benefits requires that diesel emission control systems remain in good working order over time.

The effects of freeway routing decisions on exposure to diesel-related air pollution and neighborhood socioeconomic and demographic change are also investigated. Freeway rerouting and replacement with a street-level boulevard are urban transportation policies that may help redress disproportionate air pollution burdens resulting from freeway construction that took place during the mid-20th century. However, environmental justice activism for freeway rerouting may have the unintended consequence of environmental gentrification. I focus on the effects of rerouting the Cypress Freeway in West Oakland along with construction of a street-level boulevard (Mandela Parkway) on the original freeway alignment. The impacts of two rebuild scenarios, freeway rebuild-in-place and reroute, on near-roadway NO_x and BC concentrations are compared. I also assess changes in demographics and land use in West Oakland between the time when the Cypress Freeway was damaged by a major earthquake and after completion of Mandela Parkway. My research indicates that freeway rerouting reduced annual average concentrations of both NO_x (–38±4%) and BC (–25±2%) along the Mandela Parkway alignment. However, there is evidence of environmentally-driven neighborhood change, given that there are larger decreases in the long-time Black population (–37%) and increases in property values (229%) along the residential portion of the Mandela Parkway corridor compared to West Oakland as a whole. There are some attributes along the Mandela Parkway that enable low-income residents to live in proximity to the street-level boulevard, such as affordable housing.

Recommendations for future research include developing longitudinal community-based participatory traffic count surveys, evaluating model predictions of air pollution using new local-level monitoring techniques, and assessing relationships between diesel policy-related changes in air pollution and associated health outcomes.

To my great-grandmother,
Hattie Bennett,
for her courage.

To my grandmother,
Ina Bennett Patterson,
for her love.

To my mother,
Rae Boganey,
for her strength.

"I changed what I could and what I couldn't, I endured."
– Dorothy Vaughan

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Chapter 1: Introduction

1.1. Air Pollution and Environmental Justice

Environmental justice is the principle that “all people and communities are entitled to equal protection of environmental and public health laws and regulations” [*Bullard*, 1993]. Unequal protection has resulted in disparities in the distribution of environmental hazards and health risks [*Bullard*, 1993]. Several studies show that environmental pollution is inequitably distributed by race and class, with racial differences persisting across socioeconomic status. Although early studies explore the relationship between demographics and pollution [*CEQ*, 1971; *Freeman*, 1972], it was not until the 1980s that racial and income disparities were framed as environmental justice [*Pulido*, 2000]. In the 1980s, two landmark studies examined the relationship between race, class, and the locations of waste facilities [*GAO*, 1983; *UCC*, 1987]. These studies demonstrated that race was the strongest predictor of where waste sites would be located in the United States.

When the 1987 United Church of Christ Commission for Racial Justice report was released, Dr. Benjamin F. Chavis, Jr., then executive director of the UCC Commission for Racial Justice, termed the racial disparities in the location of hazardous waste facilities as “environmental racism” [*Lee*, 1992]. This concept has been broadened to include not only the deliberate targeting of nonwhite communities for environmental hazards [*Lee*, 1992], but also the larger racialized sociospatial processes of residential, industrial, and land use development that produce inequities in environmental harms and benefits [*Pulido*, 2000; *Morello-Frosch*, 2002; *Taylor*, 2014]. Inequities in exposures are shaped by the legacy of job discrimination and residential segregation. Segregation stems from banking and real estate discrimination, including redlining practices and racially restrictive covenants, white flight, and exclusionary and expulsive zoning [*Pulido*, 2000; *Morello-Frosch*, 2002; *Wilson et al.*, 2008; *Taylor*, 2014]. These underlying sociospatial processes, along with exclusion from the planning and regulatory process [*Arnold*, 2007] and differential enforcement, have relegated nonwhite and low-income communities to undesirable environments with constrained mobility and increased exposures and associated health risks.

Since the early studies in the 1970s-1980s, disparities in exposures to pollution, such as waste sites, industrial facilities, and traffic, have been well documented [*Bullard et al.*, 2008; *Mohai et al.*, 2009; *Rowangould*, 2013; *Mikati et al.*, 2018]. Few studies have linked the processes that produce these disparities. Some studies have examined the relationship between residential segregation and exposure to air pollution [*Lopez*, 2002; *Gee and Payne-Sturges*, 2004; *Morello-Frosch and Jesdale*, 2006; *Jones et al.*, 2014]. These studies found that predominantly nonwhite areas are exposed to higher pollutant levels, including NO_x, $PM_{2.5}$, and toxics. *Morello-Frosch and Jesdale* [2006] showed that more residentially segregated areas had greater racial disparities in cancer risks associated with air toxics. *Houston et al.* [2004] explored the distribution of traffic. The study found that historic and structural processes including segregation, concentrated poverty, uneven land development, and construction of transportation infrastructure led to minority and high-poverty neighborhoods having higher traffic densities. More research is needed on racial and income differences in exposures to traffic-related air pollution, particularly longitudinal studies that investigate how disparities change over time [*Clark et al.*, 2017].

Current environmental justice literature asserts that it is also critical to assess how traffic-related air pollution is distributed across communities in the context of the underlying sociospatial processes that shaped and continue to affect the distribution of pollution.

1.2. Goods Movement and Air Pollution

Ports and associated yard equipment, ships, railroads, and trucks are major air pollution sources [*South Coast Air Quality Management District*, 2007]. The dominant mode of goods movement transport is heavy-duty trucks. Heavy-duty trucks are of particular concern because they are nearly all diesel-powered and major sources of nitrogen oxides (NO_x) and diesel particulate matter (PM) [*Dallmann and Harley*, 2010; *McDonald et al.*, 2012]. NO_x is associated with many adverse health effects, including increased asthma incidence [*Jacquemin et al.*, 2015], increased blood pressure [*Chan et al.*, 2015], impacts on fertility [*Frutos et al.*, 2014], increased mortality [*Jerrett et al.*, 2013], increased risk of breast and lung cancer [*Hamra et al.*, 2015; *Hart et al.*, 2015; *Hystad et al.*, 2015;], and increased visits to the emergency department for respiratory issues [*Peel et al.*, 2005]. It is also a precursor to the formation of fine particulate matter ($PM_{2.5}$, diameter < 2.5 μm) and ground-level ozone. Diesel PM has been classified as a toxic air contaminant and a carcinogen [*IARC*, 2012]. Short-term exposure to diesel exhaust is associated with impaired vascular function [*Barath et al.*, 2010]. The majority of diesel PM mass emissions is black carbon (BC), a short-lived climate forcing pollutant [*Bond et al.*, 2013].

Air pollution and associated adverse health impacts from exposure to heavy-duty diesel exhaust are elevated within distances of approximately 150-400 m of major roadways [*Zhu et al.*, 2002; *Health Effects Institute*, 2010]. The localized impacts of diesel exhaust have environmental justice implications for residents who live near roadways with high truck traffic, including freeways and major arterial roadways in goods movement corridors. Environmental justice literature indicates that nonwhite and low-income communities tend to be more concentrated in freight-impacted neighborhoods [*Gunier et al.*, 2003; *Houston et al.*, 2004]. The development of transportation infrastructure from the mid-1950s to the early 1970s contributes to disparities in traffic patterns, with roadway density highest in nonwhite and low-income neighborhoods [*Houston et al.*, 2004]. Mid-century urban planners routed freeways through poor communities of color [*Mohl*, 2000; *Mohl*, 2004; *Rose and Mohl*, 2012]. Mid-century urban transportation policies promoted segregation, physically isolating nonwhite communities in polluted environments while aforementioned policies prevented movement to other areas [*Massey and Denton*, 1993; *Geronimus*, 2000]. This has significant implications for disproportionate exposures to traffic-related air pollution, which leads to disparate health impacts, such as racial/ethnic disparities in cancer risk [*Morello-Frosch et al.*, 2001].

Transportation policies that reduce exposure to heavy-duty diesel exhaust have environmental equity and justice implications for near-roadway communities. Policy measures that mitigate diesel-related air pollution impacts include: (1) emission control systems on heavy-duty trucks; (2) designated truck routes and restrictions on truck travel; and, (3) restrictions on truck parking and engine idling in or near residential neighborhoods [*Caltrans*, 2016; *Garzón-Galvis et al.*, 2016]. Programs to deploy trucks equipped with emission control technologies reduce the air pollution burden [*Caltrans*, 2016]. Designated truck route and anti-idling strategies address local diesel impacts on communities. In 2008, California passed an anti-idling law that limits idling to five minutes [*CARB*, 2016]. Enforcement of designated truck routes and idling laws is a challenge [*Garzón-Galvis et al.*, 2016]. Some strategies to alleviate this issue include increased signage and outreach to drivers. Another strategy is to reroute truck traffic away from residential areas. Truck rerouting reduces truck traffic through and parking in affected communities [*Karner et al.*, 2009; *Gonzalez et al.*, 2011; *Garzón-Galvis et al.*, 2016]. Recent studies have investigated the emission reduction benefits from fleetwide deployment of control systems on heavy-duty trucks [*Dallmann et al.*, 2011; *Preble et al.*, 2015; *Preble et al.*, 2018]. Another study has quantified emission reductions in a neighborhood after truck rerouting [*Karner et al.*, 2009]. The emissions and exposure benefits of these measures in environmental justice communities need to be examined.

1.3. Emission Controls

Heavy-duty diesel truck emissions are being reduced over time by implementation of increasingly stringent emission standards on new engines. Recent national emission standards required reductions in PM and NO_x emissions beginning with 2007 model year engines [*California Code of Regulations Title 13*; *Code of Federal Regulations Title 40*]. These standards require 98% reductions of exhaust PM and NO_x emissions relative to uncontrolled engines. To comply with the more stringent emissions standards, exhaust after-treatment control systems have become standard equipment on new engines. Diesel particle filters (DPFs) are installed for PM reduction, and selective catalytic reduction (SCR) systems are included for NO_x control. Since heavy-duty vehicles have a long service life, natural fleet turnover results in the slow replacement of older trucks with newer and cleaner models over a time-scale of several decades. Thus, significant near-term reductions in pollutant emissions can only be achieved with additional efforts to address emissions from the legacy fleet of older trucks that remain in use. In California, the Drayage Truck and Truck and Bus Regulations were adopted to accelerate the adoption of newer emission control technologies via retrofit or replacement of older in-use engines [*CARB*, 2011; *CARB*, 2017a]. All heavy-duty diesel engines are required to be equipped with DPFs by January 1, 2018 [*CARB*, 2017a]. By January 1, 2023, all truck engines are required to be replaced with 2010 or newer models that are typically equipped with both DPF and SCR systems as emission controls for PM and NO_x, respectively [*CARB*, 2017a].

1.4. Freeway Routing

Now that some freeway infrastructure is aging and requires massive investments for repair, several cities have initiated or are considering the removal or rerouting of freeways [*Napolitan and Zegras*, 2008; *Mohl*, 2012]. More than 20 US cities have considered or planned urban freeway removals, and several cities have already completed such projects [*Mohl*, 2012]. Initial removal projects began in the 1970s in Portland, Oregon, New York City, and Boston; these cities replaced their elevated freeways with a park, boulevard, and tunnel, respectively. These removals were due to the anti-highway sentiment of the 1960s and early 1970s expressed by *Jacobs* [1961]. In the midst of the national freeway revolt and the emerging environmental movement, local grassroots organizing and environmental concerns prompted removal in these three cities [*Mohl*, 2012]. Teardown advocates seek to reroute freeways through alternative corridors or bury them in tunnels or trenches. Projects often involve building street-level boulevards in the former corridor. In the 1990s, three urban, elevated freeways were removed or rerouted in the San Francisco Bay Area and replaced with street-level boulevards after being severely damaged by the 1989 Loma Prieta earthquake: the Embarcadero and Central Freeways in San Francisco, and the Cypress Freeway in West Oakland. Seattle provides a recent example of a completed project. In 2009, it was decided to tear down the elevated, double-decked Alaskan Way Viaduct and replace it with a tunnel [*Mohl*, 2012]. Seattle opened its freeway replacement tunnel in February 2019. For communities near freeways heavily trafficked by heavy-duty truck traffic, this practice has significant implications for addressing diesel-related air pollution. Freeway rerouting can be viewed as a form of truck rerouting. A quantitative analysis of the air quality benefits in communities negatively impacted by freeway construction is needed. Freeway rerouting may be a contemporary response to the failures of mid-century urban transportation policy [*Mohl*, 2012]. More environmental justice research is needed to investigate how contemporary urban revitalization projects affect gentrification and displacement [*Taylor*, 2014].

1.5. Research Objectives

The overall goal of my research is to evaluate strategies for mitigating exposure to diesel-related air pollution, and to explore the associated environmental equity and justice implications. My specific objectives are to:

(1) Evaluate predictions of diesel-related air pollution in near-roadway settings using a line source dispersion model and air pollutant concentrations obtained from new near-roadway monitoring sites;

(2) Assess the benefits of mitigating diesel emissions using new emission control systems in conjunction with California regulations that greatly accelerate introduction of newer and cleaner engines; and,

(3) Evaluate the effects of truck routing decisions on exposure to diesel-related air pollution.

1.6. Dissertation Outline

Chapter 2 investigates the performance of the RLINE dispersion model [*Synder et al.*, 2013; *Venkatram et al.*, 2013] for diesel-dominated pollutants using data from new near-road monitoring stations in the San Francisco Bay Area. Model predictions of NO_x and BC concentrations are comprehensively evaluated against measurements from two new near-road monitoring sites that provide multiple years of continuous measurements for both pollutants. BC is a much more specific marker for diesel-related emissions than total PM mass. In my research, the use of BC rather than PM provides a more meaningful basis for evaluating predictions of near-roadway concentrations of diesel-related air pollution. I also assess the ability of the dispersion model to reproduce observed seasonal and day-of-week variations in air pollution.

Chapter 3 examines the impacts on near-roadway air pollution of the accelerated adoption of diesel particle filter (DPF) and selective catalytic reduction (SCR) emission control systems on heavy-duty diesel trucks. Impacts of diesel emission controls are compared for two neighborhoods in East Oakland: a community along the I-880 freeway with high truck traffic, and a community along a different freeway (I-580) where heavy-duty trucks are prohibited. NO_x, diesel PM, and BC emissions on each freeway are estimated prior to widespread use of diesel emission controls (2009), after universal adoption of DPFs to control diesel PM and BC (2018), and after universal adoption of SCR to control NO_x (2023). The effect of emission changes on pollutant concentrations are modeled in each neighborhood. The resulting impacts on quantitative metrics of environmental equity and justice are evaluated. Since California is at the forefront of diesel emission control efforts, results of my research provide a preview of the likely effects of a slower-moving national-scale effort to control diesel engine emissions.

Chapter 4 explores how truck routing decisions impact diesel-related air pollution and neighborhood change. I quantify the near-roadway air pollution effects of the real-world rerouting of the elevated, double-decked Cypress Freeway in West Oakland and building a street-level boulevard on the old freeway alignment, a decision resulting from environmental justice activism of West Oakland residents. Near-roadway concentrations of NO_x and BC are compared for two freeway routing scenarios: rebuild-in-place and reroute. Changes in demographic, socioeconomic, and land-use variables in West Oakland before the Cypress Freeway was destroyed (1989) and after the street-level boulevard was completed (2005) are assessed. I compare the indicators for the area along the alignment with West Oakland as a whole to examine how environmental justice activism affects gentrification and displacement.

Chapter 5 summarizes major research findings and makes recommendations for future research.

Chapter 2: Evaluating Near-Roadway Concentrations of Diesel-Related Air Pollution Using RLINE

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2.1. Introduction

On-road motor vehicles are a major source of urban air pollution. Acute and chronic exposures to traffic-related air pollution are associated with many adverse health effects, including asthma exacerbation, asthma onset, lung impairment, impacts on fertility and birth outcomes, and increased cardiovascular and respiratory mortality and morbidity [*Brauer et al.*, 2008; *Frutos et al.*, 2014; *Health Effects Institute*, 2010]. Levels of traffic-related air pollution and their associated health impacts are elevated within 150 - 200 m of major roadways [*Brugge et al.*, 2007; *Health Effects Institute*, 2010; *Zhu et al.*, 2002]. People who live, work, or attend school in close proximity to major roadways have an increased risk for adverse health outcomes. As of 2010, about 11 million people in the United States lived within 150 m of a major highway [*Boehmer et al.*, 2013]. The population living near major roadways is growing in urban areas due to new near-roadway development that results from policies encouraging transit-oriented and compact development [*CARB*, 2017b]. Non-white and low-income residents are more likely to live near major roads [*Boehmer et al.*, 2013; *Gunier et al.*, 2003; *Rowangould*, 2013]. Accurate estimates of traffic-related air pollutant concentrations are important for assessing near-roadway exposure, health risk, and environmental justice.

Dispersion models are used to predict downwind pollutant concentrations due to roadway emissions. Several models based on Gaussian formulations have been developed, including HIWAY-2 [*Petersen*, 1980], the CALINE series [*Benson*, 1984, 1992], CAR-FMI [*Harkonen et al.*, 1995], ADMS [*McHugh et al.*, 1997], and RLINE [*Snyder et al.*, 2013; *Venkatram et al.*, 2013]. RLINE is a new steady-state, line-source dispersion model developed to simulate primary pollutants in near-road environments. RLINE includes new formulations for vertical and horizontal plume spread and can simulate low wind meander conditions. Previous evaluations of the RLINE model suggest it is performing well with a majority of estimates within a factor of two of observations [*Heist et al.*, 2013; *Snyder et al.*, 2013]. However, previous model evaluations are hindered by the limited spatial coverage and duration of supporting field studies. High temporal resolution is necessary to evaluate model performance relative to diurnal, weekly, and seasonal variations in pollutant concentrations, traffic activity and emissions, and meteorology.

US EPA recently adopted new requirements to conduct near-road monitoring of air pollution, and the resulting data provide an opportunity to evaluate near-roadway dispersion models in a more comprehensive manner [*Batterman*, 2013]. In 2010, EPA added a 1-hour primary NO_2 standard and revised the minimum NO_2 monitoring requirements, mandating near-road NO_2 monitoring stations in Core Based Statistical Areas (CBSAs) with populations exceeding 500,000. EPA subsequently required co-located carbon monoxide (CO) and fine particulate matter ($PM_{2.5}$) monitoring in CBSAs with populations exceeding one million. In California, the

Bay Area Air Quality Management District (BAAQMD) measures nitric oxide (NO) and total nitrogen oxides (NO_x), with NO_2 calculated by difference, as well as CO, $PM_{2.5}$, ultrafine particles (UFPs, diameter < 100 nm), and black carbon (BC) at near-road sites [*BAAQMD*, 2015]. The new hourly near-road monitoring data enables more in-depth investigation of how model performance varies in space, time, and by pollutant. Although studies have focused on CO, NO_x, and $PM_{2.5}$ concentrations [*Batterman et al.*, 2014a; *Batterman et al.*, 2015a; *Chang et al.*, 2015; *Isakov et al.*, 2014; *Milando and Batterman*, 2018; *Snyder et al.*, 2014; *Valencia et al.*, 2018], model performance for BC, a better and more specific marker for diesel-related emissions compared to $PM_{2.5}$, is largely unexplored in the peer-reviewed literature. A recent study on RLINE performance was unable to discern the roadway contribution to $PM_{2.5}$ from background and regional sources and recommended an evaluation of BC since it is more specific to traffic emissions [*Milando and Batterman*, 2018]. Also, comprehensive model performance for NO_x and BC over time periods that encompass all seasons of the year has not yet been reported.

The main contribution of this study is to extend the evaluation of the new near-roadway pollutant dispersion model RLINE. The specific objectives are to (i) evaluate RLINE model performance by comparing model predictions to yearlong records of near-road measurements of air pollution, (ii) explore seasonal and day-of-week variations in near-roadway air pollution, and (iii) investigate model performance for diesel-dominated pollutants such as NO_x and BC. We assess NO_x and BC modeling capabilities through comparisons with two continuous near-road monitoring sites in the San Francisco Bay Area.

2.2. Methods

2.2.1. Model inputs

We apply and evaluate the RLINE dispersion model using new and extensive data from two near-roadway air monitoring sites: Laney College and San Jose – Knox Ave (Figure 2.1). The Laney College site is located near San Francisco Bay in Oakland, 20 m away from a major interstate highway (I-880), which carries approximately 230,000 vehicles per day. The San Jose near-roadway monitoring site is located 16 m away from Highway 101, with traffic volumes of approximately 260,000 vehicles per day. A bottom-up approach was used to estimate hourly emissions on individual road links where road network, traffic volume, and emission factor datasets are combined to obtain link-based emissions for the entire study domain. These emissions were used as input to a dispersion model to predict pollutant concentrations at near-roadway receptor locations. The following sections describe the input variables and datasets used to estimate air pollutant concentrations.

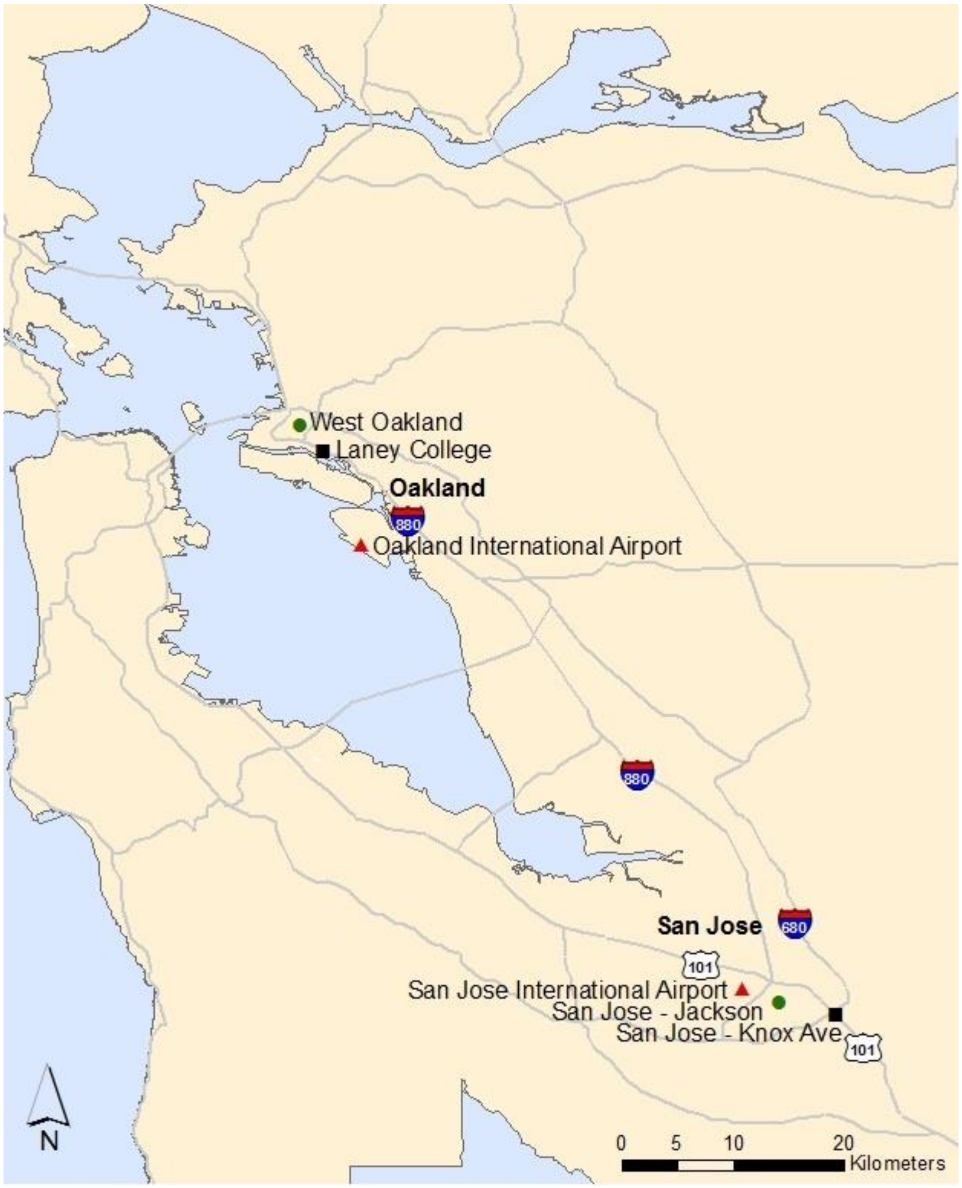

Figure 2.1. Locations of near-roadway monitors (■), background sites (●), and National Weather Service stations (▲).

2.2.1.1. Road Network

Road network data from the study area, including the coordinates (start/stop locations) of individual links, link classifications, and annual average daily traffic (AADT), were downloaded as shapefiles from the California Department of Transportation [*Caltrans*, 2017a]. The road network is represented by the corresponding centerlines. The provided roadway information contains curved and long road segments. RLINE assumes each road segment to be a straight line, so the original roadways in each study area were split into multiple segments as needed.

2.2.1.2. Traffic Activity

In this study, we used traffic count data from the California Department of Transportation. The shapefiles include link-specific counts for total vehicle annual average daily traffic (AADT), proportions of trucks, and truck traffic counts broken down by number of axles, for the years 2014 and 2015 [*Caltrans*, 2017a].

AADT values were allocated to EMFAC vehicle types [*CARB*, 2014]. We first estimated total passenger vehicles by subtracting trucks from the total vehicle count. We then disaggregated the passenger vehicles into passenger cars (LDA) and light-duty trucks (LDT) based on EMFAC estimates of vehicle miles travelled at the county-level [C*ARB*, 2014]. The truck counts were allocated to more specific vehicle types based on fuel-specific fractions as described elsewhere [*McDonald et al.*, 2014]. Two-axle/six-tire truck counts were mapped to light-heavy and medium-heavy duty trucks (LHDT and MHDT). Trucks with three or more axles were classified as heavy-heavy duty trucks (HHDT, "heavy-duty trucks"); these trucks were all assumed to be diesel-fueled [*McDonald et al.*, 2014].

Vehicle-type-specific annual traffic volumes were apportioned to hourly values using temporal allocation factors that describe patterns in traffic activity. Month-of-year, day-of-week, and hour-of-day patterns were taken from *McDonald et al.* [2014]. Briefly, weigh-in-motion (WIM) detectors were used to develop separate urban and rural temporal profiles for passenger vehicles, all trucks, and trucks with three or more axles in California. It is important to use separate temporal profiles for passenger vehicles and trucks to accurately characterize the temporal variation in traffic activity [*Batterman et al.*, 2015b]. Temporal profiles in California have not changed much over longer time scales [*McDonald et al.*, 2014]; traffic profiles for urban areas were used in this study.

We combined temporal activity profiles with annual average daily volumes to derive hourly traffic volumes for each link and vehicle type, with results as shown in Figure 2.2:

$$V_{i,v,h} = AADT_{i,v} \times M_{MON(h)} \times D_{v,DAY(h)} \times H_{DT(h)} \tag{2.1}$$

where $V_{i,v,h}$ (veh h^{-1}) is the hourly volume on link i for vehicle type v at hour h; $AADT_{i,v}$ is the annual average daily traffic (veh day^{-1}) for link i for vehicle type v; $M_{MON(h)}$, $D_{k,DAY(h)}$, and $H_{DT(h)}$ are the month-of-year, day-of-week, and hour-of-day factors, respectively, for vehicle type v at hour h; and $DT(h)$ is the day type at hour h. The day type is either Monday-Thursday, Friday, Saturday, or Sunday.

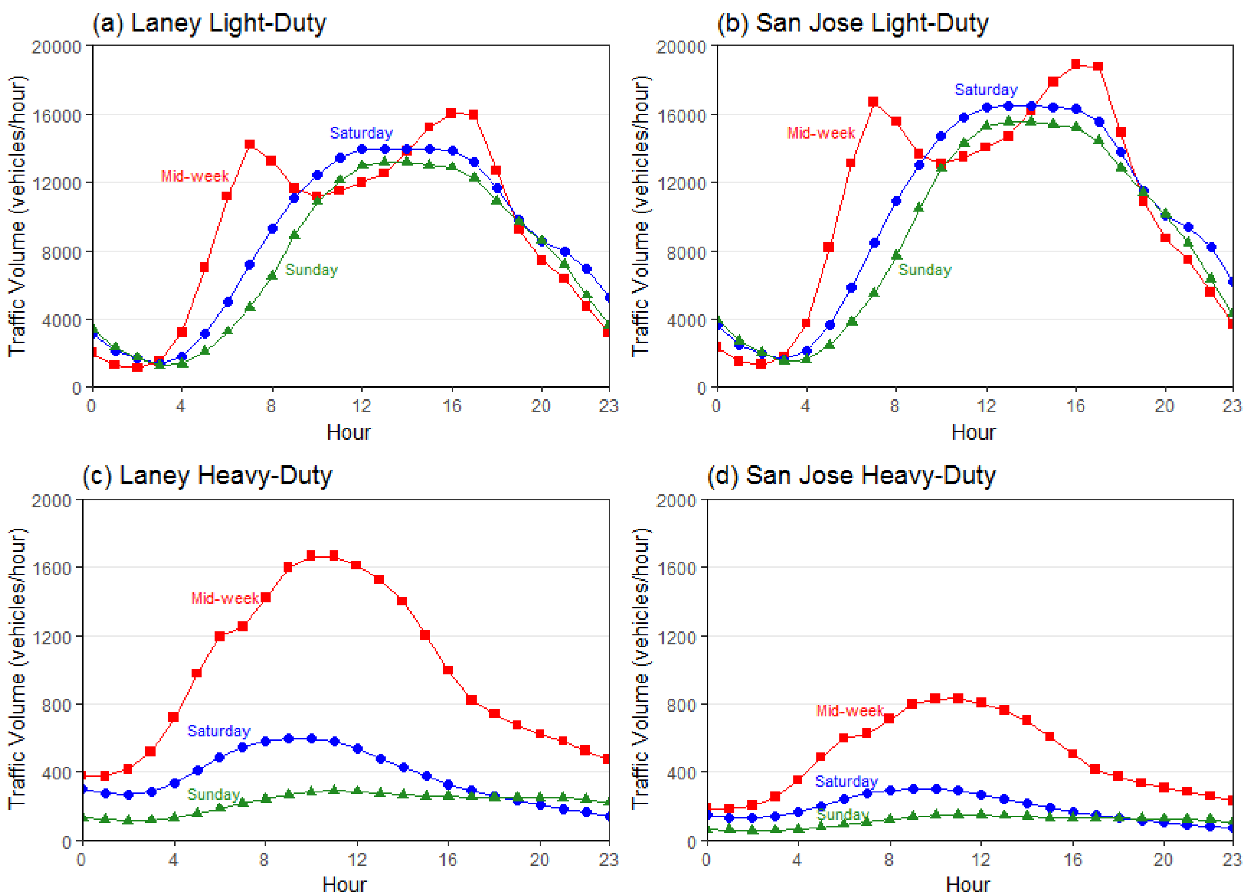

Figure 2.2. Average hourly traffic volume for mid-week and weekday days at each site. Mid-week is Tuesday–Thursday.

2.2.1.3. Emissions

Estimates of link-specific emission rates were calculated from hourly traffic volumes and emission factors. We used EMFAC model outputs at the county-level to specify NO_x and $PM_{2.5}$ emission factors expressed per unit distance traveled. Emission factors for seven vehicle types were calculated for the running exhaust mode, aggregated over all vehicle speeds and model years. We estimated BC emission factors using the EMFAC-derived exhaust $PM_{2.5}$ emission factors, combined with gasoline and diesel BC fractions for $PM_{2.5}$ emissions. The BC fractions used here were 18% and 61% for gasoline and diesel engine exhaust emissions, respectively [*Ban-Weiss et al.*, 2008; *Dallmann et al.*, 2013].

Hourly link-specific emission rates were calculated as follows:

$$E_{i,h} = \frac{1}{3600}\frac{1}{1609}\sum_{v=1}^{7} V_{i,v,h} \times EF_v \tag{2.2}$$

where $E_{i,h}$ (g m^{-1} s^{-1}) is the emission rate for link i at hour h; EF_v is the emission factor (g mile^{-1} veh^{-1}) for vehicle type v; the first fraction converts the time unit to seconds; the second fraction converts the distance unit to meters.

2.2.1.4. Meteorology

AERMET version 15181, a meteorological preprocessor [*Cimorelli et al.*, 2005], was used to process hourly surface and upper air, wind, and temperature observations, and to compute surface variables needed for dispersion modeling, including surface friction velocity (u_*), convective velocity scale (w_*), Monin-Obukhov length (L), surface roughness height (z_o), wind speed, and wind direction. Hourly surface observations from 2014 and 2015 were obtained from two National Weather Service stations: Oakland International Airport and San Jose International Airport. Upper air sounding data were obtained from the National Weather Service for Oakland International Airport.

2.2.1.5. Model Concentration

We ran RLINE version 1.2 using a unit emission rate (1 g m^{-1} s^{-1}) at release heights of 0.3 m to represent light-duty vehicles and 4 m for the heavy-duty truck portion [*Bishop et al.*, 2001]. Emission-weighted NO_x and BC concentrations were then computed by combining hourly emissions and dispersion estimates:

$$C_h = \sum_i E_{i,h,v=1-4} \times \chi_{i,h,0.3} + \sum_i E_{i,h,v=5-7} \times \chi_{i,h,4} \qquad (2.3)$$

where $\chi_{i,h}$ is the concentration in µg m^{-3} from each link, i, to the monitoring site at hour h developed using the unit emission rates [*Snyder et al.*, 2014]. The hourly concentrations were used to calculate 24-hour average and annual average concentrations.

2.2.2. Background Concentration Estimation

The total near-road concentration is the sum of traffic-related and urban background concentrations. RLINE model predictions capture the traffic-related contribution. We used the nearest-neighbor method to identify background sites (Figure 2.1). Ambient observations at the BAAQMD monitor at West Oakland were used to estimate NO_x and BC background levels at the Laney site. This background site is 2.9 km away from the Laney site. Observations at the San Jose – Jackson monitor were used to estimate background at the San Jose near-road site, which is 4.2 km away. This monitor reports NO_x and $PM_{2.5}$ concentrations. The seasonal EC/$PM_{2.5}$ ratios determined from the San Jose Speciation Trends site in *Fujita et al.* [2013] were used to estimate BC concentrations from $PM_{2.5}$.

The background sites are not free from the influence of local traffic and point sources. West Oakland is impacted by the Port of Oakland, industrial activities, and related truck traffic [*Fujita and Campbell*, 2010]. In San Jose, emissions near and upwind of the area impact NO_x levels [*BAAQMD*, 2015]. To reduce the effect of local emission sources and characterize the urban background diurnal profiles at each near-road site, we adjusted the NO_x concentrations measured at the background sites based on the 25th percentile method in *Van Poppel et al.* [2013].

Measured BC concentrations at the background sites were considered representative of urban background concentrations for this pollutant and were used without adjustment.

2.2.3. Comparison with Observations

We compared predicted concentrations of NO_x and BC to observed data at the Laney College and San Jose – Knox Avenue near-roadway measurement sites. Predicted and observed concentrations were paired in time for use in the subsequent analysis. We qualitatively compared model predictions with observations using scatter plots.

2.2.4. Model Performance Statistics

In order to evaluate model performance, we calculated statistical metrics for each pollutant, using the equations given in Table 2.1. Bias describes the tendency of the model to under- or overestimate observations and can be positive or negative. Error measures model deviation from observations and is always positive. In addition to these measures, we calculated the correlation coefficient (R) and the fraction of predictions within a factor of two of observations (FAC2). The mean fractional bias (MFB) and mean fractional error (MFE) metrics give equal weight, on a relative basis, to overpredictions and underpredictions [*Boylan and Russell*, 2006; *Boylan et al.*, 2006]. In this paper, we focus on MFB, MFE, and R to evaluate dispersion model performance.

Table 2.1. Statistical metrics and equations used to evaluate model performance.

Metrics	**Equations**
Mean bias	$\mathrm{MB} = \frac{1}{n}\sum_{i=1}^{n}(P_i - O_i)$
Mean error	$\mathrm{ME} = \frac{1}{n}\sum_{i=1}^{n}\lvert P_i - O_i\rvert$
Root-mean-square-error	$\mathrm{RMSE} = \sqrt{\frac{\sum_{i=1}^{n}(P_i - O_i)^2}{n}}$
Normalized mean bias	$\mathrm{NMB} = \frac{\sum_{i=1}^{n}(P_i - O_i)}{\sum_{i=1}^{n}O_i} \times 100\%$
Normalized mean error	$\mathrm{NME} = \frac{\sum_{i=1}^{n}\lvert P_i - O_i\rvert}{\sum_{i=1}^{n}O_i} \times 100\%$
Mean fractional bias	$\mathrm{MFB} = \frac{1}{n}\sum_{i=1}^{n}\frac{2(P_i - O_i)}{P_i + O_i} \times 100\%$
Mean fractional error	$\mathrm{MFE} = \frac{1}{n}\sum_{i=1}^{n}\frac{2\lvert P_i - O_i\rvert}{P_i + O_i} \times 100\%$

Note: P is the predicted concentration, O is the observed concentrations, and i corresponds to one of n predicted-observed pairs.

2.3. Results and Discussion

2.3.1. Emissions Contributions by Vehicle Type

Heavy-duty diesel trucks were the dominant on-road source of NO_x and BC emissions at both near-roadway monitoring sites, with contributions varying by pollutant and monitoring site (Figure 2.3). Heavy-duty trucks account for ~70% of NO_x emissions and ~80% of BC emissions at the Laney site, versus for ~47% of NO_x and ~61% of BC at the San Jose site (Figure 2.3). The greater heavy-duty diesel truck contributions to on-road BC emissions are expected given BC is a major component of diesel exhaust particulate matter [*Watson et al.*, 1994], whereas NO_x is abundant in both diesel and gasoline exhaust. That heavy-duty diesel trucks were the dominant source of on-road NO_x emissions is consistent with the increase in relative importance of diesel trucks as NO_x emissions from gasoline engines have been reduced through increasingly effective emission control technologies [*McDonald et al.*, 2012]. The contribution to on-road NO_x emissions from passenger cars is shown in Figure 2.3.

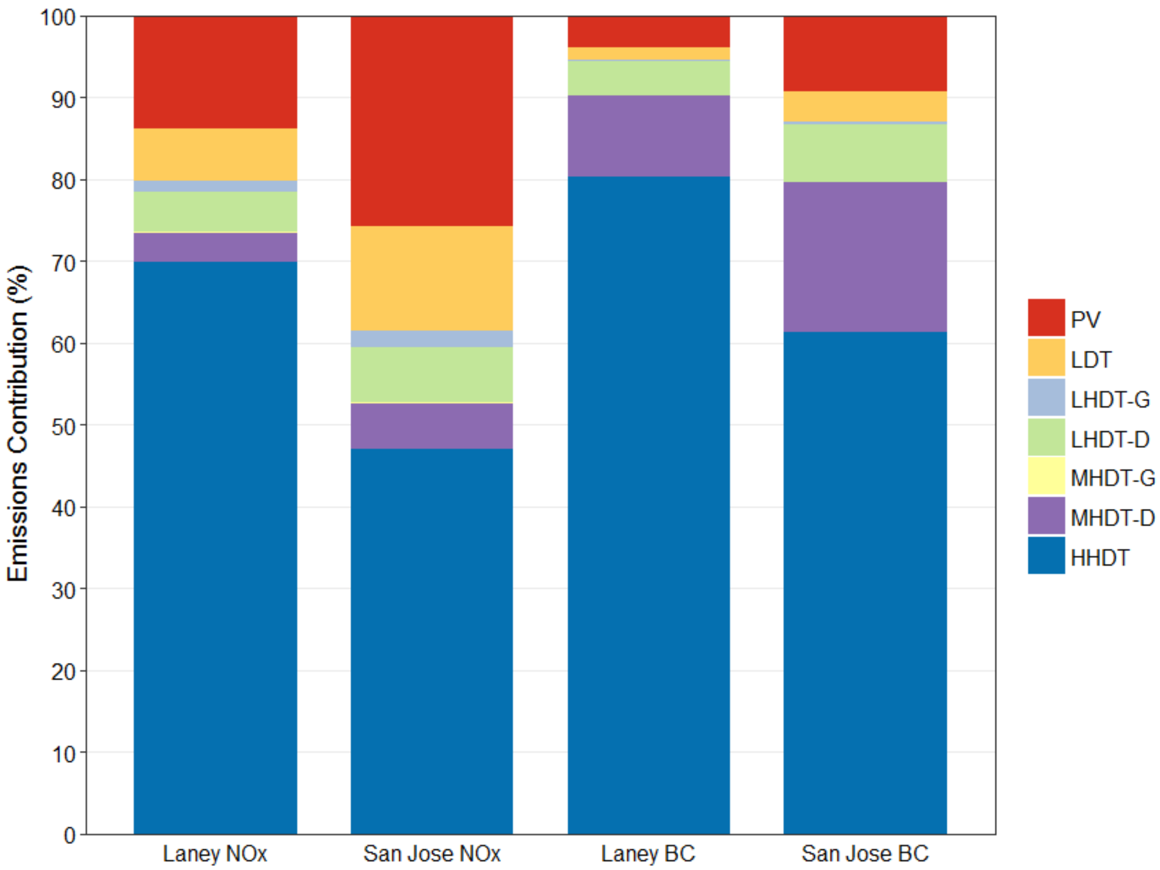

Figure 2.3. Fraction of NO_x and BC emissions by vehicle class on highways adjacent to the Laney College and San Jose – Knox Avenue near-roadway monitoring sites. Heavy-duty trucks (indicated by dark blue color) dominate emissions. (PV = passenger vehicles; LDT = light-duty trucks; LHDT-G = light-heavy duty trucks – gasoline; LHDT-D = light-heavy duty trucks – diesel; MHDT-G = medium-heavy duty trucks – gasoline; MHDT-D = medium-heavy duty trucks – diesel; HHDT = heavy-heavy duty trucks)

Emissions are more diesel-dominated at the Laney site than for the San Jose site. This is due to vehicle fleet differences. While San Jose has 30,000 more vehicles per day, ~6% are trucks with half in the heaviest (3+ axles) weight categories. In contrast, trucks account for ~11% of total traffic at Laney, of which two-thirds are in the heavy-duty category. The different traffic emissions profiles impact observed concentrations. Similar concentrations of NO_x are observed at the two sites. The observed annual average NO_x concentrations are 33 ppb and 35 ppb at the Laney and San Jose sites, respectively. While the Laney site has higher total NO_x on-road emissions, the San Jose site is impacted by emissions from nearby and upwind areas [*BAAQMD*, 2015]. Laney, the more diesel-dominated site, sees higher concentrations of BC. Observed annual average concentrations of BC are 1.4 $\mu g\ m^{-3}$ and 1.1 $\mu g\ m^{-3}$ at the Laney and San Jose sites, respectively. Predicted annual average concentrations are in better agreement with observations at the San Jose site. Annual average predictions are within 2.7–5.3% at the San Jose site, as compared to 14–15% at the Laney site.

2.3.2. NO_x and BC Performance

Figure 2.4 shows the diurnal variations of NO_x and BC concentrations. Pollutant concentrations at San Jose exhibit an earlier morning peak and more prominent evening peak compared to Laney. This pattern is consistent with an increased emissions contribution from passenger vehicles, which peak during morning and evening rush hour periods. The unimodal concentration profile observed at Laney reflects the dominance of heavy-duty diesel trucks, with associated emissions peaking in the middle of the day. The model generally captures the diurnal trends of the observations. For both pollutants, model predictions are in better agreement with the concentration profiles observed at the San Jose site. The model tends to overpredict morning and evening concentrations at the Laney site, suggesting that light-duty vehicle contributions are overestimated. Further improvements in the characterization of diesel truck traffic at heavily diesel-dominated sites may also help to improve model performance. Site-specific features of truck traffic may not be fully captured by the more aggregated statewide urban profiles for truck traffic used in this study. The model tends to underpredict midday concentrations of BC. This is most likely due to uncertainties in BC emission factors, since NO_x concentrations are predicted accurately.

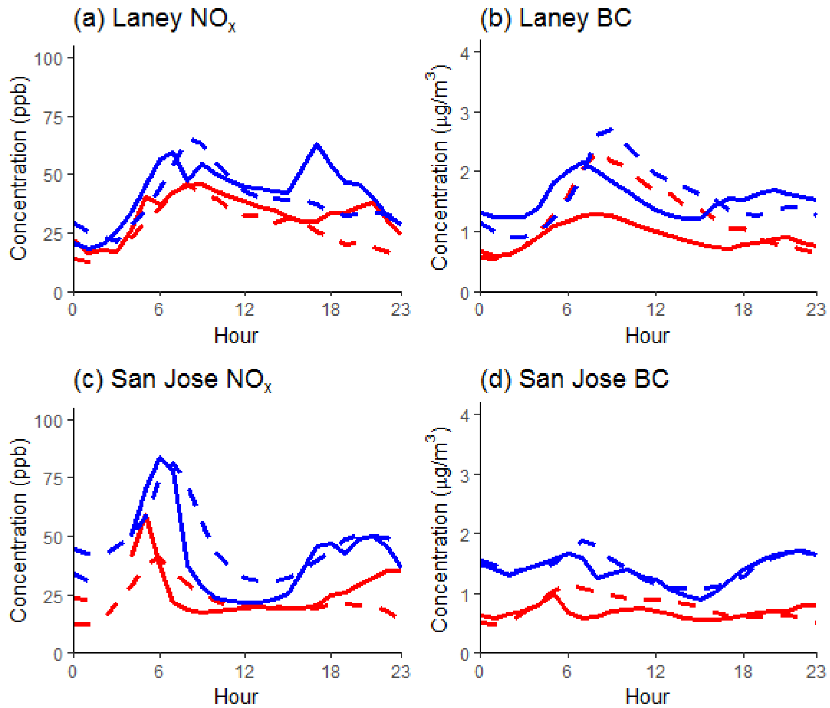

Figure 2.4. Observed (dashed line) and predicted (solid line) variations in NO_x and BC by season at near-roadway monitoring sites: (a-b) Laney College, and (c-d) San Jose – Knox Avenue. Values are averaged over all days in the data set. The red represents March – August; the blue represents September – February.

Figure 2.5 shows scatterplots comparing observed and predicted 24-hour average NO_x and BC concentrations. More than 90% of daily-average concentration predictions are within a factor of two of observations. Due to the timing of modeled diurnal trends, predictions of NO_x concentrations exceed the observed values on average, by 14% at Laney and 4% at San Jose. Predictions of BC concentrations fall below observed levels, with MFB of -20% at Laney and -5% at San Jose. The correlation between model predictions and observations is better for BC (R = 0.87–0.88) than for NO_x (R = 0.61–0.69) (see Table 2.2), which may result from uncertainty in estimates of background NO_x concentrations. In general, we found model performance to be better at the San Jose site (Figure 2.5 and Table 2.2). These results are consistent with the diurnal profile comparison described above (see Table 2.3).

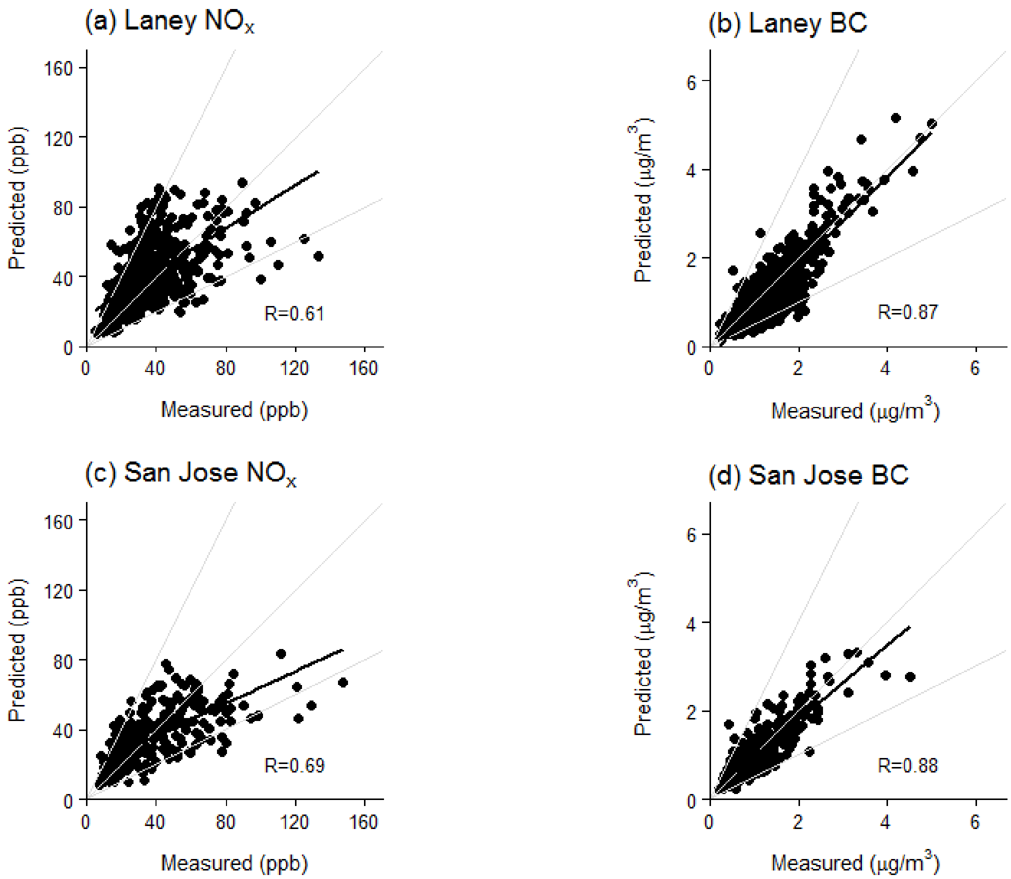

Figure 2.5. Comparison of measured and predicted 24-hour average NO_x and BC concentrations at near-roadway monitoring sites: (a-b) Laney College, and (c-d) San Jose – Knox Avenue. Grey lines show predicted to measured ratios of 2 : 1, 1 : 1, and 1 : 2.

Table 2.2. Model performance in predicting 24-hour average NO_x and BC concentrations.

	NO_x (ppb)		BC (µg m^{-3})	
	Laney	San Jose	Laney	San Jose
MB	5.0	-1.0	-0.18	-0.05
ME	11.4	10.7	0.33	0.22
RMSE	15.9	15.7	0.42	0.31
NMB	15%	-3%	-13%	-4%
NME	34%	31%	24%	20%
MFB	14%	4%	-20%	-5%
MFE	30%	30%	30%	22%
R	0.61	0.69	0.87	0.88
FAC2	0.92	0.92	0.91	0.98

Table 2.3. Model performance in predicting 24-hour average NO_x and BC concentrations by day-of-week.

	Laney		San Jose	
	Mid-week	Weekend	Mid-week	Weekend
NO_x (ppb)				
MB	7.3	0.3	1.9	-5.5
ME	14.2	6.7	11.5	9.1
RMSE	19.2	9.5	16.5	14.1
NMB	19%	1%	5%	-20%
NME	37%	30%	30%	33%
MFB	18%	5%	12%	-10%
MFE	32%	27%	29%	30%
R	0.47	0.52	0.70	0.78
BC ($\mu g\ m^{-3}$)				
MB	-0.19	-0.12	-0.05	-0.01
ME	0.36	0.21	0.22	0.20
RMSE	0.45	0.27	0.33	0.28
NMB	-11%	-14%	-4%	-1%
NME	22%	23%	19%	23%
MFB	-18%	-22%	-4%	-2%
MFE	28%	30%	20%	25%
R	0.87	0.89	0.87	0.90

2.3.3. Seasonal Variation

Figure 2.4 shows distinct seasonal patterns in diurnal profiles and NO_x and BC concentrations. The two seasons represented are March – August (“spring-summer”; indicated in red) and September – February (“fall-winter”; indicated in blue). In the fall-winter season, more prominent evening peaks are apparent. The model accurately captures higher NO_x and BC concentrations observed during the fall-winter season for hourly concentrations at both sites (Figure 2.4). Concentrations are elevated due to more stagnant atmospheric conditions. Lower mixing height and increased atmospheric stability result in higher surface level concentrations in the fall-winter season.

2.3.4. Weekday/Weekend Differences

Day-of-week profiles for NO_x and BC emissions from on-road vehicles show clear weekday/weekend differences (Figures 2.6 and 2.7). Emissions peak during the middle of the week, with large reductions on weekends. The day-of-week emissions pattern closely follows changes in activity for heavy-duty trucks. Weekend truck traffic decreases by ~70% compared to weekday averages in urban areas in California [*McDonald et al.*, 2012]. This further illustrates the dominance of heavy-duty truck emissions as indicated in Figure 2.3. On-road BC emissions show a larger weekday/weekend difference, consistent with the greater heavy-duty diesel truck contribution to BC emissions. The Laney site, which is more dominated by heavy-duty trucks, sees larger weekday/weekend differences.

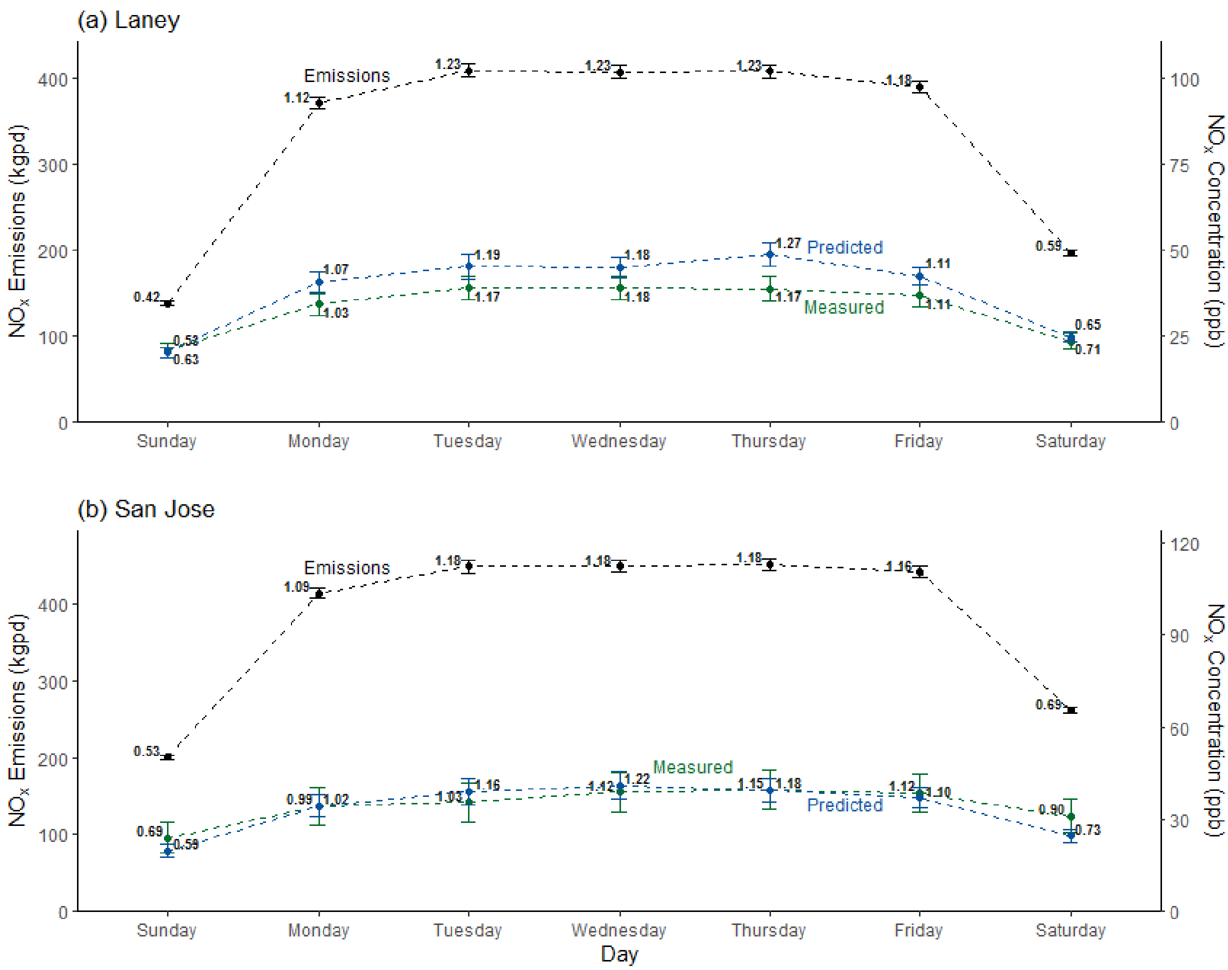

Figure 2.6. Day-of-week patterns of on-road emissions and 24-hour average NO_x concentrations at near-roadway monitoring sites: (a) Laney College, and (b) San Jose – Knox Avenue. Each point represents the average value for each day. Numbers next to each point indicate the ratio of the day-specific value to the weekly average. For the San Jose site, measured values are to the left and predicted values are to the right. Uncertainty estimates indicate 95% confidence levels.

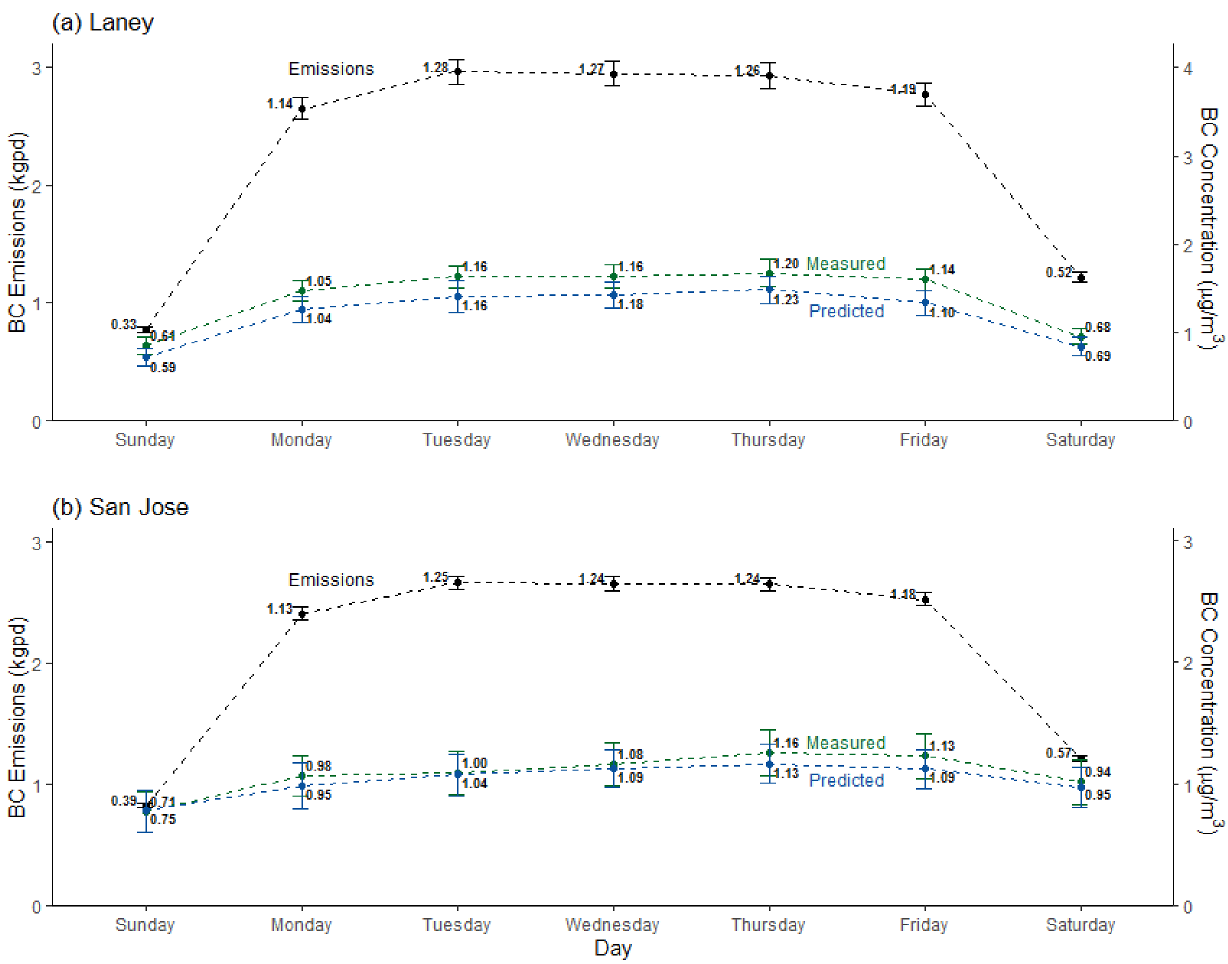

Figure 2.7. Day-of-week patterns of on-road emissions and 24-hour average BC concentrations at near-roadway monitoring sites: (a) Laney College, and (b) San Jose – Knox Avenue. Each point represents the average value for each day. Numbers next to each point indicate the ratio of the day-specific value to the weekly average. Uncertainty estimates indicate 95% confidence levels.

Observed NO_x and BC concentrations follow the same trend as day-of-week emissions. Observations show less weekday/weekend difference compared to the assumed day-of-week emissions profiles, probably due to uncertainties in the extent of route-specific emissions reductions that occur on weekends. Weekday/weekend differences are larger at the Laney site as compared to San Jose, which is consistent with a higher contribution to overall emissions from heavy-duty trucks. The observed weekday/weekend difference is larger for BC than NO_x at the Laney site, which is consistent with the on-road emissions profile. In contrast, the difference is slightly larger for NO_x at the San Jose site.

The model accurately reproduces observed weekday/weekend differences in pollutant concentrations. Weekday/weekend differences in NO_x are overestimated, with lower predicted NO_x concentration ratios on weekends than observed concentration ratios (Figure 2.6). At the Laney site, NO_x concentrations are overpredicted on weekends (MFB = 5%), whereas at the San

Jose site, weekend NO_x concentrations are underpredicted (MFB = -10%). Predictions of weekday/weekend differences in BC agree well with observations (Figure 2.7). However, BC concentrations are underestimated at both sites, which again suggests BC emission factors are underestimated.

2.4. Conclusions

In this study, we evaluated the RLINE dispersion model against yearlong hourly NO_x and BC measurements at two near-road monitoring sites in the San Francisco Bay Area. Novel features of this work include a focus on diesel-related pollutants and use of yearlong records of daily data from new near-road monitoring stations.

Significant contributions to NO_x and BC at both sites come from heavy-duty diesel trucks. Heavy-duty trucks were more important as a source of BC versus NO_x and were more dominant at Laney. The overall agreement between model predictions and observations for 24-hour average NO_x and BC was reasonably good, with more than 90% of predictions within a factor of two of the near-road monitoring data.

Overall, the model is also able to accurately reproduce seasonal and day-of-week variations in concentrations. Model performance would be improved with more accurate estimates of BC emission factors, and site-specific data on the diurnal and day-of-week variations in truck traffic, especially at locations located near major goods movement corridors, such as ports and railyards. we conclude that RLINE is a useful assessment tool in predicting diesel truck-related air pollution in near-roadway environments.

Chapter 3: Effects of Diesel Engine Emission Controls on Environmental Equity and Justice

3.1. Introduction

Freight transport is a major source of air pollution along goods movement corridors. Heavy-duty diesel trucks, the dominant mode of freight transport activity [*US DOT*, 2016], are a major source of nitrogen oxides (NO_x) and diesel particulate matter (PM) emissions [*Dallmann and Harley*, 2010; *McDonald et al.*, 2012]. The majority of diesel PM mass emissions is black carbon (BC), a short-lived climate-forcing pollutant [*Bond et al.*, 2013]. Exposures to NO_x and diesel PM have been associated with adverse health effects, such as asthma incidence, impaired vascular function, cancer, and cardiovascular and respiratory mortality [*Lloyd and Cackette*, 2001; *Barath et al.*, 2010; *IARC*, 2012; *Jerrett et al.*, 2013; *Hamra et al.*, 2015; *Jacquemin et al.*, 2015]. The air pollution and health impacts from diesel exhaust can be highly localized [*Zhu et al.*, 2002; *Kozawa et al.*, 2009]. This is of particular concern for residents living near roadways with high truck traffic, including freeways and arterial roadways that carry the majority of goods movement-related traffic [*Kozawa et al.*, 2009]. Moreover, based on location of residence, nonwhite individuals experience higher concentrations of traffic-related air pollution than white individuals [*Clark et al.*, 2014; *Clark et al.*, 2017; *Rosofsky et al.*, 2018]. For instance, one study found that average outdoor nitrogen dioxide (NO_2) levels are 38% higher for people of color than for non-Hispanic whites [*Clark et al.*, 2014].

Policies to control diesel truck emissions have great potential to reduce near-roadway air pollution, especially along goods movement corridors. National heavy-duty diesel engine emission standards require that exhaust PM emissions be reduced to 0.01 g/bhp-hr beginning with the 2007 model year engines and NO_x emissions be reduced to 0.2 g/bhp-hr beginning with 2010 engines [*California Code of Regulations Title 13*; *Code of Federal Regulations Title 40*]. To comply with these more stringent emission standards, exhaust after-treatment control systems have become standard equipment on new diesel engines. Diesel particle filters (DPFs) are installed for PM reduction, and selective catalytic reduction (SCR) systems are included for NO_x control. Since heavy-duty vehicles have a long service life, natural fleet turnover results in the slow replacement of older trucks with newer and cleaner models over a time-scale of several decades. Thus, significant near-term reductions in pollutant emissions require additional efforts to address emissions from the legacy fleet of older trucks that remain in use. In California, the Drayage Truck and Truck and Bus Regulations were adopted to accelerate the use of newer emission control technologies via retrofit or replacement of older in-use diesel engines [*CARB*, 2011; *CARB*, 2017a]. All on-road heavy-duty diesel engines are required to meet the 2007 exhaust PM emission standard by January 1, 2018, typically met using DPFs [*CARB*, 2017a]. By January 1, 2023, all heavy-duty trucks must have 2010 or newer engines that are typically equipped with both DPF and SCR systems [*CARB*, 2017a].

Studies evaluating the effect of accelerated adoption of control technologies on heavy-duty truck emissions in California show that emission reductions outpace what would have been achieved by natural fleet turnover [*Dallmann et al.*, 2011; *Bishop et al.*, 2013; *Kozawa et al.*, 2014; *Preble et al.*, 2015; *Haugen et al.*, 2018; *Preble et al.*, 2018]. While these studies quantify the emissions benefits of diesel engine control strategies, they do not consider impacts on near-roadway exposures to air pollution or environmental equity implications. California regulatory agencies, including the California Air Resources Board (CARB), are required to consider the environmental equity implications of their regulatory implementation and enforcement activities [*EO 12898*; *AB 32*], including assessing the unequal distribution of environmental hazard and health risk burden among racial, ethnic, or socioeconomic groups [*EPA*, 1992a,b; Cutter, 1995]. Prior studies of equity and justice benefits of air pollution control strategies focus on hypothetical scenarios for power plants [*Levy et al.*, 2007], urban bus fleets [*Levy et al.*, 2009], regional emissions [*Fann et al.*, 2011], and diesel engines [*Marshall et al.*, 2014].

The current study builds on this emerging literature by considering the equity implications of newly implemented heavy-duty diesel truck emission control strategies and evaluating the air quality effects of the accelerated adoption of emission control systems on heavy-duty trucks in the San Francisco Bay Area. Our specific objectives are to: (i) estimate the impact of the accelerated deployment of DPF and SCR systems on NO_x, diesel PM, and BC emissions; (ii) model the effect of emission changes on near-roadway air pollution in different neighborhoods; and, (iii) evaluate the environmental equity implications of estimated emission changes.

3.2. Methods

3.2.1. Study Sites

This study focuses on the East Oakland freight corridor in the San Francisco Bay Area. Two major interstate freeways traverse the corridor: I-880 and I-580. I-880 supports intraregional freight transport in the East Bay and connects to an interregional network that provides access to the Port of Oakland, Oakland International Airport, and the rest of California and the nation. I-880 carries the highest volume of trucks in the region [*MTC*, 2015].

Conversely, trucks are prohibited along a segment of I-580 in East Oakland (Figure 3.1). This prohibition began in 1951 along MacArthur Boulevard, prior to the construction of I-580, due to resident opposition to truck traffic [*Caltrans*, 2017b]. In 1963, as I-580 was being constructed in the same area, an ordinance was passed by the City of Oakland prohibiting trucks with gross vehicle weight exceeding 4.5 tons from using the new freeway. That same year, the California Department of Transportation and the Federal Highway Administration approved continuation of the truck ban. In 1989, part of I-880 was closed after sustaining damage by the Loma Prieta earthquake. In response to the increased traffic congestion, the California Trucking Association advocated for the study and review of the truck prohibition on I-580, but ultimately no study was conducted due to continued opposition from local residents and elected officials. In 2000, the California Legislature codified the I-580 truck ban in the California Vehicle Code [*Caltrans*, 2017b]. As a consequence, all heavy-duty truck traffic in Oakland, including Port- and airport-related cargo movement, continues to travel exclusively on the I-880 corridor.

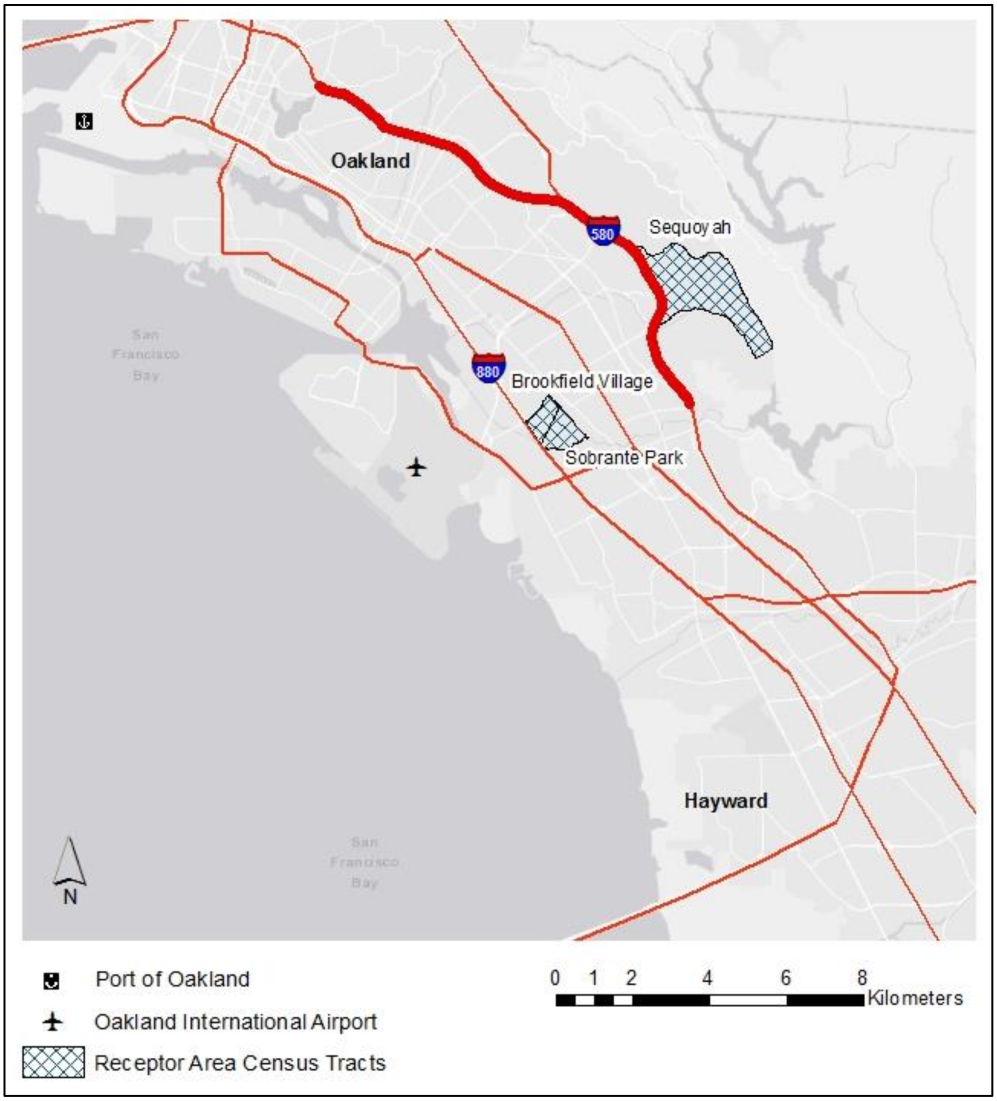

Figure 3.1. Map of study area, including receptor communities located beside major freeways. Thick red line represents segment of I-580 along which heavy-duty trucks are prohibited.

This analysis assesses the air quality impacts and equity implications of emission control regulations accelerating the deployment of DPF and SCR systems, in two communities that are adjacent to the I-880 and I-580 corridors: Brookfield Village and Sobrante Park along I-880 (hereafter referred to as "Brookfield-Sobrante") and Sequoyah along I-580 (Figure 3.1). Demographic and socioeconomic data for each community are presented in Table 3.1 [*U.S. Bureau of the Census*, 2016]. Brookfield-Sobrante, which experiences much greater traffic impacts from goods movement activities, has a higher proportion of non-white and low-income residents than Sequoyah. This is consistent with environmental justice literature that indicates more generally that non-white and low-income communities tend to be more concentrated in freight-impacted neighborhoods [*Gunier et al.*, 2003; *Houston et al.*, 2004].

Table 3.1. Demographic variables in each receptor area.[a]

Variable	Brookfield Village - Sobrante Park	Sequoyah
Total Population (count)	5997	3449
Population Characteristics (%)		
Age (years)		
< 5	9	8
5-19	23	10
20-39	30	20
40-59	27	30
≥ 60	11	32
Less than high school education	35	7
Renters	45	13
Household income < $20,000	20	6
Below poverty[b]	50	15
Race/ethnicity		
Latino (Hispanic of all races)	45	15
Non-Hispanic African-American	36	47
Non-Hispanic Asian/Pacific Islander	6	9
Non-Hispanic White	3	24

[a] 2012-2016 American Community Survey 5-Year Estimates
[b] below 200% the poverty level

3.2.2. Emissions

A bottom-up approach was used to estimate link-based emissions for the modeled roadway network. Network data, obtained from the California Department of Transportation [2017a], includes individual link locations as well as link classifications. The modeled roadway segments extend approximately 4 km along each of the two freeways.

Requirements for accelerated adoption of emission control systems on heavy-duty diesel trucks began to take effect in January 2010, with use of DPFs fully phased in by the end of 2017 and use of SCR to be fully phased in by the end of 2022. The years 2009, 2018, and 2023 were therefore selected to provide baseline, post-DPF policy, and post-SCR policy points of comparison, respectively. We used traffic count data from the California Department of Transportation [*Caltrans*, 2017a], including link-specific counts for total annual average daily traffic (AADT), proportions of trucks, and truck counts broken down by number of axles. We estimated AADT for years 2018 and 2023 by extrapolating traffic count data for years 2014 to 2016 (Table 3.2). During the earlier period (2009-2013), traffic volumes declined due to the economic recession.

Table 3.2. Changes in traffic volumes (AADT[a]) over time.

	Baseline 2009	Post-DPF 2018	Post-SCR 2023	Percent Change[b]	
				2018	2023
Brookfield-Sobrante:					
Total	215000	230000	255000	7%	19%
Heavy-duty diesel trucks	11334	12125	13443		
Sequoyah:					
Total	153000	182000	202000	19%	32%
Heavy-duty diesel trucks	45	54	60		

[a] AADT is annual average daily traffic. Heavy-duty diesel truck counts are calculated to nearest whole number.
[b] Percent changes are relative to baseline year (2009).

AADT values were allocated to EMFAC [*CARB*, 2017c] light-duty and heavy-duty vehicle types. We first estimated total light-duty vehicles by subtracting trucks from the total vehicle count. Light-duty vehicles were disaggregated into passenger cars and light-duty trucks based on EMFAC estimates of vehicle miles travelled at the county-level [*CARB*, 2017c]. The heavy-duty truck counts are grouped by axle category: two-axle/six-tire trucks and trucks with three or more axles. Two-axle/six-tire trucks include both gasoline and diesel-powered trucks that were apportioned based on fuel-specific fractions as described elsewhere [*McDonald et al.*, 2014]. Trucks with three or more axles were all assumed to be diesel-fueled [*McDonald et al.*, 2014]. For our analysis, we refer to trucks with three or more axles as "heavy-duty diesel trucks."

Traffic volumes were adjusted to hourly estimates using month-of-year, day-of-week, and hour-of-day temporal allocation factors from *McDonald et al.* [2014] (see Equation 2.1). Estimates of link-specific emission rates were calculated from hourly traffic volumes and emission factors (see Equation 2.2). NO_x and $PM_{2.5}$ emission factors for each vehicle type were estimated using EMFAC model estimates at the county-level. BC emission factors were estimated using the EMFAC-derived exhaust $PM_{2.5}$ emission factors, combined with BC fractions for $PM_{2.5}$ emissions. The BC fractions used here were 18% and 61% for gasoline and diesel engine exhaust emissions, respectively [*Ban-Weiss et al.*, 2008; *Dallmann et al.*, 2013].

3.2.3. Exposure Modeling

We predicted near-roadway pollutant concentrations using the RLINE line-source dispersion model [*Snyder et al.*, 2013; *Venkatram et al.*, 2013]. We previously evaluated RLINE model performance in predicting NO_x and BC concentrations at near-roadway monitoring sites in the San Francisco Bay Area [*Patterson and Harley*, 2019]. For the present study, model receptors were set at all census-block centroids located within 150 m of each freeway. Levels of traffic-related air pollution and their associated health impacts are reported to be elevated at distances up to 150 m downwind of major roadways [*Zhu et al.*, 2002; *Health Effects Institute*, 2010]. The meteorological inputs required for RLINE dispersion calculations were developed using the meteorological preprocessor, AERMET [*Cimorelli et al.*, 2005]. Meteorological data were obtained from the National Weather Service for Oakland International Airport for the baseline year 2009. We held meteorological conditions constant (i.e., same as 2009) when modeling the 2018 and 2023 emission scenarios so meteorology could not influence predicted changes in air quality. The model was applied using unit emissions and the roadside barrier beta-option at varying release heights for light-duty vehicles (0.3 m) and heavy-duty trucks (4 m) [*Bishop et al.*, 2001]. Dispersion model estimates were then combined with hourly emissions estimates to compute NO_x, diesel PM, and BC concentrations.

Near-road pollutant concentrations were calculated as the sum of the modeled freeway and urban background concentrations. Observations from the Bay Area Air Quality Management District's East Oakland ambient monitoring site were used to estimate background NO_x, diesel PM, and BC levels. This monitoring site provides hourly measurements of NO_x and $PM_{2.5}$ concentrations. Seasonal ratios from *Fujita et al.* [2013] were used to estimate background diesel PM and BC concentrations as fractions of measured $PM_{2.5}$. Hourly $PM_{2.5}$ monitoring at the East Oakland site began in October 2009, so we estimated 2009 background concentrations using 2010 NO_x and $PM_{2.5}$ measurements. Background concentrations for the years 2018 and 2023 were estimated using 2017 measurements. To reduce the influence of local NO_x emission sources at the background site [*BAAQMD*, 2015] and characterize the urban background contribution accurately, we used the 25th percentile of observations from the East Oakland monitor to characterize urban background levels, as recommended by *Van Poppel et al.* [2013]. Measured BC concentrations were considered representative of urban background concentrations and were used without adjustment. Background concentrations were added to near-roadway estimates of traffic-related air pollution for each location in the study domain. Daily average concentrations were computed at each receptor from predicted hourly values.

3.2.4. Environmental Equity Metric

3.2.4.1. Intake

Intake is the mass of pollutant inhaled by an exposed individual or population and is a measure of exposure. The intake rates, I ($\mu g\ d^{-1}$), of NO_x, diesel PM, and BC were estimated for each census-block as:

$$I = \sum_{i=1}^{n} C_i Q_i \qquad (3.1)$$

where C_i is the estimated daily average concentration ($\mu g\ m^{-3}$) for person i, Q_i is the average breathing rate for person i, and n is the number of people in a census-block. We assumed a time-invariant breathing rate Q_i = 12 m^3 $person^{-1}$ d^{-1} [*Layton*, 1993]. We assigned all individuals residing within a census-block to the pollutant concentrations estimated at the block centroid, so we assumed that people spend all their time at home, outdoors [*Levy et al.*, 2009; *Marshall et al.*, 2014; *Rosofsky et al.*, 2018]. Since this study focuses on exposure to traffic-related air pollution, there are no indoor sources.

3.2.4.2. Equity Metric

We used an environmental equity metric to compare exposure estimates in the two study areas and assess the spatial distribution of the exposure burden. Our metric for environmental equity is the relative percent difference (RPD) between mean intakes for residents of the two different communities we analyzed:

$$\mathrm{RPD} = \frac{|\mu_{\mathrm{SQ}} - \mu_{\mathrm{BS}}|}{\mu} \tag{3.2}$$

where μ_{SQ} and μ_{BS} (μg d^{-1}) are the mean intakes for Sequoyah and Brookfield-Sobrante, respectively and μ is the population mean intake value (μg d^{-1}). This metric captures the difference in exposures between groups.

We computed this metric for the baseline (2009), post-DPF policy (2018), and post-SCR policy (2023) years, and we examined changes to evaluate the effects of diesel truck emission control regulations on equity in exposures to air pollution.

3.3. Results and Discussion

3.3.1. Heavy-Duty Diesel Truck Emissions

Trends in heavy-duty diesel truck NO_x, diesel PM, and BC emissions along each freeway are shown in Table 3.3. As expected, between the baseline year (2009) and the post-policy years (2018 and 2023), there were large estimated decreases in heavy-duty diesel truck emissions. During this period, heavy-duty diesel truck volumes on I-880 increased, as shown in Table 3.2, but emission reductions outpaced the effect of growth in diesel truck traffic. For I-880, NO_x emissions decreased by 65% and diesel PM and BC emissions decreased by 83% between 2009 and 2018 as use of DPFs became universal. Particle filters have been shown to have little impact on total NO_x emissions [*Liu et al.*, 2008; *Herner et al.*, 2009]. The decrease in NO_x emissions is due to the replacement of older trucks with newer engines that emit NO_x at lower levels. Reductions in diesel PM and BC emissions are driven by both filter retrofits on older trucks and replacement with newer engines. Further reductions in heavy-duty diesel truck emissions of all pollutants will occur by 2023, when all trucks are required to be equipped with 2010 or newer engines, typically with both DPF and SCR. Relative to our 2009 baseline, NO_x emissions decreased by 82%, and diesel PM and BC emissions decreased by 95% for the 2023 case. Despite large relative decreases, heavy-duty diesel truck emissions remain much higher on I-880 than on I-580. Trucks account for 8.6% of total traffic along I-880, with 61% of trucks classified in the heavy-duty diesel truck category. In contrast, <1% of vehicles traveling on I-580 are trucks, of which only 6% are in the heavy-duty diesel truck category. Heavy-duty diesel trucks remain the dominant source of NO_x, diesel PM, and BC emissions on I-880.

Table 3.3. Average heavy-duty diesel truck NO_x, diesel PM, and BC emissions for baseline and post-DPF and post-SCR scenarios. The modeled freeway segment near each study area is approximately 4 km.

Pollutant	Study Area	Emission Estimates		
		Baseline	**Post-DPF (2018)**	**Post-SCR (2023)**
NO_x ($kg\ km^{-1}\ d^{-1}$)	Brookfield-Sobrante	118	40.6	21.7
	Sequoyah	0.48	0.18	0.10
PM ($g\ km^{-1}\ d^{-1}$)	Brookfield-Sobrante	4485	753	203
	Sequoyah	18.2	3.4	0.91
BC ($g\ km^{-1}\ d^{-1}$)	Brookfield-Sobrante	2721	457	123
	Sequoyah	11.0	2.0	0.55

The accuracy of predicted emission trends over time is affected by uncertainties in the underlying emission factors. In Table 3.4, we compare trends in the EMFAC-derived emission factors for heavy-duty diesel trucks used in this study with observed trends in emission factors from on-road studies [*Preble et al.*, 2019]. Studies of heavy-duty diesel trucks operating in California reported fuel-based emission rates, so we derived fuel-based EMFAC estimates for comparison. As shown in Table 3.4, EMFAC has a tendency to estimate lower heavy-duty diesel truck emission factors than on-road measurements. However, the emission reductions used in this study are consistent with observed emission reductions measured from on-road engines, which is more relevant to our temporal analysis comparing baseline and post-policy years.

Table 3.4. Evaluation of heavy-duty diesel truck emission factors and trends over time.

	Exhaust Emission Factors ($g\ kg^{-1}$)			**Percent Change[a]**	
	Baseline[b]	**Post-DPF**	**Post-SCR**	**2018**	**2023**
NO_x:					
EMFAC[c]	27.8	10.5	5.8	62%	79%
On-road[d]	31.3 ± 1.6	13.2 ± 1.0	4.4 ± 0.4	58 ± 4%	86 ± 1%
BC:					
EMFAC[c]	0.65	0.12	0.03	82%	95%
On-road[d]	0.86 ± 0.11	0.18 ± 0.04	0.04 ± 0.01	79 ± 5%	95 ± 1%

[a] Percent changes are relative to baseline year
[b] The baseline year is 2009 for EMFAC-derived emission factors and 2010 for on-road emission factors
[c] Computed from EMFAC-derived emission factors. Black carbon fraction of total diesel exhaust PM emissions was assumed to be 61% (see text).
[d] On-road measurements of heavy-duty diesel truck emissions at the Caldecott tunnel.[59] The Post-SCR case is based on values for SCR-equipped trucks only.

3.3.2. Near-Roadway Pollutant Concentrations

Figure 3.2 shows boxplot statistics of daily average NO_x, diesel PM, and BC concentrations for the pre- and post-policy periods. Overall, we found reductions in near-roadway concentrations of all pollutants. Reductions in NO_x are seen in both study areas, but reductions are larger in Brookfield-Sobrante, where mean NO_x concentrations were reduced by 63±3% and 75±3% in 2018 and 2023 relative to baseline values for 2009. For Sequoyah, mean NO_x concentrations were reduced by 52±3% and 65±3% in 2018 and 2023, respectively, relative to 2009. As shown in Figure 3.2, diesel PM and BC were reduced in Brookfield-Sobrante but increased slightly in Sequoyah. In Brookfield-Sobrante, mean diesel PM concentrations decreased 48±4% and 56±4% in 2018 and 2023, with nearly identical changes in BC over the same period. Mean diesel PM and BC concentrations in Sequoyah increased by 19±7% and 15±6%, respectively, between 2009 and 2018, and these pollutant concentrations are predicted to remain at similar levels between 2018 and 2023. Emission reductions in Brookfield-Sobrante are driven by diesel truck emission controls. We observed a larger reduction there than in Sequoyah, which has a higher proportion of light-duty vehicle traffic in the mix. These results provide quantitative support for the expectation that heavily freight-impacted communities will experience larger relative air quality benefits from the accelerated adoption of DPF and SCR systems on diesel trucks. This is especially true for diesel PM and BC. However, absolute pollutant levels remain higher in Brookfield-Sobrante.

Our analysis shows that by 2023, mean NO_x levels will remain slightly higher, but diesel PM and BC concentrations along I-880, where heavy-duty trucks are allowed, are expected to be similar to those along I-580, where heavy-duty trucks are prohibited. This highlights the benefits of modernizing the truck fleet to reduce near-roadway exposures to diesel PM in highly impacted areas. In the baseline year, the relative difference in concentrations was larger for diesel PM and BC than for NO_x. After DPF implementation in 2018 and SCR implementation in 2023, NO_x exhibits the largest relative difference between neighborhoods. An important implication of this analysis is that near-roadway measurements of NO_x alone do not fully represent the diesel source, which is critical when assessing environmental burdens of air pollution on people living close to major goods movement corridors. Our results highlight the utility of assessing a multi-pollutant suite that more completely characterizes the air pollution burden due to on-road truck emissions. Due to its usefulness as a marker for diesel exhaust emissions [*Lloyd and Cackette*, 2001], we recommend that BC be measured at more near-roadway monitoring sites nationwide, as is already being done in the San Francisco Bay Area.

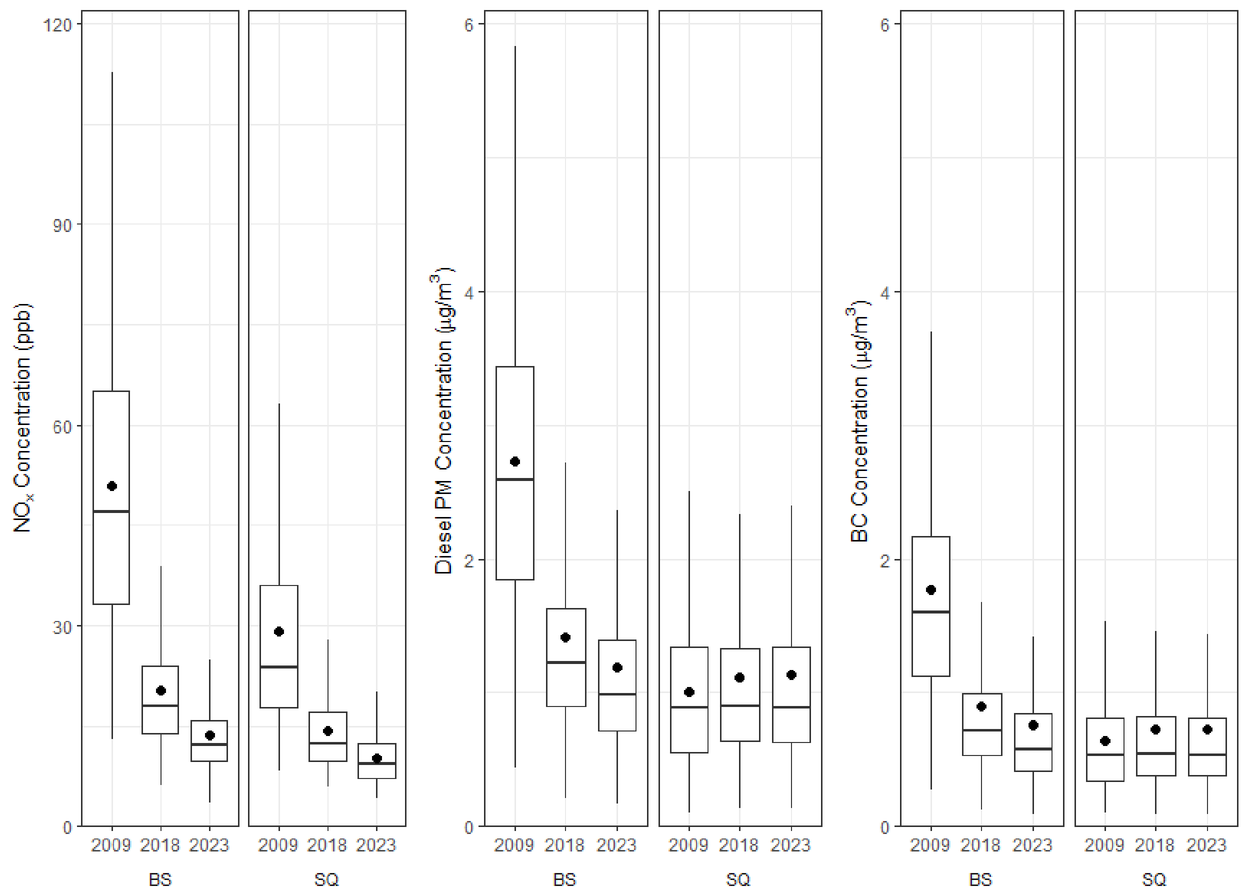

Figure 3.2. Boxplots of predicted NO_x, diesel PM, and BC concentrations for the baseline (2009 – leftmost box in each panel) and post-policy (right two boxes in each panel: 2018 – post-DPF and 2023 – post-SCR) periods. The edges of the box represent the inter-quartile range (25th to 75th percentile); the horizontal line represents the median; and the dot represents the annual average. BS = Brookfield-Sobrante with trucks on nearby I-880; SQ = Sequoyah without trucks on nearby I-580.

3.3.3. Changes in Environmental Equity Metric

Changes in the equity metric resulting from the accelerated adoption of emission controls are presented in Table 3.5. Emission reductions result in decreased exposure disparities for all pollutants over time, as measured by the relative percent difference in exposures for each year. The diesel clean-up efforts have led to a more equitable distribution of air pollution between Brookfield-Sobrante and Sequoyah. This indicates that the disproportionate air pollution burden on residents living along I-880 with high volumes of heavy-duty truck traffic is reduced. Future work should investigate changes in associated health outcomes, such as premature mortality or asthma hospitalization, in the East Oakland freight corridor for each emission control regulation. Sequoyah and other neighborhoods along the segments of I-580 where heavy-duty trucks are prohibited could also provide a useful control case for epidemiological studies, given that exposures to diesel exhaust along I-580 are low and not changing much over time.

Table 3.5. Environmental equity indicator for daily-average NO_x, diesel PM, and BC concentrations. The last two columns show absolute changes in the environmental equity metrics relative to the baseline year (2009), with positive values indicating that the policy improved equity. For example, DPF implementation improved the equity metric by 69% for diesel PM (from 91% to 22%).

	Baseline 2009	**Post-DPF 2018**	**Post-SCR 2023**	**Improvement 2018**	**Improvement 2023**
Equity Indicator (Relative Percent Difference)					
NO_x	58%	35%	27%	23%	31%
Diesel PM	91%	22%	7%	69%	85%
BC	89%	21%	5%	68%	84%

It is relevant to consider the factors that have contributed to disparate environmental patterns. Our two study areas, Brookfield-Sobrante and Sequoyah, are two distinct geographies that result from the historical racialized processes of redlining, restrictive covenants, and zoning [*McClintock*, 2011]; growth in immigration from Latin America after shifts in U.S. immigration policy in the mid-1960s [*McClintock*, 2011; *Massey and Pren*, 2012]; and black suburbanization caused by push and pull factors [*Hansen*, 1996; *Ginwright and Akom*, 2017]. As shown in Table 3.1, Sequoyah has fewer non-white residents, higher rates of homeownership, and higher household income than Brookfield-Sobrante. Home ownership is a proxy for political clout [*Morello-Frosch et al.*, 2001], and income, race, and ethnicity are among the determinants of the distribution of power in the U.S. [*Boyce et al.*, 1999]. Thus, as a result of differential access to political power [*Boyce et al.*, 1999], trucks are prohibited on I-580 in the Oakland Hills, and all truck traffic must travel along I-880 in the Flatlands, which are predominantly low-income communities of color. This pattern of not-in-my-backyard (NIMBY) campaigns driving pollution sources to be sited in economically and politically vulnerable communities of color is well documented [*Bullard*, 1990]. Our study indicates that accelerated adoption of emission controls can help to reduce inequities in exposures to diesel-related air pollution.

Post-policy conditions suggest that while the widespread use of DPFs by 2018 improves outcomes, the accelerated adoption of 2010 and newer engines by 2023 is needed to further advance equity benefits of diesel pollution mitigation (see Table 3.5). For diesel PM and BC, the intake differentials between Brookfield-Sobrante and Sequoyah remain above 20% in 2018. The intake differentials decrease to 7% for diesel PM and 5% for BC when all trucks have 2010 and newer engines. Additional controls of diesel emissions may be necessary to further improve equity and justice in air pollution exposures, particularly for diesel NO_x. In California, incentive programs are accelerating the introduction of near-zero and zero-emission technologies for freight trucks (e.g. hybrid electric engines, electric engines, and fuel cells) [*Caltrans*, 2016; *CARB*, 2018]. These additional in-use emission control regulations and incentive programs can help further reduce diesel-related air pollution within urban freight corridors. Future work should consider whether implemented diesel truck emission controls will continue to mitigate diesel-related air pollution, given expected continued growth in heavy-duty truck freight transport [*Caltrans*, 2014; *MTC*, 2016].

The analysis presented here presumes that DPF and SCR systems will remain in good working order over the study period. Prior studies indicate that some DPF-equipped trucks emit PM and BC at high levels [*Bishop et al.*, 2013; *Preble et al.*, 2015; *Haugen et al.*, 2018; *Preble et al.*, 2018]. These high emissions result from deterioration or failure of control systems over time. The environmental equity benefits reported here represent a best-case scenario. The durability and maintenance of emission control systems will affect whether the benefits presented here are fully realized and endure over time. For communities in urban freight corridors to continue to benefit from diesel emission control efforts in future years, programs are needed to identify and repair high-emitting trucks. CARB performs ad-hoc and extremely limited smoke inspections of a subset of heavy-duty trucks [*CARB*, 2010]. The California legislature is currently considering a regular inspection and maintenance program for all operating heavy-duty trucks [*SB 210*]. In addition, adequate funding mechanisms are needed to ensure proper maintenance of emission control equipment installed on trucks. After the Motor Carrier Act of 1980 deregulated the trucking industry, short-haul drivers shifted from company employees to independent owner-operators, who bear the costs associated with owning a truck while earning low wages [*Milkman*, 2008], which sometimes makes routine maintenance unaffordable. As heavy-duty engines age, deterioration or failure of filters is likely to lead to increased emissions. Thus, sustaining the equity and justice benefits of diesel truck emission control efforts relies on the ongoing proper functioning of emission control systems and/or replacement of diesel engines with alternatives.

Important simplifications to the exposure analysis in this study include not accounting for time-activity patterns, occupational and in-transit exposures, indoor-outdoor pollution relationships, and breathing-rate variability, which will affect individual daily intake rates [*Marshall et al.*, 2006], and, therefore, estimated equity benefits. Some of these factors are correlated with income, such as residential air conditioning and building and vehicle air-exchange rates. For instance, older, leakier housing is more likely to be located in low-income communities. Taking these factors into account in future work would help to better characterize the magnitude and distribution of benefits caused by the accelerated use of diesel truck emission control technologies. Also, there are additional diesel emission sources in urban freight corridors, such as ships and rail. Further research is needed to assess how these additional sources and their associated control efforts affect inequities in exposure to diesel-related air pollution.

Chapter 4: Effects of Freeway Rerouting on Air Pollution Exposure and Neighborhood Attributes

4.1. Introduction

The Federal-Aid Highway Act of 1956 called for the construction of 41,000 miles of interstate highway by 1970 and created the Highway Trust Fund to finance it. From the mid-1950s to the early 1970s, the Interstate Highway System transformed the US urban landscapes. Urban planners saw the urban freeway as a solution to growing traffic congestion in cities, as well as a tool to achieve the urban renewal goal of "slum" clearance [*Mohl*, 2004; *Kraft-Klehm*, 2015]. Planners and engineers decided where freeways would be built with little to no citizen oversight [*Mohl*, 2008]. Freeway construction resulted in the demolition, division, and forced removal of poor communities of color, particularly African-Americans [*Mohl*, 2000; *Mohl*, 2004; *Rose and Mohl*, 2012]. Freeways facilitated white flight and accelerated white suburbanization [*Massey and Denton*, 1993; *Baum-Snow*, 2007], reinforced racial residential segregation [*Massey and Denton*, 1993; *Connerly*, 2002], and increased air and noise pollution [*Bullard*, 2004; *Mohl*, 2012], mostly in communities of color. Racial borders achieved through discriminatory race-based planning processes, such as redlining, restrictive covenants, and zoning [*Silver*, 1997; *Pulido*, 2000; *Morello-Frosch*, 2002; *Wilson et al.*, 2008; *Taylor*, 2014], were concretized into the built environment with freeway construction. The adverse effects of freeway construction are environmental justice issues [*Bullard*, 2004].

Freeway removal or rerouting is viewed as an opportunity to redress the health and environmental impacts of freeway construction [*Praetzellis et al.*, 2007; *Kraft-Klehm*, 2015]. Transforming the former freeway alignment into a landscaped boulevard increases urban green space. Green space has air pollution and health benefits [*Wolch et al.*, 2014]. However, a potential unintended consequence of efforts to expand urban green space is the green space paradox [*Wolch et al.*, 2014]. Urban green space aimed at addressing environmental injustice can make a neighborhood more desirable, potentially leading to gentrification and the displacement of the residents for whom the green space was created. This paradoxical situation has been termed environmental gentrification [*Checker*, 2011]. Studies indicate that freeway removal or tunneling can increase property values [*Tajima*, 2003; *Cervero et al.*, 2009; *Kang and Cervero*, 2009; *Kraft-Klehm*, 2015] and that conversion of the old alignment to a boulevard can lead to gentrification, as measured by changes in neighborhood racial composition [*Cervero et al.*, 2009].

In West Oakland, residents successfully advocated for rerouting the Cypress Freeway and creating a street-level boulevard along the original alignment. West Oakland, a redlined neighborhood [*Begley*, n.d.] and one of the few East Bay neighborhoods where African-Americans could own homes [*Gin and Taylor*, 2010], was targeted for and adversely affected by freeway construction. In 1958, the elevated, double-decked Cypress Freeway (I-880) was completed. It bisected West Oakland and physically segregated the neighborhood. Construction of the Cypress Freeway led to property demolitions and displaced 600 families [*Federal Highway Administration*, 1998]. The later-constructed Grove Shafter (I-980) and MacArthur (I-580) Freeways further segregated the neighborhood. Freeway construction and other urban renewal projects in West Oakland destroyed over 5,000 housing units and resulted in economic decline in the area [*Gin and Taylor*, 2010].

When the Cypress Freeway collapsed during the 1989 Loma Prieta earthquake, the California Department of Transportation (Caltrans) favored a rebuild option on the same alignment [*Praetzellis et al.*, 2007]. However, legislation such as the National Environmental Protection Act of 1969 provided the community the opportunity to participate in the decision-making process, an option that was not previously available. Community activists organized to oppose reconstruction along the original route and redress economic and environmental justice issues [*Praetzellis et al.*, 2007]. After the public comment period for the draft environmental impact statement closed in 1991, Caltrans selected an alternative route around the perimeter of West Oakland in an industrial area (Figure 4.1). Some felt the proposal did not adequately address local concerns and filed a discrimination suit under Title VI of the Civil Rights Act of 1964 [*Praetzellis et al.*, 2007]). The case, *Clean Air Alternative Coalition v United States Department of Transportation*, was settled out of court and resulted in several additional mitigation measures [*Bullard et al.*, 1997; *Praetzellis et al.*, 2007], including the transformation of the former Cypress Freeway route into a landscaped boulevard, later named Mandela Parkway. Construction of Mandela Parkway began in 2002 and was completed in 2005.

In this study, we investigate the air pollution and neighborhood impacts of rerouting the Cypress Freeway and constructing a street-level boulevard in West Oakland. Our specific objectives are to: (i) quantify the local effects on air pollution of rerouting the Cypress Freeway through modeling near-roadway concentrations for two different rebuild scenarios, and (ii) and examine neighborhood socioeconomic and demographic impacts as reflected by spatiotemporal changes in indicators of gentrification and land-use to assess whether existing residents benefit from the freeway-to-boulevard conversion or are excluded through the phenomenon of environmental gentrification.

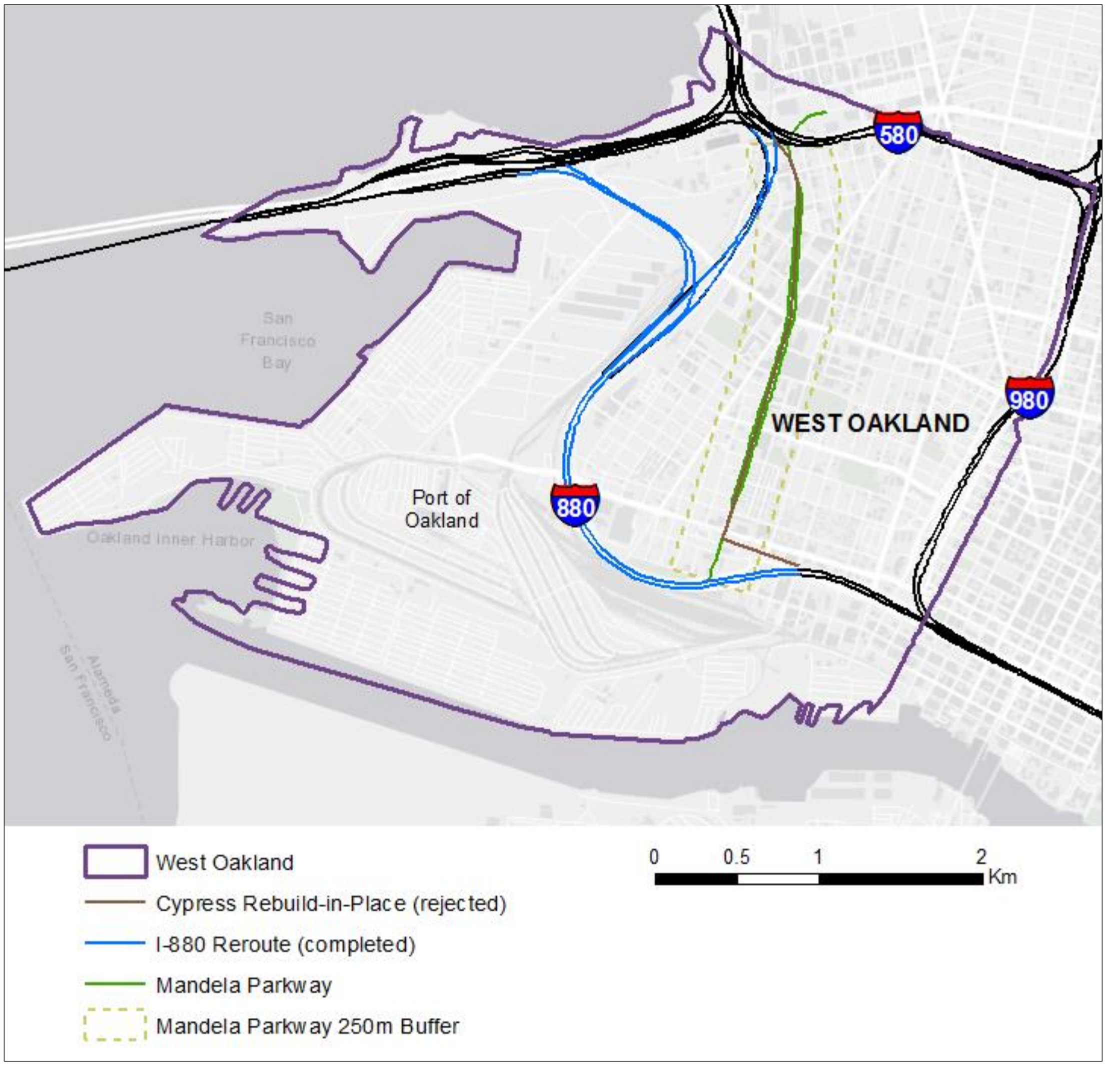

Figure 4.1. Map of West Oakland study area.

4.2. Methods

4.2.1. Exposure to Traffic-Related Air Pollution

The Cypress Freeway collapsed in 1989 and construction of the rerouted freeway and Mandela Parkway were completed in 1998 and 2005, respectively. We assessed two freeway routing scenarios: (1) rebuild-in-place: the rejected plan to reconstruct the damaged Cypress Freeway on the original alignment, which divided a residential neighborhood in West Oakland, and (2) reroute: the completed plan that reconstructed the freeway along a different route (I-880 reroute) to circle around rather than bisect a residential neighborhood in West Oakland, and replaced the damaged section of the freeway with a street-level boulevard (Mandela Parkway). Near-roadway pollutant concentrations are estimated along the Cypress Freeway rebuild-in-place and the Mandela Parkway for the year 2009. This year was selected due to the availability of traffic count data for the Mandela Parkway.

4.2.1.1. Traffic Volumes

Road network data for the I-880 reroute and Mandela Parkway were downloaded as shapefiles from the California Department of Transportation [*Caltrans*, 2017] and the *City of Oakland* [2018], respectively. Traffic count data for Mandela Parkway are from the West Oakland Truck Survey [*BAAQMD*, 2009], which provides manual truck survey counts and automatic vehicle counter data at three locations along Mandela Parkway made in August 2008. The automatic counters characterized the vehicle fleet mix, including proportions of light-duty vehicles, buses, and trucks by number of axles. This enabled us to estimate counts for other vehicle types from the manual truck counts. To align with EMFAC [*CARB*, 2017c] vehicle types, two-axle/six-tire truck counts were mapped to light-heavy and medium-heavy duty trucks as described elsewhere [*McDonald et al.*, 2014]. Trucks with three or more axles were classified as heavy-heavy duty trucks and were all assumed to be diesel-fueled, referred to as heavy-duty diesel trucks hereafter [*McDonald et al.*, 2014]. Automatic counter-derived hourly temporal profiles for each vehicle type were used to estimate hourly traffic volumes.

For the Cypress rebuild-in-place scenario, we combined measured traffic volumes for Mandela Parkway and the I-880 reroute, and we assigned all of that traffic to the original Cypress Freeway route. Traffic data for the I-880 reroute were obtained from the California Department of Transportation. Available data include link-specific counts for total vehicle annual average daily traffic (AADT), proportion of trucks, and truck counts broken down by number of axles. We estimated light-duty vehicle counts by subtracting trucks from total vehicle counts. Two-axle/six-tire trucks and trucks with three or more axles were apportioned as described above. Traffic volumes were mapped to hourly estimates using month-of-year, day-of-week, and hour-of-day temporal allocation factors from *McDonald et al.* [2014].

4.2.1.2. Vehicle Emissions

Estimates of link-specific emission rates were calculated from hourly traffic volumes and emission factors. NO_x and $PM_{2.5}$ emission factors by vehicle type were estimated using EMFAC model outputs at the county level. For Mandela Parkway, emission factors were defined using estimates for running exhaust emissions, with an average speed of 30 mph, which was the average vehicle speed indicated by the automatic traffic counter [*BAAQMD*, 2009]. For the freeway, emission factors were calculated for aggregated vehicle speeds. We estimated BC emission factors using the EMFAC-derived $PM_{2.5}$ emission factors, combined with gasoline and diesel BC fractions of 18% and 72%, respectively [*Ban-Weiss*, 2008; *Dallmann et al.*, 2013; *Dallmann et al.*, 2014].

4.2.1.3. Near-Roadway Air Pollutant Concentrations

We predicted traffic-related air pollutant concentrations using the RLINE line-source dispersion model [*Synder et al.*, 2013; *Venkatram et al.*, 2013]. We previously evaluated RLINE model performance in predicting NO_x and BC concentrations at near-roadway monitoring sites in the San Francisco Bay Area [*Patterson and Harley*, 2019]. The study domain was overlaid with a 50 m grid, and model receptors were set at grid centroids. We modeled concentrations within 250 m of Mandela Parkway and the I-880 reroute, since traffic-related air pollution levels are known to be elevated at distances of about 200 m of major roadways [*Brugge et al.*, 2007; *Health Effects Institute*, 2010; *Zhu et al.*, 2002]. The meteorological inputs required for RLINE dispersion calculations were computed using AERMET [*Cimorelli et al.*, 2005]. Meteorological data were obtained from the National Weather Service for the nearby Oakland International Airport. We ran RLINE using a unit emission rate (1 g m^{-1} s^{-1}) at release heights of 0.3 m for light-duty vehicles and 4 m for heavy-duty trucks [*Bishop et al.*, 2001]. Dispersion model results were combined with hourly emissions estimates to compute emission-weighted NO_x and BC concentrations.

Total near-road pollutant concentrations were calculated as the sum of modeled traffic-related and urban background concentrations. Ambient observations at the Bay Area Air Quality Management District (BAAQMD) monitor at West Oakland were used to estimate NO_x and BC background levels for this study. To reduce the influence of local NO_x emission sources at the background site [*BAAQMD*, 2015] and characterize the urban background contribution accurately, we defined background NO_x concentrations using the 25th percentile method of the West Oakland monitoring site data [*Van Poppel et al.*, 2013]. Measured BC concentrations were considered representative of urban background concentrations for this pollutant and were used without adjustment. Background concentrations were added to dispersion model estimates of traffic-related air pollution for each modeled receptor within the study domain. Predicted hourly concentrations were then used to compute annual average concentrations at each receptor.

4.2.2. Neighborhood-Scale Changes in Demographics and Land Use

4.2.2.1. Census Data

Block-group level data from the 1990 Census [*U.S. Bureau of the Census*, 1992] and the 2006-2010 American Community Survey [*U.S. Bureau of the Census*, 2010] were used to investigate impacts of the freeway rerouting and conversion of the old alignment to a street-level boulevard on neighborhood demographic, socioeconomic, and housing characteristics. Demographic variables include the total population and the percentage of Black, Latino, and nonwhite (i.e., non-(white non-Hispanic)) residents. Socioeconomic indicators include the median household income, percentage of residents with at least a bachelor’s degree, and percentage of residents living in poverty. Housing characteristics include median rent, median home value, and percentage of renter-occupied dwellings.

We compared 1990 and 2010 census variables within 250 m of the Mandela Parkway alignment to corresponding values for all of West Oakland (Figure 4.1). We used an area-weighting method to estimate the demographic, socioeconomic, and housing composition within the 250 m band for each year. A 250 m buffer was intersected with census block group areas using a Geographic Information System (GIS). The percentage of each block group’s area within the buffer was computed, and raw census data were weighted using these percentages.

4.2.2.2. Land Use Data

We examine current land use attributes and land use projections in order to gain qualitative insights into how rerouting the freeway and replacing the old alignment with a street-level boulevard influenced land use changes. Land use changes may offer possible explanations for observed neighborhood change. Data on existing and future land use in West Oakland were obtained from the *City of Oakland* [2014, 2016].

4.3. Results and Discussion

4.3.1. Spatial Distribution of NO_x and BC Concentrations

Figure 4.2 illustrates the impact of the Cypress Freeway rebuild-in-place and reroute scenarios on pollutant concentrations in the middle of West Oakland. The maps in Figure 4.2 substantiate that by rerouting the Cypress Freeway, high concentrations shift from the middle of West Oakland (Figures 4.2a and 4.2c) to around the periphery (Figures 4.2b and 4.2d). Large pollutant reductions are observed in the middle of West Oakland. For the rebuild-in-place scenario (Figures 4.2a and 4.2c), mean annual average NO_x and BC concentrations are 36.1±1.2 ppb and 1.72±0.30 μg m^{-3}, respectively. For the reroute scenario (Figures 4.2b and 4.2d), mean annual average concentrations are 22.3±0.8 ppb for NO_x and 1.29±0.03 μg m^{-3} for BC.

The estimated pollution reductions highlight the significance of roadway type routed through residential areas. Concentrations are higher on the Cypress Freeway rebuild-in-place compared to the Mandela Parkway, as indicated by the visibility of the freeway alignment in Figures 4.2a and 4.2c. Traffic volumes on the Mandela Parkway are substantially lower than those on the Cypress Freeway rebuild-in-place. Annual average daily traffic volumes on the Cypress Freeway rebuild-in-place range from approximately 108,000 to 126,000. Heavy-duty diesel trucks account for 7.3% of traffic. In contrast, the Mandela Parkway has average traffic volumes of approximately 500 to 3,300 vehicles per day, with heavy-duty diesel trucks accounting for < 5% of total traffic. These results provide quantitative support for the expectation that reducing the traffic-carrying capacity through residential areas provides air quality benefits. Replacing a freeway with a boulevard is one transportation policy that reduces the traffic burden in residential areas, particularly of heavy-duty diesel trucks. Other such policies include revising designated truck routes [*Gonzalez et al.*, 2011].

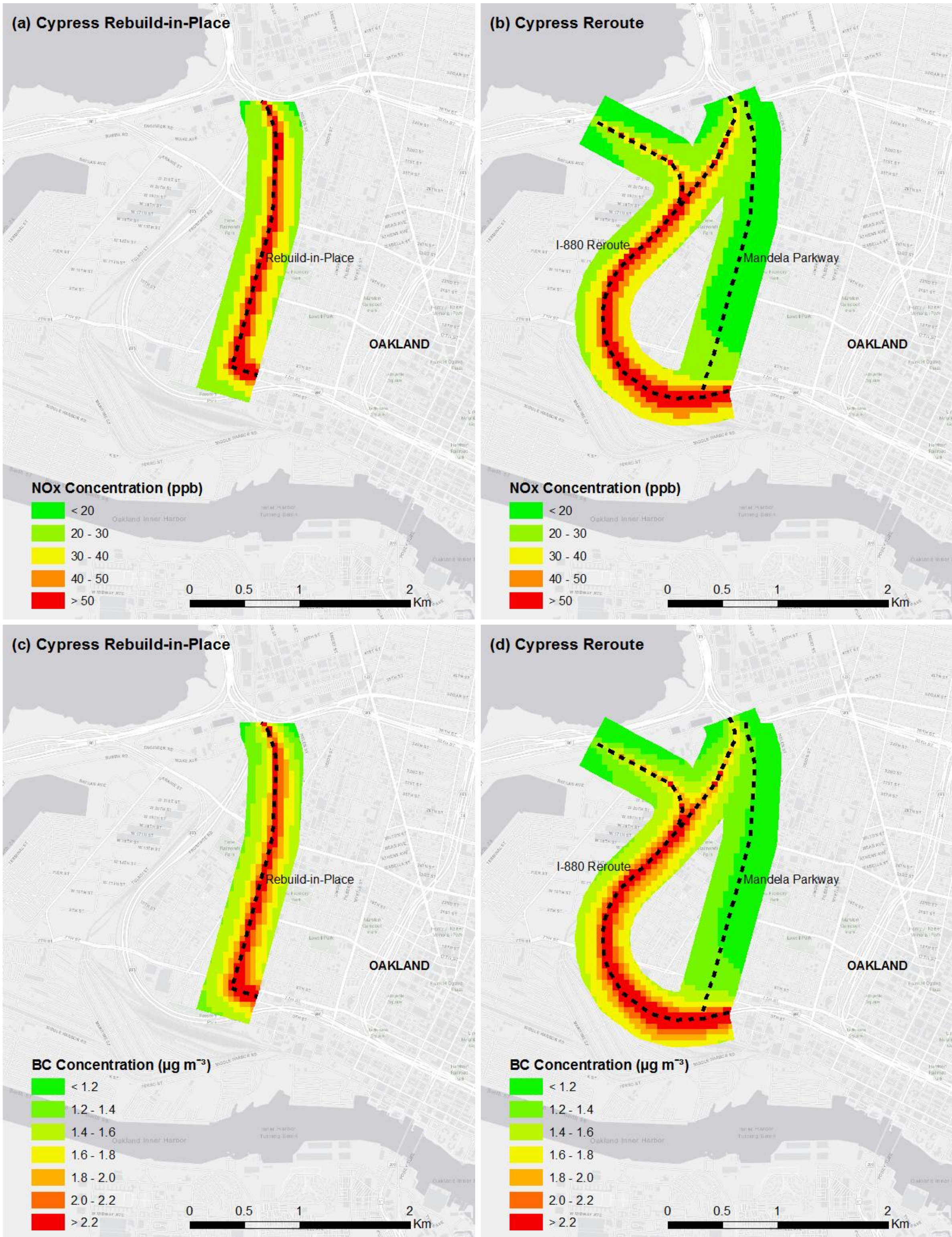

Figure 4.2. Annual average concentrations in West Oakland for (a, c) the Cypress rebuild-in-place scenario, and (b, d) the Cypress reroute scenario. Dashed black lines represent the centerline of each route.

On average, reductions are larger for mean annual average NO_x concentrations (-38±4%) than for mean annual average BC concentrations (-25±2%) in the middle of West Oakland. This may be due to the influence of the I-880 reroute on air pollution near the Mandela Parkway, particularly the intersecting segment at the south end, which is evident when comparing the spatial pattern in annual average concentrations shown in Figures 4.2b and 4.2d versus Figures 4.2a and 4.2c. A stronger influence on BC than NO_x is visible in Figures 4.2b and 4.2d, and suggests that BC decays less rapidly than NO_x, which is consistent with previous studies [*Karner et al.*, 2010]. These results reveal that ideally, alternative freeway corridors would not have segments in close proximity to the impacted area of concern. The options for an alternative route were limited in West Oakland due to existing freeways, port, and railroads.

4.3.2. Distance-Decay Curves

Figure 4.3 shows average annual concentrations of NO_x and BC at increasing distances from the Mandela Parkway alignment. Each data point in the figure represents an averaged value for all model receptors located within 25 m distance bands. For the Cypress rebuild-in-place scenario, we observe much higher pollutant concentrations to the east at all distances. Mean annual average concentrations of NO_x and BC for a distance range of 0 to 25 m east of the freeway are 71.2±6.0 ppb and 2.6±0.1 $\mu g\ m^{-3}$, respectively. In contrast, mean annual average concentrations for distances of 0 to 25 m west of the freeway are 55.6±6.0 ppb for NO_x and 2.2±0.2 $\mu g\ m^{-3}$ for BC. The concentrations decrease noticeably when moving away from the freeway in both directions, following distance-decay relationships similar to those observed in other studies [*Karner et al.*, 2010; *Zhu et al.*, 2002]. Pollutants decay at a similar rate on both sides of the Cypress Freeway, with mean NO_x concentrations decreasing by 49±9% in the first 150 m east of the freeway and 50±11% in the first 150 m to the west. BC concentrations decrease by 32±6% and 33±7% in the eastward and westward directions, respectively. The more rapid decline in NO_x concentrations relative to the roadway edge is consistent with results from previous studies [*Karner et al.*, 2010].

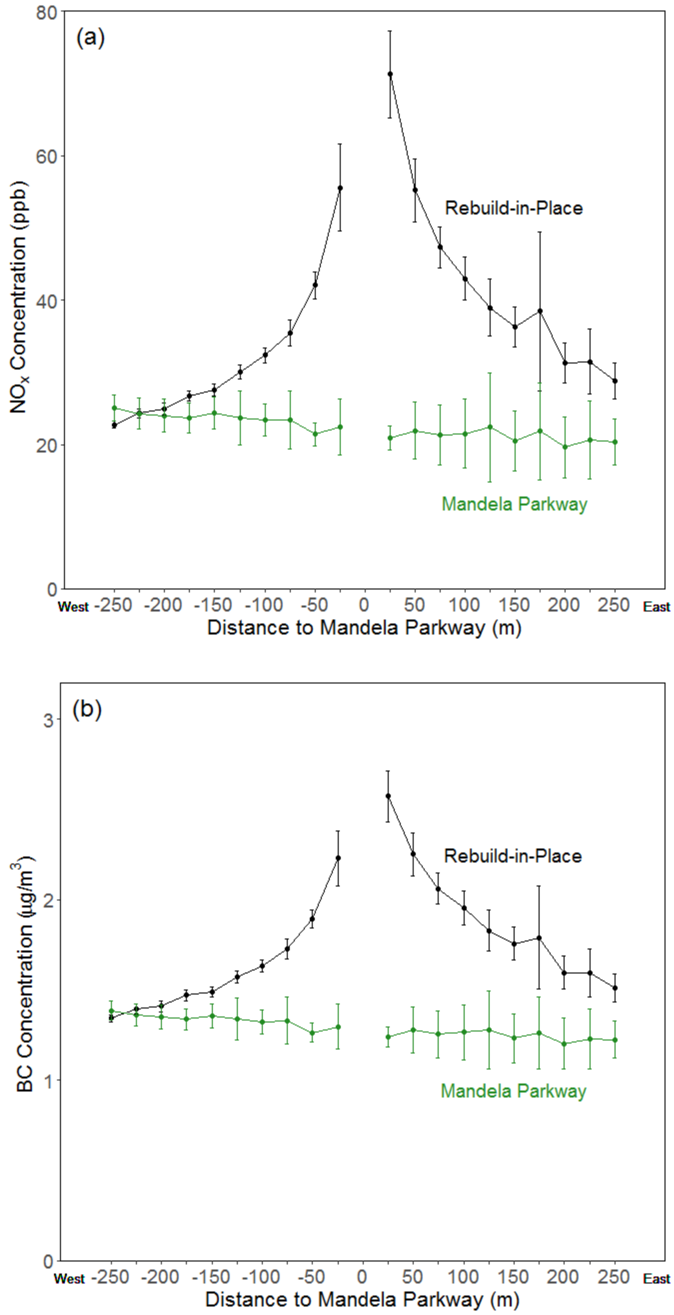

Figure 4.3. Mean annual average concentrations of (a) NO_x and (b) BC versus distance from the Mandela Parkway alignment. Each data point in the figure represents an averaged value for all measurements within a 25 m distance band. Negative distances are to the west; positive values are to the east. Uncertainty estimates indicate 95% confidence levels.

Figure 4.3 further demonstrates the large concentration reductions observed in Figure 4.2 that result from rerouting the Cypress Freeway. Mean annual average pollutant concentrations within 25 m of the Mandela Parkway alignment decrease by 66±8% for NO_x and 48±5% for BC as a result of rerouting. Figure 4.3 shows that concentrations are higher on the west side of Mandela Parkway than to the east. In general, locations to the west of Mandela Parkway are in closer proximity to the I-880 reroute. We do not observe linear decreases in concentrations from east to west of Mandela Parkway because of the varying distances separating it from the I-880 reroute, including a freeway segment that intersects with Mandela Parkway at its south end. Overall, this analysis quantitatively substantiates claims that freeway rerouting reduces the air quality burden on residents.

4.3.3. Neighborhood Measures

4.3.3.1. Population Density

Residents advocated for the replacement of the original Cypress Freeway alignment with a boulevard and relocation of the freeway to more industrial areas of West Oakland, instead of rebuilding along the original alignment that ran through the residential areas. As shown in Figure 4.4a, the population density in West Oakland is much more concentrated around Mandela Parkway than the I-880 reroute, indicating that on average, residents experience air quality benefits resulting from freeway rerouting (Figures 4.2 and 4.3). Figure 4.4b shows a map of land use zoning designations for West Oakland. Residential use and industry, commercial, and truck-related uses account for approximately 60 and 23% of the land area in West Oakland, respectively [*City of Oakland*, 2014]. One primary area of residential use is along the southern portion of Mandela Parkway, which corresponds to the area with the highest population density (Figure 4.4a), while industrial uses are concentrated along the northern portion of Mandela Parkway. This difference in land use explains the difference in population density between the two sections of Mandela Parkway.

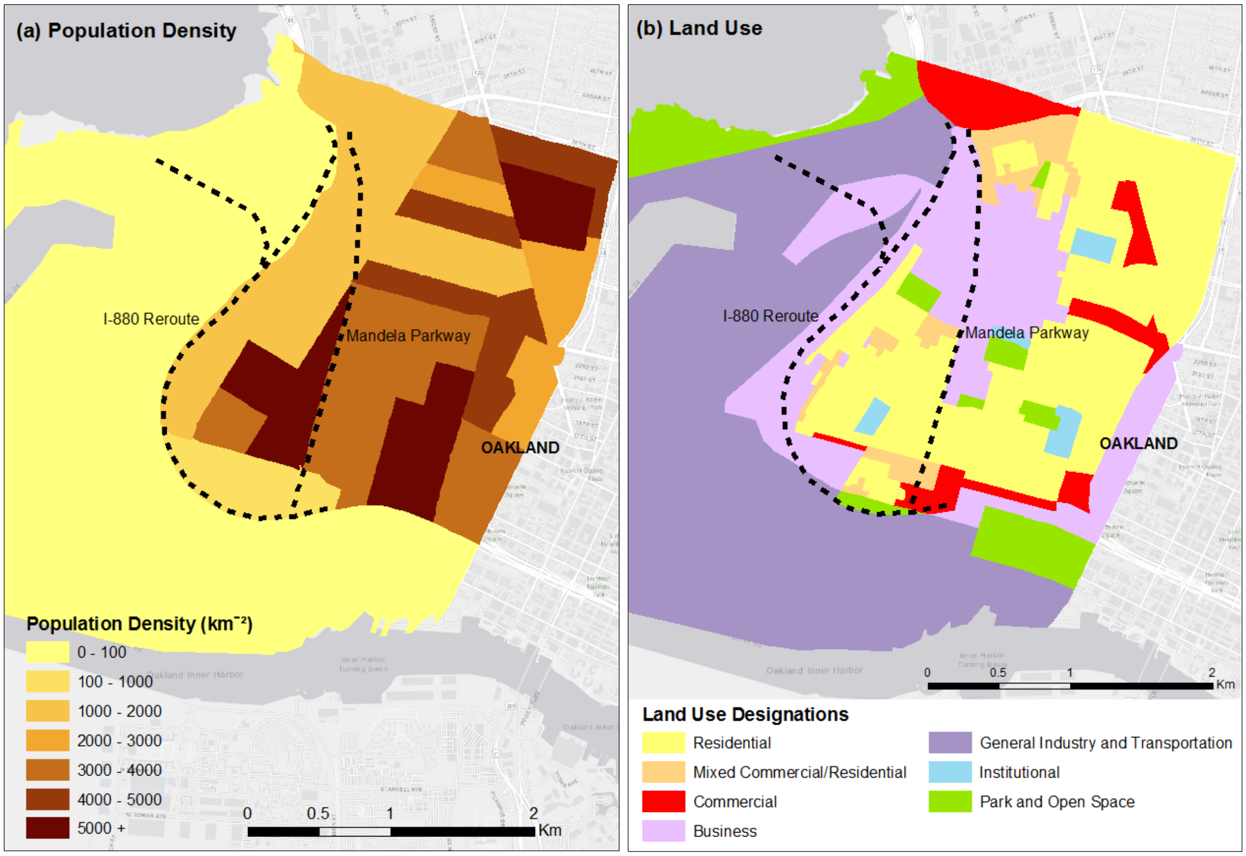

Figure 4.4. (a) Population density (people per km^2) in West Oakland in 2010. (b) Current West Oakland zoning land-use designations. Dashed black lines represent the centerlines of the I-880 Reroute (left) and Mandela Parkway (right).

4.3.3.2. Neighborhood Change

Changes in West Oakland demographic, socioeconomic, and housing indicators from 1990 to 2010 are apparent in Table 4.1. Between 1990 and 2010, there was a decrease in the nonwhite, low-income population in West Oakland, with substantial decreases in the Black population (-30%). Reductions in the Black population may be due to increases in the cost of housing. As shown in Table 4.1, the median home value in West Oakland increased 124% and median gross rent increased 21% between 1990 and 2010. The area along the Mandela Parkway has seen an even larger decrease in the Black population (-37%), where the median home value has increased 229% and median gross rent increased 27% along the Mandela Parkway. Larger property value and rent increases along the Mandela Parkway compared to West Oakland as a whole may be due to the freeway rerouting and conversion to the street-level boulevard, which would be consistent with previous studies [*Tajima*, 2003; *Cervero et al.*, 2009; *Kang and Cervero*, 2009; *Kraft-Klehm*, 2015]. In order to support this evidence of environmentally-driven neighborhood change, further analysis using a hedonic pricing model is needed to empirically attribute the larger rise in property values and rental prices to the street-level boulevard. Table 4.1 also indicates that during this same period, there was a large increase in the Latino population. The percentage change of nonwhite residents can ignore such an inter-ethnic shift [*Pastor et al.*, 2001], which is important to consider when investigating environmental benefits for existing residents.

Table 4.1. Change in racial and socioeconomic composition between 1990 and 2010 within 250 m of the Mandela Parkway alignment and for all of West Oakland.

	250 m of Mandela Parkway			West Oakland		
	1990	2010	Δ%	1990	2010	Δ%
Total Population	2742	3725	35.8	23312	21767	-6.6
% Nonwhite	95.9	88.0	-8.2	94.4	86.4	-8.5
% Non-Hispanic Black	75.8	47.4	-37.4	77.0	53.8	-30.1
% Latino (Hispanic)	13.0	26.0	100	8.0	17.4	119
Mean Median Household Income[a]	$31896	$36717	15.1	$27863	$35439	27.2
% Poverty[b]	77.1	60.6	-21.3	69.4	56.8	-18.2
% Renter Occupied	80.1	77.7	-3.1	79.1	75.5	-4.6
Mean Median Gross Rent	$746	$944	26.6	$736	$890	20.9
Mean Median Home Value	$133049	$438095	229	$172633	$386206	124
% College Educated[c]	7.2	19.1	165	8.8	20.1	129

[a] In 2011 inflation-adjusted dollars
[b] Percent of households with income less than twice the poverty level
[c] Includes college and advanced degrees

Table 4.1 suggests that gentrification occurred throughout West Oakland, including the area within 250 m of Mandela Parkway, between 1990 and 2010. This is reflected by decreases in percent of nonwhite residents, percent of residents living in poverty, and percent renter-occupied units, along with growth in median household income, median rent, median home value, and percent of college educated. The racial and socioeconomic changes along the Mandela Parkway occurred without land use changes over the same period. Land use projections in the West Oakland Specific Plan also show continued industrial and mixed-use zoning designations along the northern portion of the Mandela Parkway [*City of Oakland*, 2014]. A case study of the Greenpoint neighborhood in Brooklyn found that retaining industrial zoning after environmental cleanup and green space creation helps prevent residential development that drives out long-term working-class residents [*Curran and Hamilton*, 2017]. In the present case, maintaining industrial land uses did not stop market-based processes of neighborhood change.

The area along the Mandela Parkway had a smaller increase in median household income (15% versus 27%) and maintains higher percentages of residents living in poverty (61% versus 57%) than West Oakland as a whole. This finding implies that certain attributes of this area may enable low-income residents to reside along Mandela Parkway. One potential factor is the presence of affordable housing. There are three affordable housing sites in the southern portion of Mandela Parkway [*City of Oakland*, 2014], where the residential population along Mandela Parkway is concentrated (Figure 4.4a).

4.4. Conclusions

West Oakland residents saw the rerouting and replacement of the Cypress Freeway as an opportunity to redress past social and environmental justice harms that urban freeway construction had caused on their neighborhood. Our air pollution maps and distance-decay curves reveal that rerouting the Cypress Freeway resulted in substantial reductions in annual average NO_X and BC concentrations in the middle of West Oakland when compared to the Cypress Freeway rebuild-in-place scenario. These air quality benefits highlight the importance of roadway type planned through residential neighborhoods, such as freeways and designated truck routes. We observe that the new freeway route still impacts air pollutant levels in the Mandela Parkway corridor, so it is critical to select an alternative route that does not have segments in close proximity to residential areas. Limitations may be present in port communities and communities impacted by goods movement activities.

Environmental justice activism sometimes has unintended paradoxical consequences, where efforts to improve a neighborhood make existing residents vulnerable to displacement [*Checker*, 2011]. This displacement is facilitated through economic revitalization efforts that do not prioritize the needs of existing residents. While the urban freeway was thought of as a tool for urban revitalization by mid-century transportation planners, the removal and rerouting of the urban freeway are viewed as opportunities for redevelopment. An investigation of West Oakland indicates that freeway rerouting results in some environmental gentrification, with property value increases and the displacement of long-time Black residents, similar to freeway removal and tunneling. To ensure existing residents benefit from freeway rerouting and construction of a street-level boulevard, affordable housing and other anti-displacement strategies, such as inclusionary zoning and renter protections, should be instituted [*California Environmental Justice Alliance*, 2017].

There are some limitations in our analysis methods. Traffic counts on the Mandela Parkway were based on a short-duration traffic survey. These counts were extrapolated to annual counts using temporal allocation factors from *McDonald et al.* [2014] that were derived from freeway traffic count data. Although traffic activity profiles can vary by roadway type [*Lindhjem and Shepard*, 2007], data on arterial traffic patterns were not available. More extensive traffic count data on local arterials is needed to improve estimates of the air quality impacts of freeway rerouting. Additionally, our analysis of neighborhood change was restricted to the block group level due to available census data. Using an area-weighting method to estimate demographic and socioeconomic variables along the Mandela Parkway introduces error. As freeway removal receives increased attention as an urban transportation policy, it is critical to accurately determine who benefits from such projects.

Chapter 5: Conclusions

5.1 Summary of Major Findings

The goal of this dissertation was to evaluate the environmental justice and equity outcomes of policies to control and reroute diesel truck traffic and associated emissions. This research began with an evaluation of capabilities for modeling near-roadway air pollution, with a novel focus on diesel-related pollutants and featuring evaluation against yearlong data records from new near-roadway monitoring sites. The body of work is an original contribution because it assesses newly implemented diesel emission control regulations and the effects of real-world freeway rerouting decisions on air quality and environmental justice. My research provides systematic approaches for incorporating an equity lens within analyses of goods-movement related policies. Insights from my dissertation research can serve to better inform the design and implementation of emission control regulations and urban transportation planning to advance environmental equity and justice.

In Chapter 2, I evaluated a near-roadway dispersion model for prediction of diesel-related air pollutants, using continuous data from new near-roadway monitoring sites in the San Francisco Bay Area for evaluation purposes. Temporal variations in emissions and concentrations were also characterized.

- Heavy-duty diesel trucks were the dominant on-road source of NO_x and BC emissions at both near-roadway monitoring sites, with diesel contributions varying by pollutant and monitoring site. Heavy-duty diesel truck contributions were greater for BC emissions than for NO_x.

- The model generally captures the diurnal trends of the observations. Model predictions are in better agreement with the concentration profiles of the less diesel-dominated site. Improvements in the characterization of diesel truck traffic at heavily diesel-dominated sites and reducing uncertainties in BC emission factors may help to improve model performance.

- More than 90% of predicted daily average concentrations were within a factor of two of observations at both near-road monitoring sites. The correlation between model predictions and observations is better for BC (R = 0.87-0.88) than for NO_x (R = 0.61-0.69).

- The model responds appropriately to seasonal variations in meteorology and day-of-week variations in emissions. The model accurately captures higher NO_x and BC concentrations that are observed during the fall and winter seasons. The model accurately reproduces observed weekday/weekend differences in pollutant concentrations, which peak during the middle of the week and decrease on the weekends.

In Chapter 3, the effects of the accelerated use of diesel particle filter (DPF) and selective catalytic reduction (SCR) systems on diesel-related emissions, concentrations, and metrics of environmental equity were assessed. Estimates were made for two neighborhoods in the East

Oakland freight corridor differentially impacted by truck traffic: Brookfield-Sobrante, along I-880 with heavy truck traffic, and Sequoyah, along a segment of I-580 where heavy-duty trucks are prohibited.

- Heavy-duty diesel truck emissions are much higher on I-880 than on I-580. Heavy-duty diesel trucks are the dominant source of NO_x, diesel PM, and BC emissions on I-880, while light-duty vehicles are the dominant source of these pollutants on I-580. Between the baseline year (2009) and the post-policy years (2018 and 2023), there were large decreases in heavy-duty diesel truck emissions. Emission reductions outpaced the effect of growth in diesel truck traffic.

- Reductions in near-roadway pollutant concentrations were larger in Brookfield-Sobrante than in Sequoyah for both post-policy years. After universal adoption of DPFs (2018), reductions in Brookfield-Sobrante were 63±3% for NO_x and 48-49±4% for diesel PM and BC. In Sequoyah, reductions in NO_x concentrations were smaller (52±3%), and diesel PM and BC concentrations increased by 19±7 and 15±6%, respectively. By 2023, when all trucks are required to be equipped with 2010 or newer engines, NO_x concentrations remain higher in Brookfield-Sobrante, but diesel PM and BC concentrations will be similar in both neighborhoods.

- Reductions in diesel emissions also led to improvements in environmental equity. Decreases in the relative percent difference for all pollutants indicate a more even distribution of air pollution between Brookfield-Sobrante and Sequoyah. The accelerated adoption of 2010 and newer engines by 2023 further advances equity benefits of diesel pollution mitigation, reducing the diesel PM and BC intake differentials between Brookfield-Sobrante and Sequoyah from above 20% to 7% for diesel PM and 5% for BC.

- The environmental equity benefits reported here depend on the durability and maintenance of emission control systems. Programs should identify and repair high-emitting trucks and provide adequate funding for proper maintenance of emission control equipment installed on trucks.

In Chapter 4, the impacts of rerouting a freeway (Cypress Freeway) and replacing the old alignment with a street-level boulevard (Mandela Parkway) on near-roadway concentrations of diesel-related air pollutants are modeled. Estimates are compared for rebuild-in-place and reroute scenarios along the Mandela Parkway alignment in the middle of West Oakland. Demographic, socioeconomic, and land-use variables are assessed before and after freeway rerouting and construction of the boulevard.

- Freeway rerouting and replacement with a boulevard results in large pollutant reductions in the middle of West Oakland. Near-roadway annual average concentrations are reduced by 38±4% for NO_x and 25±2% for BC. Reduced traffic-carrying capacity on Mandela Parkway contributes to the estimated air quality benefits.

- Rerouting the freeway has led to shifts in the air pollution burden to the periphery of West Oakland, a less populated area that is zoned for commercial or industrial rather than residential use. Modeled spatial patterns of air pollution reveal that the I-880 reroute still influences pollution levels in the middle of West Oakland due to the close proximity to some freeway segments.

- Between 1990 and 2010, West Oakland saw a decrease in its nonwhite, low-income population. During this period, there were larger decreases in the long-time Black population (–37%) and increases in property values (229%) along the Mandela Parkway compared to West Oakland as a whole, providing evidence of environmental gentrification along Mandela Parkway.

- Between 1990 and 2010, area along the Mandela Parkway had a smaller increase in median household income (15% versus 27%) and maintains higher percentages of residents living in poverty (61% versus 57%) relative to West Oakland as a whole. The presence of affordable housing may be one explanatory factor.

5.2 Recommendations for Future Research

5.2.1. Developing Longitudinal Community-Based Participatory Traffic Count Surveys

In Chapters 2-3, near-roadway model predictions along freeways were presented. Traffic activity data are needed to make near-road air pollution estimates. Currently, annual average daily traffic (AADT) counts are limited to freeways and some major arterials and weigh-in-motion stations capture continuous traffic count data at select freeway locations. Traffic count data on arterial roadways are not sampled or archived in as comprehensive a manner as the freeway network. Vehicle mix and temporal patterns vary on different road types [*Batterman et al.*, 2015b]. In California, ~ 60% of total vehicle traffic on urban arterials occurs without traffic counts [*McDonald et al.*, 2014]. This gap inhibits the prediction of near-roadway pollutant concentrations on local roads, which is necessary for higher resolution emissions mapping and for improved assessments of local-scale traffic-related air pollution.

To improve the characterization of traffic patterns on arterials, new data sources are needed. In Chapter 4, model predictions along Mandela Parkway were possible due to manual survey and co-located automatic diurnal data [*BAAQMD*, 2009]. The community-based participatory research (CBPR) partnership between West Oakland Environmental Indicators Project, the Bay Area Air Quality Management District, and Sonoma Technology, Inc demonstrates how community-based organizations can participate in collecting traffic count data on arterials. Periodic manual count surveys or developing a network of automatic counters are necessary to characterize traffic activity on arterials and then accurately model air quality in communities. Community-based organizations and researchers can collaborate to conduct the manual surveys or identify the locations of automatic counters to help study a community-identified traffic issue. Some possible issues to study include the air quality impacts of designated truck routes, increased truck traffic due to facility (e.g., warehouse) siting or expansion, or roadway construction or removal. Periodic measurements allow for the longitudinal study of air quality impacts. For instance, new count data on Mandela Parkway would enable the comparison of current against past air quality conditions, as well as model evaluation using newly available measurements of air pollution levels in West Oakland [*Apte et al.*, 2017].

5.2.2. Evaluating Model Predictions of Air Pollution Using New Local-Level Monitoring Techniques

In Chapter 2, near-roadway model predictions were evaluated against new near-road monitoring sites. These sites satisfy the US EPA's recently adopted requirement to conduct near-road monitoring of air pollution. However, monitoring has largely relied on a sparse network of stationary air quality monitors that capture regional air quality trends. The existing network fails to capture neighborhood level exposures, particularly in areas that host major emission sources, such as major roadways. This is particularly true for primary pollutants, such as black carbon, which exhibit high intra-urban variations in concentrations [*Apte et al.*, 2017]. Addressing this gap is necessary to validate local-level model predictions, such as those presented in Chapters 3-4. This step is critical for tracking and verifying predicted air quality impacts of diesel emission control regulations and land use decisions in highly impacted areas.

New monitoring techniques have helped address spatial limitations of the current network. These methods include: mobile monitoring with fleet vehicles, and deployment of low-cost sensor technologies. *Apte et al.* [2017] equipped Google Street View vehicles with air pollution instruments and repeatedly sampled air pollution on streets in West Oakland for one year. The street-level air quality data from this mobile monitoring approach can be used to validate air pollution model predictions. Another measurement approach is deploying high-density, low-cost air quality sensor networks. Expanding the regulatory air monitoring network using traditional monitoring technologies is cost prohibitive. Advancements in low-cost sensor technologies has resulted in deployment of reliable low-cost sensor networks that measure a variety of pollutants, including NO and NO_2 [*Mead et al.*, 2013], PM [*Williams et al.*, 2014], and BC [*Caubel et al.*, 2018]. *Caubel et al.* [2019] deployed 100 low-cost BC sensors across 100 locations in West Oakland for 100 days. Widespread use of such dense low-cost sensor networks produces fine spatial and temporal scale measurement data that can also be used to evaluate and improve local-scale predictive capabilities of air pollution models.

5.2.3. Assessing the Relationship Between Diesel Policy-Related Changes in Air Pollution and Associated Health Outcomes

It is important to understand the effects of diesel-related policies on the health of residents who live in close proximity to roads, particularly in busy urban transportation and goods movement corridors. In Chapters 2 and 3, pollutant concentrations were modeled before and after implementation of diesel emission control and truck routing policies at near-roadway census block and grid centroids. Highly resolved spatiotemporal near-roadway model predictions can be used to estimate traffic-related exposures for epidemiological studies [*Batterman et al.*, 2014b]. Exposure estimates are then combined with longitudinal health data to assess the relationship between changes in exposure and health outcomes. Daily average concentrations are relevant for acute health measures, such as asthma exacerbation. Monthly and annual average concentrations are relevant for long-term health effects, such as birth outcomes and cancer risk. Epidemiological studies are needed to investigate whether diesel-related policies are effective in improving community health. The work described in Chapter 3 provides a foundation for further work to evaluate the health effects of reductions in exposure to diesel exhaust for residents who live along I-880, using results for residents who live along the segment of I-580 where heavy-duty trucks are prohibited as a control group.

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