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Toward Community-Scale Modeling of Air Pollution:
Transportation Equity Applications and Updated Ship Emission Factors

By

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DISSERTATION

Submitted in partial satisfaction of the requirements for the degree of

DOCTOR OF PHILOSOPHY

in

Civil & Environmental Engineering

in the

OFFICE OF GRADUATE STUDIES

of the

UNIVERSITY OF CALIFORNIA

DAVIS

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2021

ABSTRACT

Air pollution emitted by transportation sources such as cars, trucks, buses, trains, and ships negatively impacts human health, and the exposure burden is not shared evenly across populations. Therefore, working towards environmental justice, or the equitable distribution of environmental risks and benefits, requires accurate modeling of air pollution at the community scale. High-resolution air pollution models can help identify local pollution hotspots within cities, assess environmental health risks of residential developments near polluting transportation corridors, and predict or evaluate the impact of regulatory interventions or other pollution mitigation measures. The studies presented in this dissertation broadly relate to how air pollution emissions from transportation sources can contribute to local-scale heterogeneities in pollutant concentrations and resulting inequities in environmental health impacts, with a focus on how improved community-scale modeling can help assess these inequities. Chapter 1 describes how near-source air pollution impacts have historically been approximated by the distance from the source and argues that this approach should be replaced with a more nuanced modeling approach to better capture local-scale concentration gradients, particularly when evaluating the impact of transportation emissions on new residential developments. Some of the challenges and gaps in current modeling approaches are elaborated, including the need for accurate and spatiotemporally specific emission factors.

To mitigate inequities in transportation planning and funding allocations, some metropolitan planning organizations (MPOs) have adopted equity analysis frameworks for their regional transportation plans (RTPs). Air pollution exposure related to transportation can exacerbate other transportation-related inequities, but in most equity analyses it is only addressed superficially, if at all. Chapter 2 assesses how three Californian MPOs consider traffic-related air pollution (TRAP) burden in equity analyses of RTPs. Patterns of exurbanization and gentrification

have, in some regions, displaced communities outside of urban core public transit networks, and therefore excluded them from the benefits of reduced local vehicle miles traveled (VMT). In these exurban communities, investments tend to be skewed towards freeway expansion and improvement rather than public transportation. An analysis of the RTPs of three Californian MPOs shows that current equity metrics do not adequately capture the historical and current burdens on people in outer suburban and exurban communities. One notable way that MPOs miss the inequities experienced by communities outside the urban core is by assessing compliance with air pollution standards and VMT reduction targets solely at the regional level, despite significant local variation in dimensions of transportation equity such as TRAP exposure and long car commutes. The MPOs that did consider more local variations approximated TRAP exposure solely by distance from a roadway. However, it is argued in this chapter that this is not sufficient for comparing urban core communities living near transit hubs to exurban communities living near commuter freeways. This study suggests that equity analyses of transportation projects would benefit from improved methods for modeling TRAP exposure at the community scale.

Ships contribute substantially to air pollution in port communities, but most modeling studies in these areas use emission inventories developed from decades-old measurements of ship engines. Chapter 3 updates brake-specific particulate matter (PM) mass emission factor (EF_{PM}) estimates from ships via a meta-analysis of 55 lab and field studies. This meta-analysis incorporates newer studies that capture changing fuel standards and incorporates more measurements of operating modes and ship types associated with near-shore emissions that contribute to urban air pollution in port communities. The primary influence examined is the dependence of EF_{PM} on fuel sulfur content (FSC), which has been targeted by the International Maritime Organization as well as local jurisdictions in efforts to reduce PM emissions from

shipping. The results support the assumption that limiting FSC has the potential to significantly reduce EF_{PM} and demonstrate a statistically significant linear dependence of EF_{PM} on FSC, although the incorporation of new studies weakens the relationship relative to an earlier meta-analysis. Across the included studies, switching from a low-grade, high sulfur fuel to a high-grade, low sulfur fuel reduces EF_{PM} by 50-80%. The updated emission factors for high sulfur fuels and low sulfur fuels are not significantly different than those derived from earlier studies and corroborate the literature. Some studies of EF_{PM} and engine load show increased EF_{PM} at very low engine loads relative to medium engine loads, likely due to reduced combustion efficiency. The greater heterogeneity of EF_{PM} as compared to emission factors for gaseous pollutants suggests the need for additional studies to investigate the influence of individual factors. Additionally, more studies of ship types (such as tugs and pilots) and operating conditions (such as low speeds and idling) typical of near-shore ship operation would better support high-resolution modeling of ship contributions to PM pollution in port communities. The results of this chapter provide evidence to validate policy interventions that lower the maximum allowable FSC to mitigate PM emissions from shipping, corroborate EF_{PM} estimates used in studies based on previous measurements, and identify areas where further measurements are warranted.

Dedication

To my Oma, Rosemarie Heyer, and my Mormor, Siv Alhorn.

Acknowledgments

First, I would like to thank my advisor, Christopher Cappa for his support and guidance. I would also like to thank Deb Niemeier and Holly Oldroyd for their valuable input as my dissertation committee and Heather Holmes for being on my qualifying exam committee. A special thanks to graduate program coordinator Lauren Worrell for her encouragement and help throughout the final stages of the dissertation process. To my friends from the Air Pollution Seminar, in particular Sara Forestieri, Dominique Young, and Alexandra Boris, thank you for inspiring me with your work, building a sense of community, providing feedback on my presentations, and of course for all the tasty snacks. I am also grateful to Matthew Palm for his help finalizing the paper in Chapter 2.

I spent much of my time and energy over the past few years teaching at California State University, Sacramento and I would like to acknowledge my colleagues there, in particular John Johnston, for supporting my growth as a teacher and encouraging me to keep working on my dissertation even when teaching felt overwhelming. I'm also grateful to my friend and colleague Ingrid Behrsin for being a model of healthy and supportive collaboration.

Through the challenges of grad school, my friends have been an invaluable source of support. Thank you to Rachel Woods-Robinson for being a fellow music and science nerd and commiserating with me throughout our academic journeys since the first day of M87 at UCLA. To Firas Abu-Sneneh for the steady supply of cute cat pictures, long walks on pretty Sacramento evenings, and for being an ally through some difficult experiences over the past few years. To Matthew Nesvet for giving the best pep talks, buying me random meals, and pushing me to try my hand at popular writing. To María José Durán, Miles Jones, and Candela for welcoming me into your home and reminding me to take breaks for the beach, the park, music, and fun with friends.

To my friends at KDVS radio for giving me a creative outlet, introducing me to all the best jams, and keeping one tiny corner of campus weird and independent. To the Sacramento Tenants Union for welcoming me into your community and inspiring me with all your hard work and passion for a better world. And to Carlos Tello Barreda for being a constant source of comfort and support over the past two years.

Finally, I would like to thank my family, especially my parents, Oliver Heyer and Lyn Alhorn, and my siblings, Adrian Gebhart and Theo Heyer, for believing in me, pushing me, and being understanding about all those missed holidays and IOUs. None of this would have been possible without you.

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Chapter 1 : Introduction

Exposure to air pollution emitted by mobile transportation sources such as cars, trucks, buses, trains, and ships negatively impacts human health (Hoek et al., 2013; Orellano, Reynoso, Quaranta, Bardach, & Ciapponi, 2020; "The Lancet Commission on pollution and health," 2018). In the United States, federal environmental legislation, such as the Clean Air Act, along with locally implemented fuel quality and vehicle standards have dramatically reduced the concentrations of air pollutants across the country, in part by reducing transportation emissions (Timothy, 2018). However, these improvements have not benefitted everyone equally (Ann E. Carlson, 2018; Clark, Millet, & Marshall, 2017). Beyond some regional failures to attain the standards, local variations in pollutant emissions and transport can create pollution hotspots – areas of high concentrations – even within regions that are overall in attainment. Achieving environmental justice, or the equitable distribution of environmental risks and benefits across a population, requires accurate modeling of air pollution at the community scale, including identification of local pollution hotspots.

1.1 Beyond 500 feet: The importance of community-scale air pollution modeling

Historically, air pollution monitoring and modeling have characterized ambient, or background, air pollution concentrations at the regional or city scale. Early evidence of the effect of air pollution on human health showed an increased risk of mortality based on exposures estimated by measuring long-term background concentrations of pollutants at a central monitor (C. Arden Pope et al., 1995; Dockery et al., 1993). While regional differences do significantly impact mortality rates, the recent proliferation of low-cost air pollution sensors in some cities and neighborhoods has illuminated the substantial variability of both the concentrations of pollutants that are easily measured using these low-cost sensors and associated health effects across much

smaller distances (Apte et al., 2017; Baldauf et al., 2009; Richard, 2013; Steffens et al., 2017). These gradients are particularly pronounced for pollutants with specific sources, such as BTEX compounds (benzene, toluene, ethylbenzene and xylene) near a refinery, as compared to multi-source pollutants. The micro-scale variability and elevated concentrations near emission sources underscore how "fenceline communities" living near pollutant sources are exposed to higher concentrations than their neighbors who live farther away.

Scientific studies and legislation addressing traffic-related air pollution have largely focused on defining a pollutant-independent "safe" distance from roadways—that is, a distance at which the concentrations of all road-associated pollutants presumably return to background levels (A. A. Karner, Eisinger, & Niemeier, 2010). Urban zoning laws also attempt to mitigate exposure via distance by physically segregating particular pollution sources, such as ports or industrial facilities, from residential areas (Shertzer, 2016). While distance from sources is a good first approximation for exposure, this method of identifying pollution hotspots fails to capture the variations that result from wind patterns, physical obstacles, traffic dynamics, and other local factors. For instance, a resident living upwind of a freeway has a different exposure than a resident equally distant but downwind from the freeway. Similarly, a resident living next to a section of freeway that experiences frequent bottlenecks may be exposed to more fine metals from brake wear and emissions related to acceleration than a resident living next to a freely flowing section of the same freeway.

1.2 Urban infill & traffic-related air pollution in the environmental review process

A more nuanced approach to characterizing pollution gradients surrounding transportation corridors is increasingly important as cities around the world move to incentivize urban infill—the development of underutilized urban spaces—as a solution to many modern urban problems

(Sciara, 2017). Transit-oriented infill development has the potential to improve mobility and decrease overall vehicle miles traveled, especially in combination with other policies (Salon, Boarnet, Handy, Spears, & Tal, 2012). However, these sites may also expose residents to air pollutants generated along freeways, ports and ship channels, or train corridors (Ann E Carlson, 2018; Corbett et al., 2007; A. Karner, Eisinger, Bai, & Niemeier, 2009; A. A. Karner et al., 2010).

In California, the environmental review process mandated for new residential developments by the California Environmental Quality Act (CEQA) omits consideration of transportation-related air pollution exposure for future residents. Although the health impacts of living near pollution sources are well documented, there is no mandatory process for assessing these risks for new residential developments near transportation corridors or other pollutant sources. The CEQA process exclusively considers the impact of a new project's construction and operation on the existing, surrounding community and not the impact of existing infrastructure or pollution sources on future residents, as affirmed in 2015 in *California Building Industry Association v. Bay Area Quality Management District* (62 Cal.4th 369).

In CEQA, the assessment of risks to future residents is acknowledged but delegated to poorly defined local jurisdictions, such as air quality management districts, or confined to the public input component of the review process. In the absence of a clear protocol, the evaluation of air pollution health risks for near-roadway developments can be either under- or overemphasized. On the one hand, the public input phase of CEQA's environmental review process is vulnerable to political objections masquerading as environmental concerns that end up blocking developments that would advance environmental objectives like densification or improved public transit. On the other hand, in localities with no local jurisdiction and limited public engagement, problematic developments that pose a legitimate threat may be approved without scrutiny.

The unexamined health impacts of siting residential developments near transportation corridors raise environmental justice concerns for populations with less political and economic leverage, who have historically been confined to areas with more environmental pollution, including air pollution (Bowe, Xie, Yan, & Al-Aly, 2019; Bullard, 1990; Hajat, Hsia, & O'Neill, 2015; Mohai, 2009; Tessum et al., 2021). As discussed in Chapter 2, transportation funding equity analyses by Metropolitan Planning Organizations (MPOs) in California have inconsistent methodologies for assessing equity with respect to traffic-related air pollution (TRAP) exposure. Some MPOs do not include air pollution exposure in their equity analyses, while those that do consider it quantify exposure using different metrics. Improved community-scale modeling of transportation-related air pollution would bolster efforts to assess changes in exposure related to transportation funding decisions.

A particular scenario in which this type of modeling would be beneficial is in predicting different TRAP exposure between exurban communities living near freeways with little investment in improved public transit versus urban core communities where most public transit spending is concentrated and where infill development is heavily subsidized. These exurban communities are also frequently exposed to agricultural emissions (Almaraz et al.). In the urban core, if new infill developments are placed in areas with elevated pollution without appropriate evaluation and mitigation measures, vulnerable populations may end up living in pollution hotspots. On the other hand, many TRAP reduction efforts such as improved public transit are concentrated within densifying urban core areas. Patterns of exurbanization and gentrification have in some instances displaced communities outside of urban core public transit networks, and therefore excluded them from the benefits of reduced local vehicle miles traveled (Chapter 2). In these exurban communities, investments tend to be skewed towards freeway expansion and

improvement rather than improved public transportation (Chapter 2). Approximating TRAP exposure by distance from a roadway is not sufficient for comparing changes in exposure for urban core communities living near transit hubs to those for exurban communities living near a commuter freeway. Equity analyses of transportation projects would benefit from improved methods for modeling TRAP exposure at the community scale.

1.3 Challenges in modeling air pollution at the community scale

Exacerbating the legislative gap in CEQA and the procedural gaps in transportation funding equity analyses is the lack of an appropriate methodology for predicting the risks of nearby emission sources on future residents. Long-term air quality monitoring is not a reasonable requirement for all projects. An effective review process would rely on easily measured criteria and/or numerical models to distinguish between acceptable and unacceptable near-roadway projects. However, most air quality models are either intended to predict concentrations at the mesoscale of a region or city, or for scenarios with few sources and simple terrain with few large obstacles. Because these models are not designed to incorporate the effects of obstacles such as buildings and other features of the urban environment that affect local-scale pollutant transport dynamics, they are less effective at predicting concentrations at ground level, where human health impacts occur. Downscaling models from the mesoscale to the micro, or community, scale is not simply a question of computing power and simulation time—it requires modeling of phenomena, such as obstacles and heterogeneous terrain, that become important at the local scale. It also requires detailed emissions inventories with a high spatiotemporal resolution. There have also been efforts to use neural networks in combination with numerical models to improve accuracy and reduce computational burden, but this approach has so far only been used to predict ozone concentrations (Sayeed et al., 2021).

One transportation-related pollutant of concern is particulate matter (PM), which has thoroughly documented health impacts in epidemiological studies (Pope, Coleman, Pond, & Burnett, 2020). The chemical transport model WRF-Chem has been adapted to model PM concentrations from diverse pollution sources in a port city at a spatial resolution of 250 meters, or about 2-3 city blocks using large eddy simulations (Joe et al., 2014). While this is a much smaller scale than previous chemical transport models, it is still based on the treatment of obstacles as surface "roughness" affecting turbulence rather than as individual small-scale obstacles. This modeling approach would not capture the different exposures of fenceline communities versus communities slightly farther away from a pollutant source and beyond the area of highest concentrations. Gaussian plume models, such as AERMOD, are similarly developed to model over longer distances and timescales and have limited capacity to model the impact of obstacles.

1.4 The need for accurate emission factors in high-resolution air pollution modeling

As researchers improve the spatiotemporal resolution of air pollution models to incorporate these scenarios, it is important that model inputs keep pace; emission inventories have been identified as one limiting factor in modeling micro-scale pollution gradients in urban areas(West et al., 2016). Emission factors quantify the emissions of a given pollutant per unit of activity. It is critical that emission factors be quantifiable on a spatiotemporal scale comparable to the model in which they are used. It is also essential that emission factors reflect changing regulations, technology, fleet compositions, and fuel standards. As air pollution interventions are proposed and implemented, emission factors must accurately represent the changed emissions related to a given intervention. This is important for modeling exposure as well as predicting and evaluating the impact of regulations and mitigation measures.

Emission factors for ships, in particular, have changed dramatically over the past decade in response to increasingly strict international fuel standards imposed by the International Maritime Organization as well as stricter local regulations in many areas of the world. Specifically, the International Maritime Organization's maximum allowable fuel sulfur content has been reduced from 3.5% to 0.50% m/m in 2020 and is limited to 0.1% in Emission Control Areas. Sulfur is a substantial component of PM emissions from ships and so the changing fuel standard will likely reduce the mass of PM emissions from ships (Lack et al., 2011; Winebrake, Corbett, Green, Lauer, & Eyring, 2009). As described in Chapter 3, despite the publication of many new studies of ship emission factors, ship emission inventories often rely on emission factor measurements made prior to these changes in fuel standards. Emission factors are also disproportionately representative of cruising conditions characteristic of emissions at sea. While most ship emissions occur at sea, far from human population centers, port cities and particularly fenceline communities leaving adjacent to ports or ship channels, are exposed to emissions while ships approach shore as well as during maneuvering and at berth. Emissions during these near-shore activities remain less fully characterized than those during at-sea cruising conditions (Chapter 3). Evaluating the potential reductions in exposure for fenceline port communities as fuel standards are improved and as other mitigation measures are introduced requires accurate emission factors representative of current fuel and operating conditions.

Broadly, the studies in this dissertation consider how air pollution emissions from transportation sources contribute to local-scale heterogeneities in pollutant concentrations and how improved community-scale air pollution modeling can help assess environmental health inequities. Chapter 2 examines how the failure to consider local-scale air pollution impacts hinders equity analyses of regional transportation plans, particularly in the context of the exurbanization and

displacement of disadvantaged communities from transit-rich urban core areas. The transportation equity metrics used in three different metropolitan areas of California (San Francisco Bay Area, San Diego, and Fresno) are reviewed to determine whether any metrics capture air pollution health impacts from transportation sources. Patterns of internal displacement and exurbanization within each metropolitan area are also investigated. Chapter 3 aims to improve model inputs for high-resolution air pollution modeling in port communities by updating estimates of brake-specific particulate matter (PM) mass emission factors (EF_{PM}) from ship engines via a meta-analysis of 55 lab and field studies. The dataset incorporates new measurements of ship emission factors that reflect changing fuel standards as well as more diverse ship types and operating conditions. The primary focus of the meta-analysis is the dependence of EF_{PM} on fuel sulfur content (FSC), which has been targeted by the International Maritime Organization as well as many local jurisdictions to reduce PM emissions from shipping. EF_{PM}'s dependence on other factors such as ship type and operating conditions is also investigated with a particular focus on ship types and operating conditions typical of near-shore emissions that contribute to urban air pollution in port communities.

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Chapter 2 : Are we keeping up? Accessibility, equity, and air quality in regional planning

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https://doi.org/10.1016/j.jtrangeo.2020.102891

Abstract

Many US metropolitan areas have undergone dramatic shifts in socioeconomic organization. As urban areas gentrify, many low-income residents and communities of color have transitioned towards the exurban periphery. These suburban neighborhoods tend to have fewer employment opportunities and are fairly disconnected from public transportation networks serving the urban core. Using regional transportation plans (RTPs) for three California MPOs, we show that the transportation accessibility and environmental health issues affecting these exurban communities are unique and inadequately captured by the MPOs' current equity metrics. MPOs' performance evaluation is regional and achieving equity within the urban core communities will not address emerging equity, accessibility, and air quality concerns for exurban communities. With a brief history and a focused case study of RTPs for the San Francisco Bay Area, San Diego, and Fresno, we examine how air pollution, equity, and transportation interact in three different types of 21st century cities. We find that when allocating limited transportation funds, California metropolitan planning organizations (MPOs) prioritize the improvement of existing public transportation in urban core areas over expansion of transit networks towards disconnected exurbs. This approach is an effective way to reduce vehicle miles traveled (and thus, air pollution) at the regional level due to high population concentrations in urban cores. However, this approach also

concentrates the air quality benefits of VMT reduction in these same urban core areas. Exurban residents' on-road and near-road exposure to Traffic-Related Air Pollution (TRAP) will not be reduced by improving public transit within the urban core. We argue that although these suburban and exurban communities are a small percentage of the regional population, they have a right to share in the benefits of transportation investments, particularly given the historical and ongoing patterns of displacement and economic exclusion from urban core areas.

2.1 Introduction

Since the 1970s, many progressive thinkers have decried American suburbanization as a failure of urban planning and a furthering of structural inequalities (Lipsitz, 1998), which have been enabled by highly racialized patterns of highway development (Lewis, 1997; Rose and Mohl, 2012). However, the exclusion and displacement of disadvantaged households from increasingly unaffordable urban areas over the past few decades has fundamentally changed how suburbs fit into regional transportation networks (Wang and Woo, 2017). Forty-eight percent of the increase in US residents living below the federal poverty line between 2000 and 2015 took place in the suburbs (Knee- bone, 2017). Urban planner Alex Schafran argues, "To be anti-sprawl or anti-suburbs now is to be against the homes and dreams of millions of working-class families, many from communities of color, in suburbs and exurbs around the United States," (Schafran, 2013). The changing character of urban and suburban spaces challenges planners and policymakers to reconsider where and to whom the benefits and costs of transportation planning decisions accrue.

Planners have long advanced urban infill—dense, mixed-use development of underutilized urban spaces— as the leading strategy to curb sprawl and reduce vehicle miles traveled (VMT) (Handy,

2008). The logic of infill is that dense, mixed-use urban development reduces on-road emissions of greenhouse gases (GHGs) with the added benefit of reducing criteria air pollutants by making public transit and active modes of transport more accessible (Salon et al., 2012). As we will show, transportation planning - informed by an infill approach - prioritizes public transit investments in urban core areas where populations of disadvantaged residents are declining. We also highlight the increasing numbers of suburban and exurban neighborhoods lining commuter freeways, most populated by growing disadvantaged communities. These communities are excluded from the benefits of urban core infill transit projects yet retain the localized impacts of near-road air pollution and long car commutes.

We argue that transportation researchers and practitioners must consider how poverty's crawl to outer suburban and exurban communities challenges the working paradigm of equitable infill-based regional planning. Using the San Francisco Bay Area, San Diego, and Fresno as three California case studies, we investigate how the spatial distribution of poverty has changed over the past two decades and whether MPOs are accounting for these demographic shifts in the way they treat accessibility, equity, and air quality in their regional transportation plans (RTPs). We begin with a background section describing: (1) California's urban infill approach to regional planning, and (2) the ways in which spatial patterns of poverty complicate how we think about equity in transportation planning, particularly around air quality. Our results begin with maps of regional transit access and the suburbanization of disadvantage in each case study using American Community Survey data. We then review how our case study MPOs operationalize the interaction of transportation accessibility, air quality, and equity in the RTP metrics they have chosen to assess progress towards a more equitable transportation system. In our conclusions, we consider policy implications and research gaps.

2.2 Background

2.2.1 The urban infill approach to regional planning

In the US, MPOs set broad transportation development strategies at the regional level and produce long-term RTPs that serve as blueprints for the next several decades of transportation development (Sciara, 2017). In 2008, the state of California enacted Senate Bill 375 (SB 375) to mandate that RTPs include a Sustainable Communities Strategy (SCS) integrating land-use, housing, and transportation planning to reduce GHG emissions. SB 375 requires that two performance measures be included in all California RTPs: (1) reduce per-capita CO_2 emissions from cars & light-duty trucks by 15% by 2040, and (2) house 100% of projected population growth by income level. Beyond these two statewide performance measures, the MPOs creating the plans have the freedom to define their own region-specific priorities. It is also worth noting that California MPOs receive 75% of the state and federal funds designated for transportation improvements (SB 45, Kopp, 2020) so their actions and decisions play an influential role in transportation provision.

While air quality improvement is not an explicit performance measure of SB 375, air pollution is inextricably tied to both climate change mitigation strategies and transportation planning. Urban air quality in the US is regulated by the Clean Air Act (CAA), which sets standards for ambient concentrations of air pollutants, known as the National Ambient Air Quality Standards (NAAQS). With respect to transportation planning, the CAA prohibits federal funding of transportation activities that would cause a new violation of the NAAQS or contribute to or prolong ongoing violations. RTPs for regions not in attainment must include an air quality conformity analysis ensuring that the plan advances the state's overall strategy for attaining the standards. In combination, the requirements to dramatically increase housing and reduce emissions of both GHGs and criteria air pollutants at the regional level points towards an infill or

smart growth approach (Barbour and Deakin,

2012).

And yet, despite SB 375, plans to increase affordable housing in high-density urban areas have yet to manifest. Statewide, new housing is being produced at only slightly over 50% of the required annual rate of 180,000 homes per year to meet projected need (Palm and Niemeier, 2017; DHCD, 2018). The last time California produced housing at the required rate was in 2006 (DHCD, 2018). An analysis of planned housing capacity in California clearly shows there is currently not enough capacity for the needed increase in housing units and further, that much of the capacity that does exist is outside urban areas (Monkkonen and Friedman, 2019).

The equity of the urban infill approach depends on an integrated treatment of transportation, housing, GHG emissions reduction, and air quality improvements. The complex interaction between these dimensions of planning is exposed, for example, in fierce debates between agencies and advocates in the domains of housing and public health over the state's increasing reliance on highway-adjacent land parcels to host subsidized affordable housing in both core and exurban communities (Barboza and Zahniser, 2017; CARB, 2018). The MPOs' failure to integrate these issues throughout the regional planning process aggravates emerging spatial patterns of social disadvantage in cities.

2.2.2 The suburbanization of poverty: Complicating transport equity

Transportation's role in re-enforcing social inequities is well documented. More than 7.5 million U.S. households lack a vehicle, and while 90% of these households have access to transit, they face a commute of approximately 90 min to access just 40% of the available jobs (Tomer, 2011). The dependence on transit as a sole mobility provider is crushing, especially for African American and Latino commuters (Karner and Niemeier, 2013; Karner and Duckworth, 2018).

The interplay between jobs, housing, education, and transportation creates structural barriers that prevent low-income households from escaping poverty (Kutty, 2009). Economists now offer strong evidence that neighborhood poverty can compound household disadvantage by limiting access to opportunities (Chetty and Hendren, 2018). This body of work positively associates lower commute times with upwards intergenerational mobility (Chetty et al., 2014), suggesting that transportation policy may play a role in breaking the intergenerational transmission of poverty.

Because MPOs receive federal funding, certain federal regulations prohibiting discrimination motivate them to consider equity issues in the planning process. Title VI of the Civil Rights Act of 1964 prohibits them from discriminating on the basis of race, color, and national origin in their programs and activities, including RTPs. Executive Order (EO 12898), enacted in 1994, guides federal agencies to identify and address cases where their actions cause disproportionate health and environmental effects on minority and low-income populations. MPOs adhere to federal transport agency guidance interpreting EO 12898. Increasingly, MPOs are including equity analyses in their RTPs.

Despite significant awareness of transportation's role in creating structural disadvantage, policy efforts to incorporate equity and civil rights into transport planning remain fraught with methodological challenges (Karner and Niemeier, 2013). While federal law mitigates the most egregious transgressions, important disparities in transportation access – across all modes – persist between affluent white users and low-income minority users (Golub et al., 2013). Prior reviews of transportation equity metrics encourage practitioners to consider, for example, transit accessibility benefits in transportation planning (Manaugh et al., 2015; Karner, 2018), proposing a "ladder of justice" framework for evaluating policy responses to transport

accessibility injustice in the United States (Martens and Golub, 2019). They define the most progressive standard of equity in this approach as restorative equalization, in which policies result in greater benefits to historically disadvantaged communities as a corrective for past injustices. Others have advanced a Rawls-inspired focus on vertical equity in transportation, or an emphasis on providing services to those with the greatest existing disadvantage (Van Wee and Geurs, 2011).

Regardless of the path to remedy injustices, the spatial relocation of disadvantaged households in the United States complicates our ability to remediate historical injustice in transit accessibility. In many cities across the United States, communities of color and low-income populations have been either displaced or excluded from gentrifying urban core areas in what some scholars characterize as an urban inversion (Frey, 2003; Ehrenhalt, 2013). These patterns became particularly acute after the 2008 foreclosure crisis, which dramatically shifted disadvantage along lines of race and class towards the exurbs and outer suburbs (Schafran, 2013; Kneebone and Garr, 2010). This inversion significantly reduces displaced households' transit accessibility, the metric most extensively championed by scholars considering transport equity within a US legal framework (Martens and Golub, 2019). The lower concentration of both jobs and public transportation in some suburban/exurban neighborhoods also exacerbates the "spatial mismatch" between employment opportunities and job-seekers. An analysis of low-income job seekers in the San Francisco Bay Area, for example, found that a majority of zip codes had an excess of low-income jobs available and a deficit in the number of job-seekers within a reasonable distance (Stacey et al., 2019). The ex-urbanization of poverty also compromises the equity of transport funding approaches such as fuel taxes and bridge tolls, by which ex-urban lower income households often pay into the very public transit systems that

excludes them (Karner and Duckworth, 2018).

There is evidence that in some cases, new public transit development can actually contribute to the displacement of lower income households, further complicating efforts to address historical injustices in transport planning (Chapple et al., 2017). Scholars have begun associating new public transit investments with gentrification along lines of both income and race (Brown, 2016; Boarnet et al., 2017; Palm and Niemeier, 2018). Some scholars posit that these investments may also contribute to exclusionary displacement by making core urban neighborhoods too expensive for new low-income households (Zuk et al., 2017).

If we turn to air quality, it is not surprising that equity questions are rarely accounted for in many regional plans (Wernham, 2011). The CAA does not require intra-urban analysis of the distribution of air quality improvements resulting from transport investments as part of the RTP process. It does require project-level "hot-spot analysis" of certain transit or highway projects that could cause concentrations above the NAAQS at the sub-regional scale, but this analysis occurs separately from the RTP process and is not usually targeted at urban infill or exurban development. Under urban infill strategies, the intensity of exposure to elevated pollutant concentrations increases if alternative transport strategies fail to materialize (Mansfield et al., 2015), whereas impaired air quality outside the urban core occurs mostly adjacent to major roadways, where concentrations are high in the immediate vicinity of the road, but quickly return to background away from roadway edge (Karner et al., 2010). There is compelling evidence that inequalities in air pollution exposure cannot be captured by the current regulatory monitoring system (Tessum et al., 2019). This is due in part to reliance on measurements from a geographically sparse network of ambient air quality monitors that cannot capture variations within regions (Van Le and Tham, 2017). This sparsity can make mitigation strategies appear

ineffective when evaluated at the regional level, despite providing important benefits to disproportionately impacted communities at the local level (Karner et al., 2009). Although newer, more spatially and temporally resolved air quality data from mobile sensors is beginning to highlight important variations in exposure – and health effects – at finer geographic scales (Apte et al., 2017), the governance processes by which we identify and analyze hot-spots remains constrained. Ex-urbanization can also increase low-income residents' exposure to air pollution by inducing travel mode shift; evidence suggests that car commuters have higher air pollution exposures than both active commuters and bus commuters (Cepeda et al., 2017).

In sum, we argue that California's infill approach to reducing GHG emissions, improving air quality, and housing a growing population exacerbates social inequities facing communities in the suburbs and exurban periphery. Despite ambitious housing goals, the state's lower income households continue settling in suburban and exurban communities due to a combination of displacement and exclusion from gentrifying urban core areas. This upends the traditional conception of suburbs as self-excluding communities with more socioeconomic privilege than inner-city communities. Scholars have yet to consider how the intersection of these dimensions may reorder the distribution of transportation infrastructure benefits and burdens along lines of income and race. Our research fills this gap in our understanding.

2.3 Methods

We focus our analysis on three metropolitan planning regions of California: (1) the San Francisco Bay Area and the Metropolitan Planning Commission (MTC); (2) San Diego and San Diego Association of Governments (SANDAG); and (3) Fresno and Fresno Council of Governments (FCOG). We selected California because of the state's policies, such as SB375 and AB617 Community Air Protection Program, indicating a strong commitment to addressing both equity concerns and air pollution. The regions we consider all experienced some of the fastest

population growth in the state over the past decade (**Figure 2-1**). MTC includes nine counties surrounding the Bay Area, SANDAG comprises San Diego County, and FCOG includes Fresno County. We began by empirically establishing the growth of low-income commuters in automobile-oriented neighborhoods. To do this, we first measure income changes at the census track level between the 2000 and the 2014–2018 American Community Survey (ACS). We use the percent of residents in or below 150% of the federal poverty level and take the difference between the 2000 Census and the 2014–2018 ACS. This produces our first statistic: the percentage change in the share of residents in or below 150% of the federal poverty line by census tract. Adjustments for census tract changes between both datasets are made using the Census Bureau's Census Tract Crosswalk file, which provides weights for associating 2000 data into 2010 census tracts using population-based re-weighting (Census Bureau, 2020). For the remainder of the paper, we refer to this statistic as the tract-level change in the percent of residents in or near poverty. Despite using a benchmark higher than the federal level of poverty, our reference phrase "in or below poverty level" takes into consideration two important dynamics. First, the official poverty threshold reflects "a notion of economic need based on living standards that prevailed in the mid-1950s" which has since been tied solely to inflation (Gabe, 2013, p. 3). Were the initial methodology re-conducted in 2015, the resulting rate would be "be over three times higher than the current thresholds" (Gabe, 2013, p. 3). Second, the federal poverty rate fails to account for large geographic variations in the cost of living (Citro and Michael, 1995), suggesting that it may significantly undercount the share of households experiencing poverty in places like California.

We then visually compare the resulting tract level changes in poverty with the percent of commuters in or near poverty who commute via automobile in the 2014–2018 ACS,

including both single occupancy driving and carpooling. We use ordinary least squares to test the association between commute mode and our poverty variable. We also examine the distribution of in or near poverty residents using the neighborhood-level auto commute rates within each MPO. For these data, we hypothesize that the percent of in or near poverty rate population has risen in neighborhoods where commute by auto reliance is high. Conversely, these rates will have declined in places where those in or near poverty rely on non-auto modes. Finally, using these results, we qualitatively assess the degree to which our three MPOs' current equity metrics account for the unique issues affecting communities of concern residing outside the urban core.

2.4 Results

2.4.1 Relocating disadvantage

2.4.1.1 Metropolitan Transportation Commission (San Francisco)

San Francisco's regional public transportation system is oriented around a widely used BART subway system and the city's light rail and bus system, MUNI, both of which connect to various municipal and regional bus systems as well as five intercity rail lines: Amtrak, Caltrain, Sonoma Marin Area Rail Transit (SMART), Altamont Commuter Rail, and the Santa Clara Valley Transit Authority (VTA). The region decentralizes towards several dispersed urban nuclei with gentrifying urban core areas that have driven communities of color and low-income residents into the exurban periphery (Schafran, 2013). From 2000 to 2018, we observe large decreases in poverty within the gentrifying urban core areas of San Francisco, the East Bay, and San Jose (**Figure 2-2**a). During the same period, we can observe large increases in poverty in the outer limits of these urban core areas as well as in suburban/exurban nuclei including those lining the I-80 (Vallejo, Fairfield, Vacaville), US-101 (Santa Rosa), and SR-4 (Pittsburgh/Antioch)

freeways (Figure 2-2).

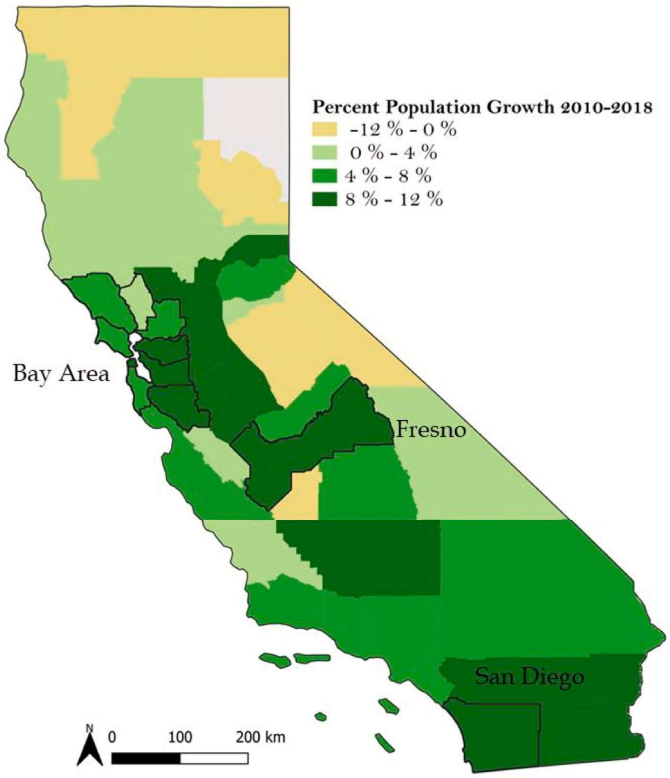

Figure 2-1 Percent population growth by county between 2010 and 2018, from US Census data. Counties in case study regions outlined in black

We find that workers in or below 150% of the federal poverty level living outside of urban core areas overwhelmingly drive to work. These same households are more likely to use non-auto modes when they live within the relatively transit rich core urban areas. Notably, areas with higher transit density with more workers in lower income brackets opting not to drive are also areas that have experienced declining low-income populations over the past two decades. San Francisco and Alameda contain 65% of the MTC tracts where poverty decreased between 2000 and 2014–2018. The correlation between poverty-level households and commute by auto is significant at $p < 0.001$ level (coefficient: 0.049, r-squared: 0.04).

2.4.1.2 San Diego Association of Governments (San Diego)

San Diego County is characterized by a small urban core surrounded by a sprawling

network of auto-oriented suburbs that extends into two adjacent suburban counties: Imperial and Riverside. Its transport system includes an inner-city oriented trolley network plus HOV lanes and bus rapid transit. Despite recent transit investments (largely as the result of a lawsuit), the region is known for its heavy reliance on cars: in 2017, 76% of workers over the age of 16 in the metropolitan area drive alone to work, 8% carpool, and only 3% take public transportation (US Census Bureau, 2017). The region has grown, on average, at much the same rate as the national rate. However, there have been areas of accelerated growth in the suburban counties of Imperial and Riverside, with significant numbers of people commuting into the San Diego urban area for higher quality employment opportunities (SANDAG, 2011).

As in the Bay Area, we observe the greatest increases in poverty in San Diego's outer suburbs concurrent with decreases in poverty in San Diego's urban core district (Figure 2-3a). We also observe decreases in poverty and a lower share of workers in or below 150% of the federal poverty level driving to work along the Coaster light rail route, which connects San Diego to Oceanside (Figure 2-3). These trends reflect a shift in lower income households away from central cities and towards outer suburbs that are not well served by transit. Workers earning less than or at 150% of the FPL residing outside the urban core are more likely to drive to work than counterparts in the urban core (Figure 2-3). Still, unlike San Francisco, even the urban core area of San Diego County has low public transit ridership by this income segment, with nearly half driving either alone or carpooling. The correlation between commute by mode vehicle and poverty is significant at $p < 0.001$ level (coefficient: 0.067, r-squared: 0.02).

2.4.1.3 Fresno Council of Governments (Fresno)

Fresno, in California's agriculturally dominated Central Valley, recently experienced one of the fastest growing and most diverse populations in California. The county, centered on the

Fresno Clovis metropolitan area (FCMA), represents a new kind of metropolitan area that began to boom in the second half of the twentieth century. Between 1970 and 2010, Fresno County's population increased by 125% compared to 87% and 52% for the state and nation. It is one of the metropolitan areas receiving emigrants who can no longer afford the rapidly gentrifying urban centers of California, such as the Bay Area (Romem and Kneebone, 2018). In this way it acts in a similar role to suburban and exurban nuclei within MTC. Most of the county's population is concentrated within the FCMA; there are small communities outside this area, but no major suburbs as in the Bay Area or San Diego. Fresno's transit options are limited in both scale and number. Fresno has no local passenger rail network, although it does have one inter-city Amtrak stop. The Fresno Area Express (FAX) bus system, which does not extend far past the urban core, has seen consistently declining ridership since 2010 despite the implementation of new Bus Rapid Transit lines (FCOG, 2017). Fresno is adjacent to both SR99 and I5, two major freight routes.

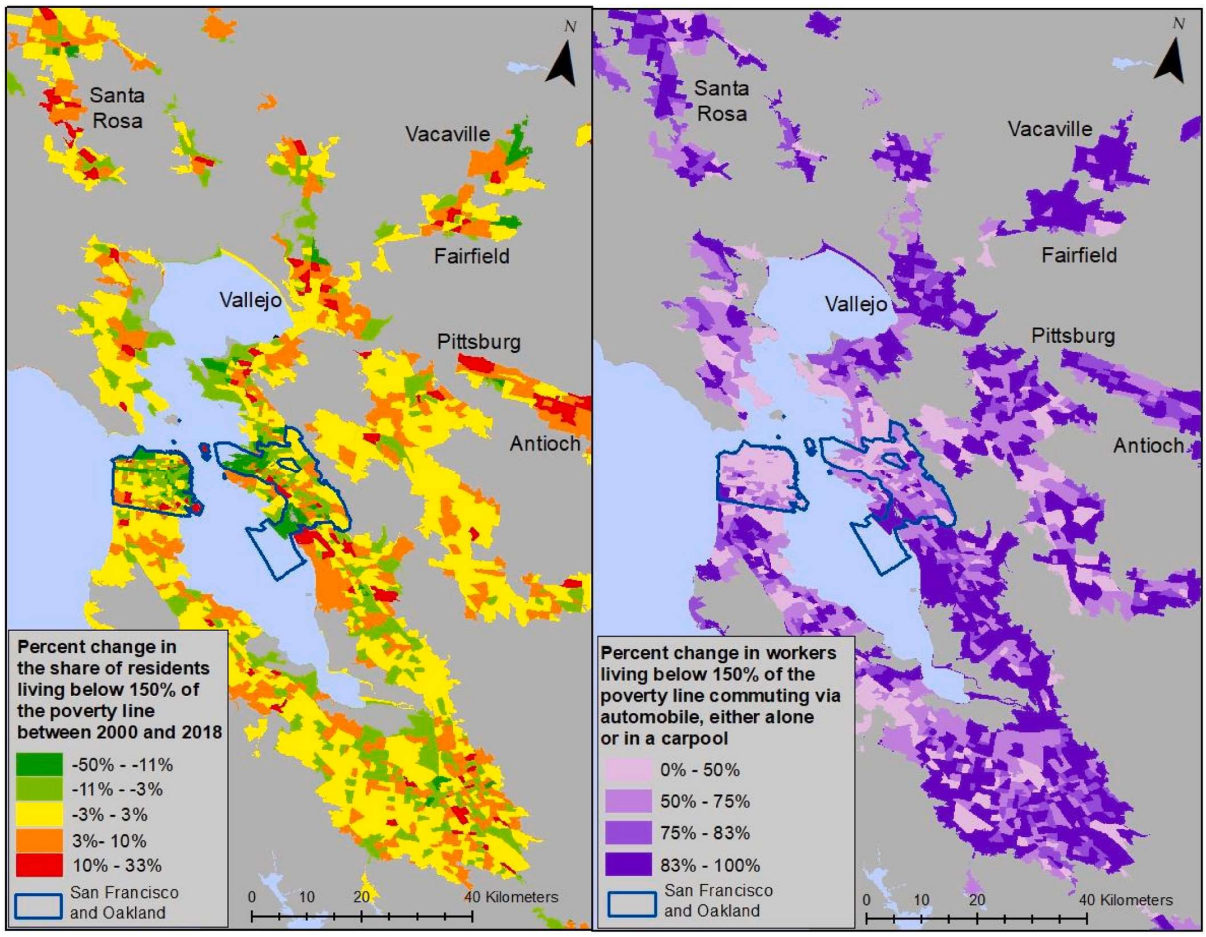

Figure 2-2 MTC SF Bay Area: (left) Percent change in the share of residents living below 150% of the poverty line between 2000 and 2018. (right) Percent change in workers living below 150% of the poverty line commuting via automobile, either alone or in a carpool.

Fresno also struggles with some of the worst air quality in the country—it is currently in nonattainment for five of six NAAQS pollutants. Research on the equity analyses conducted by smaller and more rural MPOs, such as FCOG, is limited and highlights the need for further investigation of the dynamics of equity in less densely populated areas with limited transit options (e.g., see Karner, 2016).

In Fresno, we do not observe a strong pattern of poverty suburbanization, in part due to Fresno's lack of major suburbs. Within the FCMA, the greatest decrease in poverty, the smallest share of those workers in or below 150% of the FPL driving to work, and the highest transit

density occur within the urban core; we observe the reverse in the area directly to the north of the urban core (Figure 2-4). The region overwhelmingly drives to work and ultimately there is little difference in mode choice between the urban core and the rest of the county. Outside the FCMA, increasing poverty in at least one community (Selma City) along SR-99. In this region, the relationship between vehicle commuters and poverty is not significantly correlated.

2.4.1.4 The distribution of low-income workers

The percentage changes in neighborhood poverty levels that we note in the preceding sections could exaggerate the share of residents in poverty who live in auto-dominated neighborhoods depending on baseline concentrations of neighborhood poverty levels. To rule this out, we calculate the distribution of residents below 150% of the federal poverty level by neighborhood auto mode share among workers in this income group. Results for all three regions are presented in Figure 2-5.

Figure 2-5 illustrates that in each region, the vast majority of residents below 150% of the poverty level live in neighborhoods where workers in the same income band commute via auto modes. Even in the San Francisco Bay Area, less than 20% of residents below this low-income threshold live in neighborhoods where a large majority of workers in this income group are able to commute without an automobile. In Fresno and San Diego, over 90% of low-income residents live in neighborhoods where at least 60% of low-income workers commute by car. This highlights how the suburbanization of poverty, in combination with an existing automobile-dominated landscape, locks lower income residents in these regions into automobile-based travel.

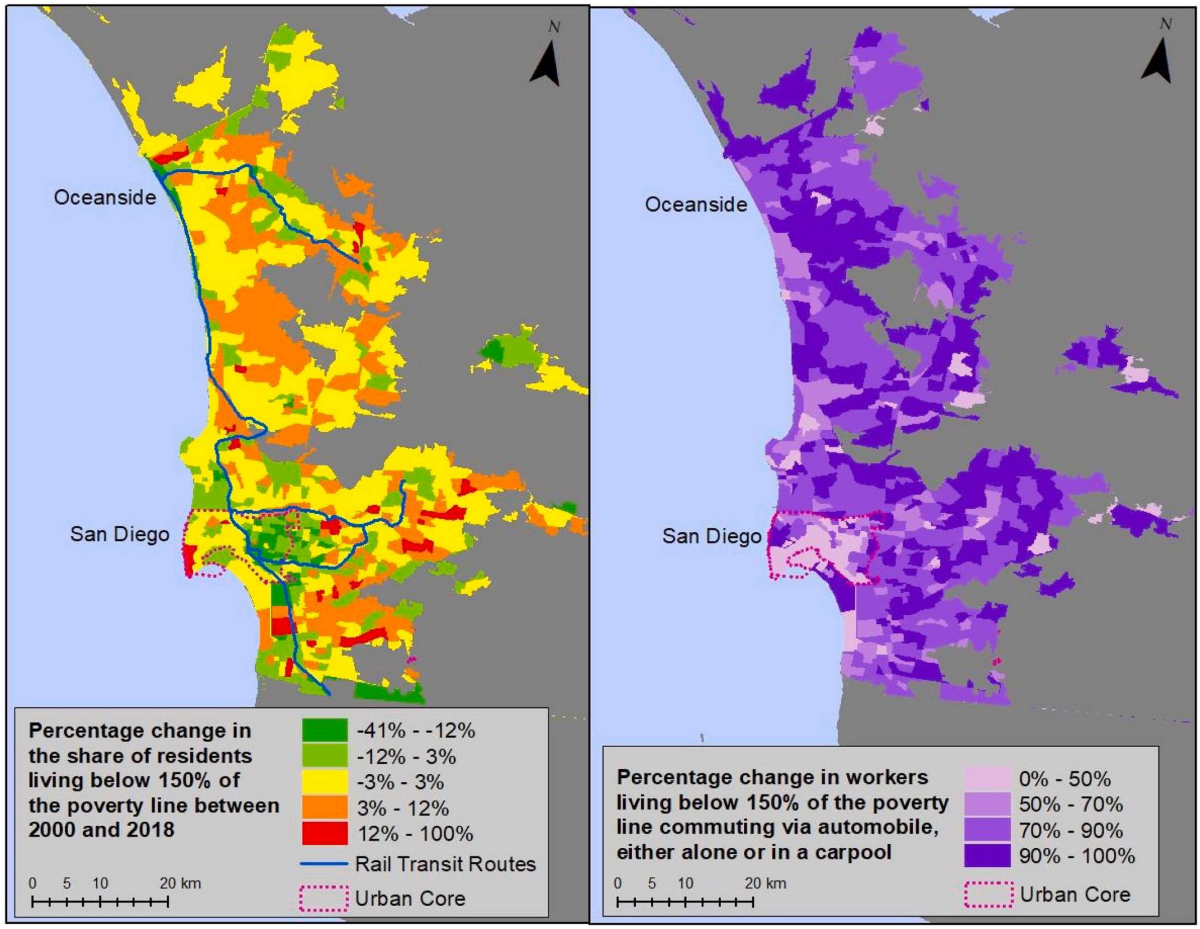

Figure 2-3 San Diego County: (left) Percentage change in the share of residents living below 150% of the poverty line between 2000 and 2018. (right) Percentage change in workers living below 150% of the poverty line commuting via automobile, either alone or in a carpool. Data for both maps from US Census Bureau.

2.4.2 Equity metrics

As part of the required equity analyses, each MPO defines specific equity issues and metrics to assess progress attained through implementation of the RTP. For the purpose of comparison, we group the equity metrics defined in the RTPs into the four critical dimensions of equity in transportation systems supported by the literature: (1) *Accessibility*: how convenient it is for people in different communities to travel from their homes to workplaces and amenities, (2) *Health effects*: how the health burdens and benefits associated with various modes of transportation are distributed between different communities, (3) *Distribution of funds*: how equitably transportation investments are allocated to different communities, and (4) *Potential for*

displacement: how transportation and housing policies change affordability and the ability of disadvantaged communities to continue living in their homes (Table 2-1). Although there is a clear and compelling interaction between these dimensions, it is very clear from our analysis that the MPOs consider each dimension of equity separately (Table 2-1). With these dimensions separate, it is easy to lose track of the ways in which they interact for any given project or locality. Here, we assess whether the equity issues and the associated metrics selected by each MPO adequately account for the important interactions between the four dimensions of equity in transportation planning defined above, and whether they are likely to capture inequities experienced by disadvantaged communities outside urban core areas.

2.4.2.1 Accessibility: Travel time, traffic congestion, and proximity to transit

Accessibility, long considered the key metric for the transportation system, is the only dimension of equity considered in every RTPs' equity analyses. All three MPOs use travel time is the accessibility metric for establishing how well the transportation system serves different communities; SANDAG included two additional travel-time-related metrics:
percentage of the population below an acceptable threshold for trip times and percentage of vehicle work trips less than 30 min long. The aggregation of travel time metrics across all transportation modes obscures differences *between* modes in terms of in-vehicle air pollution exposure, travel cost, and quality of life. SANDAG's percentage of vehicle work trips less than 30 min partially captures the extra burden on commuters who drive longer distances, but does not directly draw out these populations by presenting percent driving longer than 30 min. MTC's accessibility metrics apply an average travel time computed
across growing disadvantage in places like Antioch and Fairfield and shrinking disadvantaged populations in inner San Francisco (Figure 2-2a). The aggregated metric will lag the changing

demographic trends. This lag translates to a late warning flag which comes only after a sufficiently large enough share of low-income populations have relocated to the periphery to raise the average substantially.

Travel time is a useful metric for communities of concern living outside the urban core who may experience long commute times regardless of mode. Distinguishing between modes would begin to capture the different equity issues, such as cost and air pollution exposure, associated with long commutes by different modes. Making clear low-income populations traveling longer than 30 min is also important.

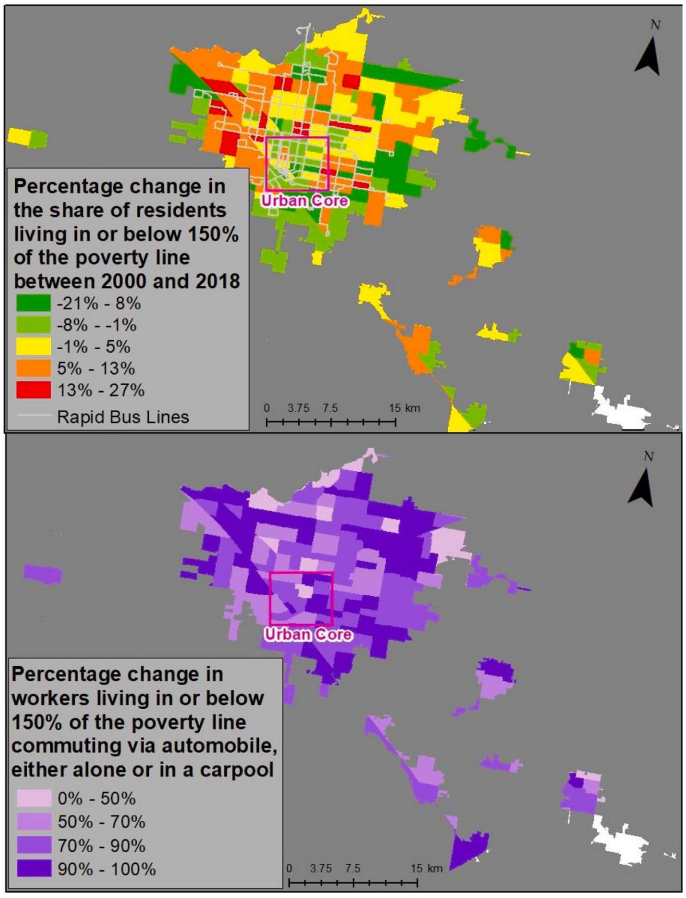

Figure 2-4 Fresno County: (top) Percentage change in the share of residents living in or below 150% of the poverty line between 2000 and 2018. (bottom) Percentage change in workers living in or below 150% of the poverty line commuting via automobile, either alone or in a carpool. Data for both maps from US Census Bureau.

2.4.2.2 Traffic-related air pollution exposure

Air pollution, environmental justice, and transportation are inextricably linked, and yet there is relatively little discussion of TRAP in the environmental justice analyses of the three RTPs. MTC considered the role of TRAP first in the overall RTP targets, under the category of "Healthy and Safe Communities." In the original 2013 plan, the equity analysis includes the TRAP-related metrics of average daily VMT per populated square mile within 1000 ft of heavily used roadways and reduction in number of premature deaths from exposure to particulate emissions. However, the plan does not account for potential air pollution hot-spots. For instance, PM_{10} reductions are quantified as the quarterly-averaged, nine-site mean concentration for the Bay Area rather than concentrations at individual air quality monitors. This is sufficient for producing a regional picture but does not speak to hotspots or neighborhood-level air pollution effects that have been clearly demonstrated in the literature (e.g., see Apte et al., 2017; Karner et al., 2009).

MTC's 2017 Equity Analysis Report considered air pollution more in depth, raising biker and pedestrian safety concerns about potential increases in vehicle traffic and congestion. The MTC Equity Analysis is noteworthy in that it considers the burden of diesel PM in EJ communities by examining the percent of truck traffic along corridors adjacent to EJ communities. The Plan also points to the BAAQMD's Community Air Risk Evaluation (CARE) Program, which targets communities with disproportionate exposure to air pollution. Notably, there are several outlying suburban communities with a large presence of communities of concern which also experience some of the highest truck traffic in the Bay Area, including Fairfield and Vacaville (Metropolitan Transportation Commission, 2017b, Map 32). Despite the acknowledgment of TRAP exposure in communities of concern outside urban core areas in the Equity Report, MTC's funding priorities do not reflect a commitment to ensuring that these

communities share equally in the air pollution benefits of reducing VMT at the regional level.

SANDAG and FCOG did not incorporate air quality in their equity analyses to the same degree. SANDAG did not consider air quality at all in its environmental justice targets and only included it in the RTP as part of the required air quality conformity analysis. FCOG measured the burden of TRAP in near-road communities with the percentage of EJ population relative to non-EJ population living within 500 ft of Class 1 (major highways) and Class 2 (expressways) roadways. This metric does not account for variable traffic volumes or congestion, which may include trucks at a volume sufficient to create TRAP conditions, on different roadways. It also would not capture the different reductions in freeway traffic between the core and the periphery under infill development.

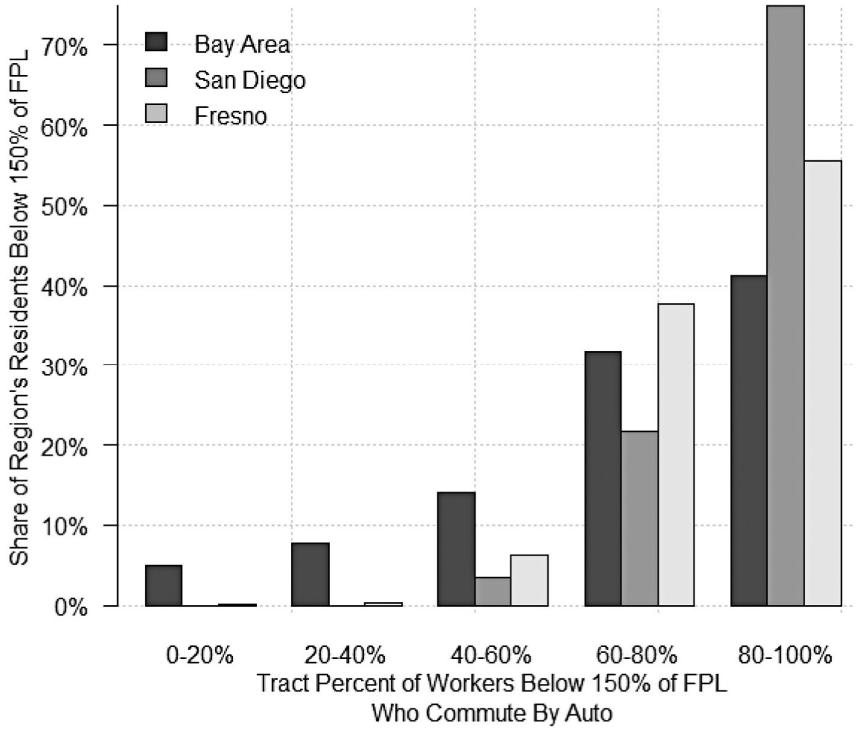

Figure 2-5 The regional distribution of workers below 150% of the poverty level by neighborhood share of low income workers who commute by auto modes.

Table 2-1 Dimensions of transportation equity and metrics used by the MPOs in each RTP

	MTC	SANDAG	FCOG
Accessibility: *Travel time, traffic congestion, and proximity to transit*	Average travel time (commute trips) Average travel time (non-work-based trips)	Average time per person per trip, all modes/all types of trips % of SOV/HOV work trips ≤30 min during peak periods % of population within (via auto or transit): 30 min of higher education 30 min of airport 15 min of basic health services 15 min of parks or beaches % of homes within ½ mile of transit stop	Average travel time (morning peak hours) Average travel time (evening peak hours) % VMT at level of service E or worse
Health: *Traffic-Related Air Pollution Exposure*	Average daily vehicle miles traveled per populated square mile within 1000 ft of heavily used roadways	None	Number of households within 150 m of class 1 or 2 roadway
Distribution of transit funds	None	Estimated costs of transit investment distributed per person within 3 miles of a transit or highway project	Increase in % Person Miles Traveled on transit Transportation (all modes) investment per PMT
Potential for Displacement: *Affordability of transportation and housing*	% income spent on housing and transportation by low-income households % rent-burdened households in high-growth areas	None	Increase in amount of multifamily housing options

Overall, the MPOs use TRAP exposure metrics that simply compare an estimated exposure of all EJ communities in the region to all non-EJ communities in the region. This approach will miss localized hotspots and suburban and rural communities with a unique history of displacement or exclusion along with high TRAP exposure.

2.4.2.3 Transportation funding and equity

SANDAG and FCOG define metrics to assess the equitability of transportation investments. MTC did not include a funding equity metric, despite having funding equity as one of two guiding EJ principles. Each region's funding plan broadly subscribes to the ideals of urban infill development. MTC's Plan Bay Area describes a future of "neighborhoods where transit, jobs, schools, services, and recreation are conveniently located near people's homes" (MTC, 2013). Similarly, SANDAG's plan envisions a "compact urban core where more people live and use fewer resources." FCOG also aspires to the urban infill ideal, with the goal of creating "communities that are more compact, walkable and transit-oriented," (FCOG, 2017). Despite common language, funding by mode (Figure 2-6) as well as the metrics deployed to evaluate the equity of these infill investments varies significantly by region.

MTC, which already has a substantial public transportation network servicing the urban core, is the only case study to allocate most of their funding (64%) to public transportation projects. But as of the 2017 update to the RTP, about 87% of transit funding is dedicated to operation, maintenance, and modernization of the existing transit network compared to just 10% dedicated to expansion projects (MTC, 2017a, 2017b). The largest of these expansion projects is the planned high-speed rail project; other priority projects include extending BART to Silicon Valley (Santa Clara County), extending Caltrain to downtown San Francisco, extending light rail service in Santa Clara County, and increasing frequencies and capacity for BART, bus, and

rail throughout San Francisco proper.

Solano County, home to highway-adjacent communities with increasing low-income populations that mostly drive to work and have limited regionally connected transit options, will be the site of only one of the top 20 planned transit projects: upgrading an existing rail station into a more locally connected multimodal station in Fairfield/Vacaville (MTC, 2017a, 2017b). These decisions reflect MTC's prioritization of high quality, accessible transit in the urban core over inclusion of peripheral communities.

SANDAG RTP allocates about half of its funding to public transportation projects and significantly invests in the expansion of transit. The priority transit projects are distributed between core improvements, such as a new tunnel for the trolley in downtown and expansion of the trolley and rapid bus transit lines towards urbanized areas without transit connections. SANDAG assessed the equity of fund distribution using the assumption that people living within three miles of a transit or highway project will accrue benefits from that investment, an assumption heavily critiqued in the transportation literature (Manaugh et al., 2015; Karner and Niemeier, 2013; Martens and Golub, 2019). A more informed measure might attempt to track how the resulting investment impacts affordability and air pollution exposure in these locations (Zuk et al., 2017).

FCOG took a different approach to assessing the equitable distribution of transportation investments: it considered the total transportation project investment across all modes per person miles traveled in EJ vs. non-EJ communities, suggesting that funding should be prioritized in EJ areas with high VMT. In an urban sprawl scenario in which low-income or minority people living in peripheral areas travel longer distances to reach employment opportunities and amenities, this metric would incentivize investments aimed at improving their transportation options. However,

importantly, it does not distinguish between transit and highway investments. Despite the intention of fostering walkable, transit-oriented cities, FCOG dedicates less than 20% of its funding to public transportation projects. Ultimately, FCOG prioritizes rehabilitating and maintaining its existing vehicle infrastructure, to address roadway disrepair and alleviate congestion.

2.5 Discussion

Urban infill development can greatly reduce transportation-related emissions for both GHGs and other air pollutants at the regional scale, while also improving the efficiency of the transportation system. Legislation such as SB375 links climate mitigation, transportation planning, and equity issues, such as affordable housing and air pollution exposure. However, based on our analysis, we conclude that because the current approach to urban infill development in California does not go far enough in holistically integrating *across* these dimensions of equity, it is actually more likely to exacerbate existing structural inequities.

An equitable implementation of urban infill development depends on equal opportunity to live in the urban core; our findings show that, over the past two decades, this has not been the case. We observe pronounced shifts in poverty away from the urban core and towards peripheral communities, mostly along freeways, where lower income workers have less access to regional transit and are more likely to drive to work than their urban counterparts. The changing identity of the suburbs undermines the fairness of policies, such as distance-based commuter rail pricing or fuel taxes designed to disincentivize driving, which have been developed and promoted under the assumption that suburbs are self-excluding enclaves of privilege. Given the state's ongoing failure to address the affordable housing crisis that continues to both displace and exclude disadvantaged populations from gentrifying urban areas, it is unjust to deprive these communities of affordable and convenient transit connections to regional job centers.

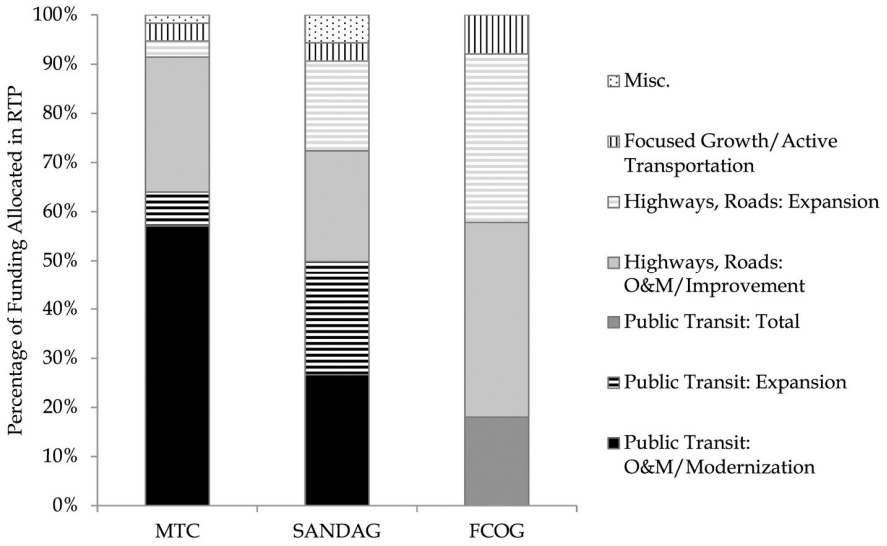

Figure 2-6 Funding allocations by each MPO for different categories of transportation projects. FCOG did not distinguish between public transportation operations & maintenance and expansion.

Through our RTP analysis, we show that our case study MPOs' current equity metrics do not adequately capture the historical and current burdens on disadvantaged people in outer suburban and exurban communities. One key way that MPOs miss the inequities experienced by communities outside the urban core is by assessing air pollution standards and VMT reduction targets solely at the regional level, despite dimensions of transportation equity, such as TRAP exposure and long car commutes, varying significantly at the local level. Attainment of NAAQS at the regional level does not preclude the possibility of hotspots within the region. Project-level CAA hot-spot analyses occur separately and sometimes at a temporal distance from the RTP, rather than as a concern integrated into regional planning. For instance, even if VMT are reduced at the regional level, near-road communities that have not received an equal share of improved transit, such as those we identified in our analysis, including Fairfield, Pittsburg, Vacaville, Antioch and parts of Fresno, all of which may continue to face long car commutes and near-road TRAP hot-spots.

The framework for comparing "communities of concern" to the general population

substitutes regional aggregation for within region heterogeneity. Burdens and benefits measured regionally for communities of concern don't capture the transit needs and burdens of low-income residents. For example, the burdens and needs in Vacaville are different from the burdens of those in the San Francisco urban core. Evaluating communities of concern as a homogenous group obscures the unique needs and burdens of populations outside the urban core.

A restorative equalization approach to equity analysis further problematizes this regional aggregation of disadvantaged communities. Just lumping these diverse communities together misses the opportunity to address the injustices of gentrification, displacement, and the often limited economic opportunities in exurban areas. Gentrification has disrupted and displaced communities and created distinct transportation issues in the suburbs and exurban periphery, such as the lack of regionally connected transit that we observe in communities like Fairfield.

To more meaningfully address equity in regional transportation planning, the dynamic interactions between climate change mitigation, air quality, transit accessibility, and housing should be incorporated throughout the analysis rather than compartmentalized as individual metrics. Exposure to TRAP is fundamentally an equity issue and should be treated as such in RTPs. The relationship between transit investment and displacement merits further investigation and should also be considered within the equity analyses of RTPs. Addressing these issues in a timely manner in the context of shifting population dynamics presents a temporality problem, wherein populations change faster than models. Another gap in the research is how transportation equity manifests and is measured in small, but growing regions, such as Fresno (Karner, 2016). As metropolitan regions expand and overlap, cities like Fresno often serve as exurban nuclei for urban core areas like San Francisco. As these cities grow, it is important to consider how the spatial dynamics of poverty and transit accessibility play out, as well as what

types of equity metrics are being assessed in these regions.

In conclusion, infill development in California risks further exacerbating inequities experienced by low-income populations that have shifted away from the urban core towards the periphery. Prioritizing transit funding for densely populated urban core areas that already have greater transit accessibility than suburban areas is likely to have regional benefits in terms of reduced regional air pollution and lower GHG emissions; however, it is also likely to accrue greater benefits to urban core areas, where disadvantaged populations have been declining. It is tempting to argue that building more affordable housing in urban areas would solve the problem. Yet, it is not at all clear that housing targets will be met and, more troublingly, much of the area that is most accessible for residential development falls outside urban core areas (Monkkonen and Friedman, 2019). What's more, many communities have already settled and established lives in cities that continue to grow on the peripheries of urban regions. An equitable approach to infill development requires recognition of these communities' existence and distinct transportation burdens and needs.

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Chapter 3 : Meta-Analysis of Particulate Emission Factors from Ships

3.1 Introduction & Background

Port cities are important centers of trade where different modes of transportation meet to move goods across sea and land. Air pollution from truck, train, and ship traffic along with industrial activity associated with ports and transportation corridors can particularly affect adjacent neighborhoods (Agrawal et al., 2009; Apte et al., 2017; Browning, Hartley, Bandemehr, Gathright, & Miller, 2012; A. Karner et al., 2009; A. A. Karner et al., 2010). Emissions of key gas-phase pollutants and particulate matter (PM) from on-road diesel trucks have been extensively characterized, allowing for the development of reasonably accurate emissions inventories for trucks to be used in air pollution models (Franco et al., 2013). In contrast, ship emission factors, especially for PM, are less thoroughly characterized, despite the well-documented contributions of ship emissions to local and global particulate pollution (Corbett et al., 2007; Dalsoren et al., 2009; Winebrake et al., 2009)

One of the primary regulatory mechanisms for reducing PM emissions from ships has been limiting the sulfur content of marine fuels. It was established as early as the 1980s that fuel sulfur content (FSC, % sulfur by weight) strongly affects the magnitude of PM emissions from fuel combustion (Baranescu, 1988; Wall, Shimpi, & Yu, 1987). Combustion of high sulfur fuels also releases sulfur dioxide gas, which reacts in the atmosphere to form secondary PM, but here we focus on primary or directly emitted PM. Consequently, international, national, and regional regulatory bodies have required location-specific reductions in FSC to curb both SO_2 and PM emissions. In 2012, the International Maritime Organization (IMO) lowered the global FSC cap from 4.5% to 3.5%, while local jurisdictions had already set more stringent limits in some sulfur emission control areas near land. The IMO's global cap was further reduced to 0.5% effective January 1, 2020 (MARPOL Annex VI, 2008 revisions; (Pirog, 2018).

In 2007, the California Air Resources Board (CARB) compiled marine engine PM emission factors from 10 studies conducted between 1994 and 2007 (Sax & Alexis, 2007). The authors found a linear relationship between FSC and PM emission factors, but with a weaker dependence than they expected for their dataset. This result could be due in part to the sensitivity of PM components other than sulfate (e.g., organic matter, elemental carbon) to factors other than FSC, such as engine characteristics and operating conditions, particularly engine load (Lack et al., 2009). Different measurement methodologies and reported emission factor units further complicate comparisons between studies. The CARB study authors recommended additional research to support a more robust investigation of the influence of FSC on PM emissions.

Since 2007, more than 30 new papers reporting PM emission factors for marine engines have been published, and increasingly stringent fuel grade regulations have led to substantially altered composition of typical marine fuels compared to the past. To comply with stricter near-shore limits on FSC, some ship operators switch from lower grade fuels with higher FSC at sea to higher grade fuels with lower FSC near shore. Many studies indicate that changing fuel grade alters the total mass and composition of particulate matter emitted by ships (Celo, Dabek-Zlotorzynska, & McCurdy, 2015; Jalkanen et al., 2012; Lack et al., 2011). Given the changing composition of marine fuels and the publication of new data since CARB's meta-analysis, a reevaluation of the available data on shipping PM emission factors is warranted.

Accurate and current PM emission factors for marine engines are useful in high-resolution air pollution modeling, particularly in port-adjacent neighborhoods. A 2016 review of progress and future directions in research on the human health effects of air pollution highlighted the importance of fine-scale modeling in urban areas, but identified the quality of emissions inventories as a significant limiting factor (West et al., 2016). In port cities, high-resolution

modeling facilitates the evaluation of shipping pollution policy interventions, such as near-shore emission control areas and shore power (provision of electricity at port facilities to eliminate the use of ship engines as power generators at berth). The accuracy of high-resolution modeling is generally limited by the ability to model turbulence from the ground surface as well as physical and chemical transformations of aerosols in the atmosphere on short timescales; however, as models of these processes improve, the accuracy of emission factors can also be a limiting factor.

Chemical transport modeling of air pollution in port-adjacent neighborhoods has been pushed to resolutions as low as 250 m, but traditional, activity-based ship emission inventories have not matched this improvement in resolution (Joe et al., 2014). Activity-based inventories, calculated by multiplying the volume of goods moved in a port by an activity factor, have limited capacity to allocate emissions spatially and temporally within the port area (Jalkanen et al., 2012). Authors compiling inventories have noted the importance of increasing the certainty of emission factor estimates (Winnes & Fridell, 2009). Real-time Automatic Information System (AIS) satellite data on ship locations, speeds, and characteristics around the globe offer an alternative to activity-based inventories, with improved spatiotemporal resolution (Jalkanen et al., 2009; Jalkanen et al., 2012). However, many AIS-based air pollution models in port cities rely on emission factors originally reported in just two studies: Lloyd's Register of Shipping (1995) and (Whall et al., 2002). The emission factors in these sources do not adequately reflect changing fuel standards over the past decade, or the conditions most relevant to near-shore emissions.

Many early emission factor datasets are oriented toward the development of global scale emissions inventories and are not directly applicable to fine-scale air pollution modeling in coastal environments that could be used to evaluate impacts on fenceline port communities or coastal ecosystems (Hobbs et al., 2000). For example, because the bulk of global ship emissions occur at

sea, emissions under high-engine-load, cruising conditions have been characterized more extensively than emissions during harbor maneuvering and at-berth operation. When ships are near or in ports, engine operating conditions are quite different than at cruising. Main engines operate at lower loads during deceleration and slow-speed maneuvering. During docking, main engines are shut off while tugboats do the maneuvering and ship power is provided by auxiliary engines, which tend to be higher speed and lower power than the main engines used for propulsion during cruising. Depending on local regulations, auxiliary engines may continue running to provide power at berth. Assessment of near-port emissions requires knowledge of emissions from main engines at low loads and from auxiliary engines (Cooper, 2003).

Here, we provide a new assessment of primary PM emission factors, incorporating numerous new measurements since the 2007 CARB report. We include data from 55 separate studies, including laboratory test-rig measurements, onboard in-stack sampling, and plume intercepts. While our primary focus is on the influence of FSC on the magnitude of PM emission factors, we also consider the sensitivity of the emission factors to engine characteristics such as speed and rated power, and operating conditions such as engine load. Finally, we identify key gaps that limit the existing data's applicability to near-port air pollution modeling.

3.2 Methods

3.2.1 Identification and Screening of Data

We identified the data for this analysis in two phases: (1) a broad search for titles indicating possible experimental measurements of PM emissions from marine engines, (2) a thorough review of full texts to confirm the presence of relevant data.

Research on ship emissions is published in government and industry reports as well as academic literature. To identify data in these different source types meeting the initial search

criteria, we conducted: (1) a query of the Web of Science database, (2) a general internet search to identify gray literature from government and industry research, (3) a review of emission factors cited in papers modeling ship emissions, and (4) a review of the cited references of papers identified using the first three methods. We did not limit by publication date. Various terminology is used to refer to both ships and particulate matter, so we conducted a series of independent searches. Terms used to identify papers characterizing emissions from ships, as opposed to on-road vehicles, included: "ship," "commercial marine vessel," "ocean going vessel," "marine vessel," "ferry," "boat," and "marine engine." Search terms used to search for particulate matter as opposed to gaseous emissions included: "particulate", "particulate matter," "particle," "PM," "$PM_{2.5}$," and "aerosol." A total of 114 papers were identified for further review.

After the initial identification and screening of papers by title, we reviewed the full text of each study to select those reporting original experimental measurements as opposed to compiled or modeled data. We did not exclude any papers based on particle size, reported units of emission factors, fuel type, engine characteristics, ship types, operating conditions, or measurement methodology. We retained papers meeting at least one of the following criteria: (1) emission factors reported in any units for any size PM, (2) some combination of data presented that could be analyzed to yield a brake-specific PM mass emission factor, or (3) methodology description indicated that data meeting criteria (1) or (2) were collected and could be requested from the authors.

3.2.2 Standardization of Emission Factor Units

Ship particulate emission factors are typically expressed as either particle mass or particle number on the basis of either fuel consumption (fuel-specific, per kg-fuel) or engine power (brake-specific, per kWh), although other units are possible. For this study, we standardized all emission

factors to brake-specific particle mass units (g-PM kWh^{-1}). The size of PM in these units is nonspecific, as discussed below. We converted fuel-specific units to brake-specific units by multiplying by the brake-specific fuel consumption (BSFC, kg-fuel kWh^{-1}). We used the study-specific BSFC, when reported, or estimated a BSFC from the CO_2 emission factor (g-CO_2 kWh^{-1}) with an assumed carbon mass fraction in the fuel of 0.865. When neither the study-specific BSFC or CO_2 emission factor were reported, we assumed a default BSFC of 0.212 kg-fuel kWh^{-1}, which is a common conversion factor and consistent with the range of BSFC reported in the papers we included.

We included mass emission factors that authors derived from integrated particle number size distributions, but we did not calculate mass emission factors in this way when the original authors opted not to. The assumption of spherical, unit-density particles likely leads to an underestimate of the mass of sulfur-dominated particles and an overestimate of particle mass for soot particles, which tend to have non-spherical shapes and size-dependent density. Accounting for the methodological uncertainty that results from these assumptions is beyond the scope of this analysis. We analyzed a subset of papers reporting both particle mass and particle number to assess the possibility of using a general, study-independent number-to-mass ratio to convert units, but a consistent relationship was not discernible.

Fifty-five studies were retained after standardizing units. The emission factor conversions are documented in Table 3-9 A full screening summary of studies we identified but ultimately excluded from this analysis is presented in Table 3-6. Some studies or emission factors meeting the inclusion criteria were excluded from specific analyses depending on the measurement characteristics. For instance, measurements of boilers and engines using emission control

technology were excluded from the main dataset. This left 53 studies and 456 emission factors in the dataset.

3.2.3 Particle Size Fraction Assumptions

PM emission factors can be reported for different particle size fractions. Particles with aerodynamic diameters less than 2.5 microns ($PM_{2.5}$) are often the preferred fraction for air pollution modeling related to human health impacts because they are small enough to enter the human respiratory tract and have regulatory standards in most jurisdictions, unlike smaller fractions such as PM_1 or ultrafine particles. The standard engine emission factor measurement methodology used in many test rig studies of marine engines, ISO8178, does not specify any particle size segregation, so the default is to measure total suspended particles (TSP, suspended particles of any diameter). However, there is no consistent size range reported across studies, presumably due to the diverse applications of these measurements.

Given the variety of reported size fractions, we investigated whether emission factors could be standardized to a common size range, $PM_{2.5}$, or whether to include emission factors for different size fractions in the same analyses. In IVL (2004), the authors assumed that $PM_{2.5}$ emission factors equal the measured TSP emission factors, justified by their assumption, guided by previous findings, that PM emissions from non-marine diesel engines are dominated by PM_1. In Entec (2010), the authors assume that primary $PM_{2.5}$ can be calculated as 90% of TSP based on Cofala (2007). Some authors assumed that there is no significant mass above 400 nm, implying that all larger size fractions (PM_1, $PM_{2.5}$, PM_{10}, TSP) are interchangeable. However, several studies have shown the presence of coarse particles in ship exhaust (Beecken et al., 2015; Moldanova et al., 2013; Peng et al., 2016).

Where an emission factor is reported for a particle size range other than $PM_{2.5}$, it is possible to calculate a $PM_{2.5}$ emission factor by assuming a typical ratio for, e.g., $PM_{2.5}$ as a fraction of PM_{10}. However, a review of studies where emission factors were reported for various particle size ranges did not show a consistent ratio. It is also theoretically possible to derive a study-specific ratio by integrating particle number size distributions and assuming uniform density, spherical particles. However, few studies reported the necessary data and there is evidence that masses calculated from integrated size distributions may not agree with gravimetric measurements, perhaps due to size-dependent density or non-spherical morphology (Moldanova et al., 2013). Attempts to standardize the reported measurements to $PM_{2.5}$-equivalent by either of these methods yielded inconsistent or unphysical results. As such, we retained emission factors in their originally reported size range.

Whether emission factors measured for different particle size ranges can be reasonably combined for use in a single analysis depends on the particle mass size distribution of PM emitted by marine engines. For instance, if all particle mass in marine engine emissions is contributed by particles smaller than 2.5 microns, then measurements for any larger size fraction (PM_{10}, TSP) would be equivalent to measurements of $PM_{2.5}$. In contrast, if particles >2.5 microns comprise a large fraction of the total PM mass, then the EF for $PM_{2.5}$ will differ substantially from that for PM_{10} or TSP. As discussed further in the results below, we find that the majority of studies indicate that most PM mass is contained in particles < 2.5 microns, or even < 1 micron, although there are a few that indicate $PM_{>2.5}$ contributes substantially. Consequently, we have combined all EF_{PM} measurements together, regardless of the specific measurement range. We use "PM" to refer nonspecifically to measurements of any of these size fractions. However, in cases where it is important to distinguish between size ranges, we use subscripts to denote the size range (either

TSP or the upper limit diameter). Some studies measured emission factors using the engine exhaust particle sizer (EEPS), which has an upper mobility diameter size limit of 560 nm. We denote the size fraction of these emission factors as $PM_{0.56}$ as they are significantly smaller than PM_1, but not small enough to be considered ultrafine. Additionally, we acknowledge that challenges exist in comparing measurements from the EEPS, which characterizes particles by mobility diameter, with some of the other measurements in which particles are characterized by their aerodynamic diameter. We do not address the distinction between different ways of measuring particle diameter here and treat them as interchangeable.

3.2.4 Treatment of Diverse Measurement Methodologies

Laboratory studies can be controlled to characterize emissions for specific engine loads and fuel types, while in-stack sampling in the field may capture more realistic conditions of ship operation, such as fuel-switching. Plume intercept studies, in which ship plumes are measured at some distance downwind from the point of emission, either on-land (Kurtenbach et al., 2016), aboard a marine research vessel (Cappa et al., 2014; Lack et al., 2011), or on an airborne platform (Beecken et al., 2015) can capture rapid atmospheric transformations of contaminants that occur over short distances from the point of emission. This makes the measurements less generalizable to other locations and conditions, but it also provides insight into the short-range transformations that occur before human exposure to ship plumes, which several authors have argued are not negligible (Beecken et al., 2015; Zhang et al., 2016). Plume measurements do not distinguish between emissions from main engines, auxiliary engines and boilers on the same ship and it is sometimes challenging to separate the plume signal from background concentrations. Plume measurements also lack the detailed data on power use and fuel consumption of the ship's engine that can be monitored during an onboard study; while it is possible to obtain this information for

specifically chosen research vessels by communication with ship operators, this practice is uncommon.

Differences in sampling instrumentation and configuration may also contribute to variability in measured EF_{PM}. For instance, mass emission factors derived measured particle number size distributions may disagree with gravimetric filter-based measurements if particle density and morphology is assumed to be independent of size, as discussed previously. For in-stack sampling, the length of the dilution tube that transfers exhaust from the stack to instruments can affect the measurements due to particle wall loss as well as dilution, which can affect the degree of condensation of semi-volatile components (Sax & Alexis, 2007). Here, we consider all measurement methodologies and instruments together, but investigate the potential influence on EF_{PM} measurements.

3.2.5 Assessment of the influence of fuel sulfur content, ship/engine characteristics, and operating conditions on emission factors

After screening and unit standardization, we compiled all the emission factors to investigate their dependence on (1) fuel sulfur content, (2) engine characteristics, and (3) engine operating conditions. The specific parameters reported within each of these three categories varied widely between studies. For instance, some studies characterized the operating conditions during testing by reporting the engine's load as the percentage of maximum rated power, while other studies reported a categorical operating mode, such as "cruising" or "idling." We grouped the reported parameters into the three categories of interest according to Table 3-1 and assessed which specific parameters we could use in our analysis by evaluating the percentage of datapoints that were reported with each parameter. We filled in missing parameters from available data where possible, as described later in this section.

Table 3-1 Summary of levels, range, and percentage of data points missing key parameters in the dataset, after filling in missing parameters where possible

Category	Parameter	Levels or Range
1. *Fuel Sulfur Content*	FSC (%S, wt.)	0-4.5
	IMO FSC tier (%S, wt.)	<0.1, 0.1-1, 1-4.5
2. *Engine Type & Characteristics*	Type/Function	Main, Auxiliary
	Stroke	2, 4
	Speed Class	Slow, Medium, High
	Maximum continuous rated power (kW)	80 – 68700
	Maximum rated speed (rpm)	90-3000
	Ship type	Cargo, tanker, dredger, fishing, naval, passenger, special craft, tug, lab test rig
3. *Engine Operating Conditions*	Operating mode	Cruising, cruising – low speed, slow steaming, maneuvering, at-berth/idling
	Engine Load (% Max. continuous rated power)	0-110
	Ship velocity (knots)	2.9-30

Fuel sulfur content was investigated as both a continuous variable (%S, wt.) and a categorical variable corresponding to the tiers defined by progressive IMO caps: <0.1%S, 0.1-1 %S, and 1-4.5 %S. Of the five studies that did not report FSC, we assigned one the maximum legal FSC for the time and location of the measurement (Westerlund, Hallquist, & Hallquist, 2015), although this is likely an overestimate (Beecken et al., 2015); we excluded the other four from our analyses of FSC as a continuous variable (BMVBS: Bundesministerium für Verkehr, 2012; Peng et al., 2016; Rideout & Radloff, 1997; Westerlund et al., 2015).

Engines were classified by type (main propulsion engine vs. auxiliary engine), stroke, speed class, maximum rated power, maximum rated speed, and ship type. Where one of these parameters was missing, we inferred values from other parameters. In some cases where the authors reported the engine make and model, we searched online for technical specification documents describing the engine's parameters, such as speed and maximum rated power. Plume intercept measurements were ascribed to main engines because, although plumes represent the sum of main engines and any other engines, main engines dominate emissions while in transit, which is when plume intercept measurements are generally taken. We assigned engine speed class based on maximum rated engine speed as: slow speed (<300 rpm), medium speed (300-1000 rpm), and high speed (>1000 rpm). There is no consistent definition of these classes across studies; where the maximum rated engine speed and reported speed class were in conflict, we reassigned the speed class according to our definitions. Engines classified as slow speed were assigned to be 2-stroke engines (all slow-speed engines in our dataset were 2-stroke). Classification of ship types is highly inconsistent across studies. We standardized ship types to the following categories based on AIS ship type categories: cargo, tanker, dredger, fishing, naval, passenger, special craft, tug. We did not use ship tonnage in any analyses due to a lack of sufficient data, and sometimes unclear specification of deadweight vs. gross tonnage.

Parameters considered to characterize operating conditions were: operating mode, engine load as a percent of maximum rated power, and ship velocity. Operating mode descriptions were inconsistent between papers, so we standardized descriptions to the following categories: cruising (>15 knots, >40% load), slow steaming (12-15 knots, 20-40% load), low speed (<12 knots, <20% engine load), maneuvering, and at-berth. Standard engine test cycles used for lab test rig studies report a weighted average emission factor. Because the weighting favors high loads characteristic

of cruising conditions, we defined the operating mode for these measurements as cruising. When a range of engine loads were reported (e.g., varying load over a real-world voyage), we used the average of the reported range for load dependence analyses. Due to the high missingness of ship velocity data (79% of emission factors), we used this parameter only to fill in missing operating mode information.

We investigated the effect of FSC, engine characteristics, and operating conditions on PM emission factors, as well as their potential interactions, by linear regressions and hypothesis tests, with an emphasis on the role of FSC. To complement our meta-analysis, we also synthesized the findings of individual studies specifically investigating each parameter of interest.

3.3 Results

The results presented in this section describe the dataset after filling in missing values based on the assumptions detailed in the methods and after excluding emission factors measured for boilers and for engines employing emissions control technology, such as selective catalytic reduction. We begin with the relationship between FSC and EF_{PM}; fishing ships and tugs are excluded from this overall analysis due to their higher emission factors relative to other ship types, as elaborated in a later section. Then, we examine how other variables such as engine characteristics and operating conditions can contribute to variability in the relationship between FSC and EF_{PM}. Finally, we consider how measurement methodology and data reporting might have affected our results.

3.3.1 Influence of fuel properties, engine characteristics, and operating conditions on PM emission factors

3.3.1.1 Fuel properties

FSC was reported for all but 17 emission factors in our dataset (Table 3-2), and ranged from 0-4.5%S. The clustering of very high and very low FSC that Sax & Alexis (2007) hypothesized as a factor in the weak relationship observed between FSC and EF_{PM} was less pronounced with our addition of more recent data (Figure 3-1).

Table 3-2: Summary of distribution of data with respect to fuel sulfur content, by studies (one study may be counted in multiple categories if reporting emission factors for multiple fuels) and individual emission factors.

FSC	***Studies***	***Emission Factors***
<0.1%S	28	145
0.1-1%S	26	222
>1%S	31	150
Missing	4	17

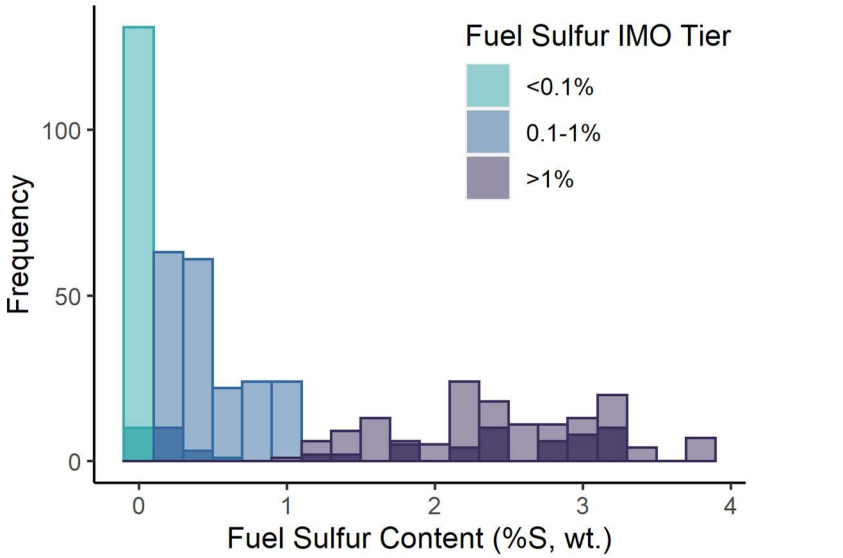

Figure 3-1: Histogram of fuel sulfur content, colored by fuel grade. Data included in Sax and Alexis (2007) are shown in a darker shade.

Through an analysis of variance (ANOVA) and Tukey's honest significant differences (HSD) test, we found that the mean and geometric mean EF_{PM} for the three FSC tiers all vary significantly ($p<0.001$) from one another, with EF_{PM} increasing at higher FSC (Figure 3-2). The geometric mean EF_{PM} was 1.2 g-PM kWh^{-1} for FSC >1%S, 0.4 g-PM kWh^{-1} for 0.1-1%S, and 0.1 g-PM kWh^{-1} for <0.1%S (Table 3-4). These values correspond to a 67% decrease in EF_{PM} for switching from >1%S fuel to 0.1-1%S fuel, and a 92% decrease for switching from >1%S to <0.1%S.

Table 3-3 Median and geometric mean EF_{PM} for different fuel sulfur content tiers across all emission factors

Fuel Sulfur Tier (*%S, wt.*)	**Median EF_{PM}** (*g-PM kWh^{-1}*)	**Geometric Mean EF_{PM}** (*g-PM kWh^{-1}*)
>1	1.4	1.2
0.1 - 1	0.4	0.4
<0.1	0.1	0.1

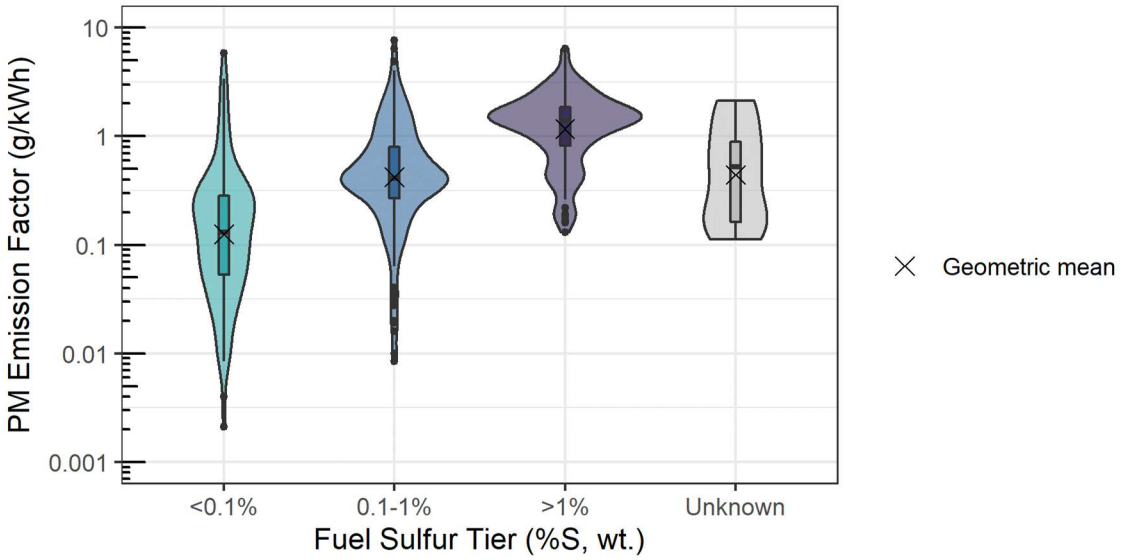

Figure 3-2 Violin plot of measured EF_{PM} at different fuel sulfur tiers for the entire dataset. The width of the violin plot indicates density of data. Inside each violin plot is a standard boxplot showing the median emission factor in each group and the x indicates the geometric mean.

A linear regression of all emission factors with FSC in our dataset yielded a statistically significant relationship ($p < 0.001$) with equation $EF_{PM} = 0.3 + 0.46 \cdot FSC$. Despite the significant linear relationship, the relationship is relatively weak ($R^2 = 0.34$), as also found in the previous critical review (Sax & Alexis, 2007). Given the wide range of EF_{PM} values we also fit a log-linear model to the data, with equation $0.18e^{0.755 \times FSC}$ ($R^2 = 0.36$). While the R^2 values between the linear and log-linear models are comparable, the sum of squared residuals for the linear model (206) is smaller than the sum of squared residuals for the log-linear model (243), when calculating residuals on a linear scale. The regression results are nearly identical to the models fit in Sax & Alexis (2007); the main difference is that the new data have marginally higher mean EF_{PM} at low FSC (Figure 3-3, Table 3-4).

Table 3-4 Linear and Log-Linear Model fits for Sax & Alexis (2007) and the current dataset, also showing the number of emission factors (n_{EF}) and the number of studies ($n_{studies}$) for each model.

Dataset	**n_{EF}**	**$n_{studies}$**	**Linear Model**	**Log-Linear Model**
CARB (2007)	75	5	$0.465 \times \mathbf{FSC} + 0.25$ ($R^2 = 0.51$)	$0.17e^{0.774 \times \mathbf{FSC}}$ ($R^2 = 0.52$)
Current study*	471	50	$0.466 \times \mathbf{FSC} + 0.29$ ($R^2 = 0.37$)	$0.18e^{0.755 \times \mathbf{FSC}}$ ($R^2 = 0.36$)

**Fishing ships and tugs excluded*

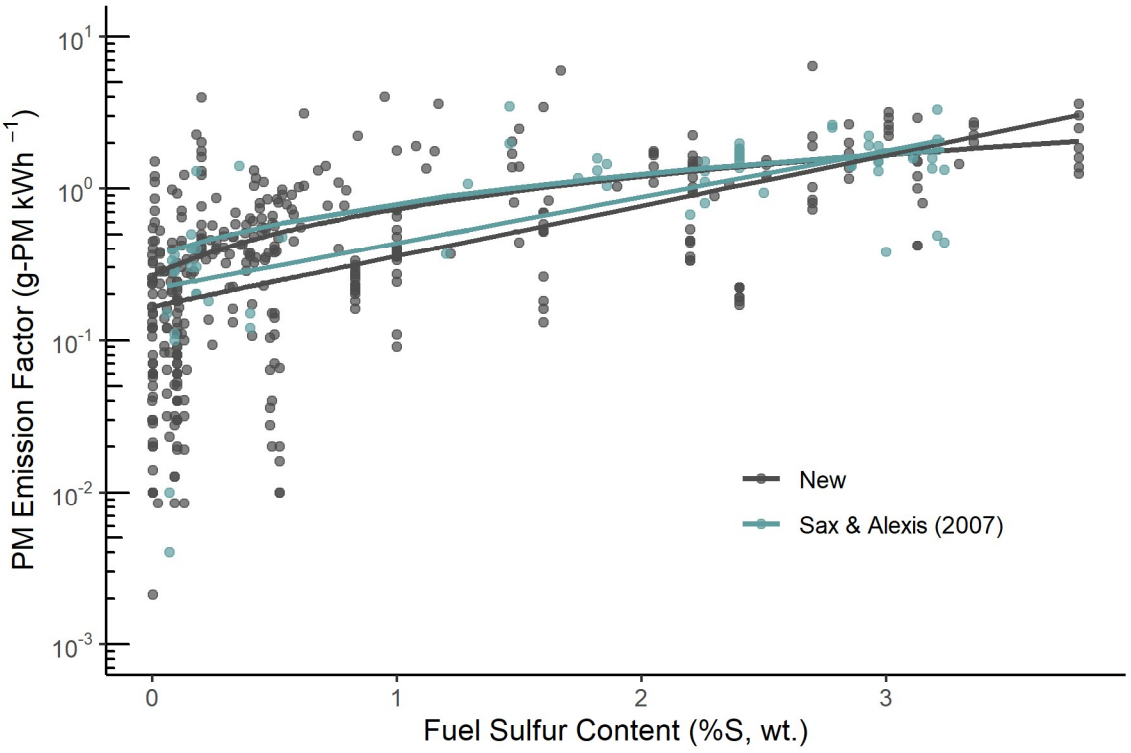

Figure 3-3 Linear and log-linear regression models of EF_{PM} vs. FSC fit separately to the data included in Sax & Alexis (2007) and the present study; both regressions shown on a log scale.

In sixteen studies, authors reported emission factors for multiple fuel grades (>0.2%S difference) tested in the same engine (Table 3-5). Some of these studies were conducted under controlled laboratory settings, while others were conducted onboard ships. Several of the onboard studies measured the effect of real-world fuel switching, a strategy some ship operators use to comply with more stringent regulations in protected near-shore areas, while continuing to use less refined and less costly fuels outside these areas. Other onboard studies characterized emission factors for real-world operation of the same engine running on two fuel grades, but with controlled engine loads and a stabilization period before measuring the effect of the fuel switch, in some cases during two separate measuring campaigns. Observed percent reduction of EF_{PM} in these multi-fuel studies ranged from 25% to 98%; in 14 of the 16 studies EF_{PM} was reduced by more than 50%. Some of the variability is attributable to different ranges in fuel sulfur content between the two

tested fuels, and some to other conditions, such as engine load or fuel properties, which can affect combustion efficiency (Table 3-5).

Table 3-5 Summary of percent reduction of $EF_{PM2.5}$ for studies where emission factors were measured for different (≥0.1%S difference) fuel grades in the same engine.

Study Type	Change in %S, wt.	PM Size Fraction	% Reduction EF_{PM}	Source
Onboard, real-world fuel switch	3.15 – 0.07	PM_1	>90%	(Lack et al., 2011)
	3.79 – 0.01	$PM_{2.5}$	~70%	(Browning et al., 2012)
	2.51 – 0.17	$PM_{2.5}$	50-70%	(Khan et al., 2012)
Onboard, controlled fuel switch	1.6 – 0.03	TSP	~44%	(Winnes & Fridell, 2009)
	1 – 0.1	$PM_{2.5}$	85%	(Moldanova et al., 2013)
	1.62 – 0.12	$PM_{2.5}$	64%	(Celo et al., 2015)
	1 – 0.5	$PM_{2.5}$	42%	(Winnes, Moldanova, Anderson, & Fridell, 2016)
	1 – 0.1	$PM_{2.5}$	83%	
	0.48 – 0.09	TSP	67%	(Zetterdahl, Moldanova, Pei, Pathak, & Demirdjian, 2016)
	0.5 – 0.1	TSP	25%	(Cooper, 2003)[a]
Lab test engine	2.36 – 0.76	TSP	62%	(Maeda, 2004)
	0.83 – 0.10	$PM_{2.5}$	27-88%	(Sarvi, Lyyranen, Jokiniemi, & Zevenhoven, 2011)
	2.7 – 0.1	-	67-97%	(Sippula et al., 2014)
	0.12 – 0.0003	$PM_{0.056}$	86-98%	(Anderson, Salo, Hallquist, & Fridell, 2015)[b]
	1.6 - <0.001	$PM_{2.5}$	77%	(Mueller et al., 2015)
	1.6 - <0.001	-	~60%	(Streibel et al., 2017)
	3.36 – 0.1	TSP	>90%	(Zhou, Zhou, & Zhu, 2017)

(a) *Different engines of the same model on the same ship, with different fuels*
(b) *Percent reduction corresponds to a change from MDO (FSC = 0.83) to averaged values for MK1 & MK3 (FSC = 0.10). The HFO (FSC = 0.12) from this study is not represented in this table, but is discussed in the text.*

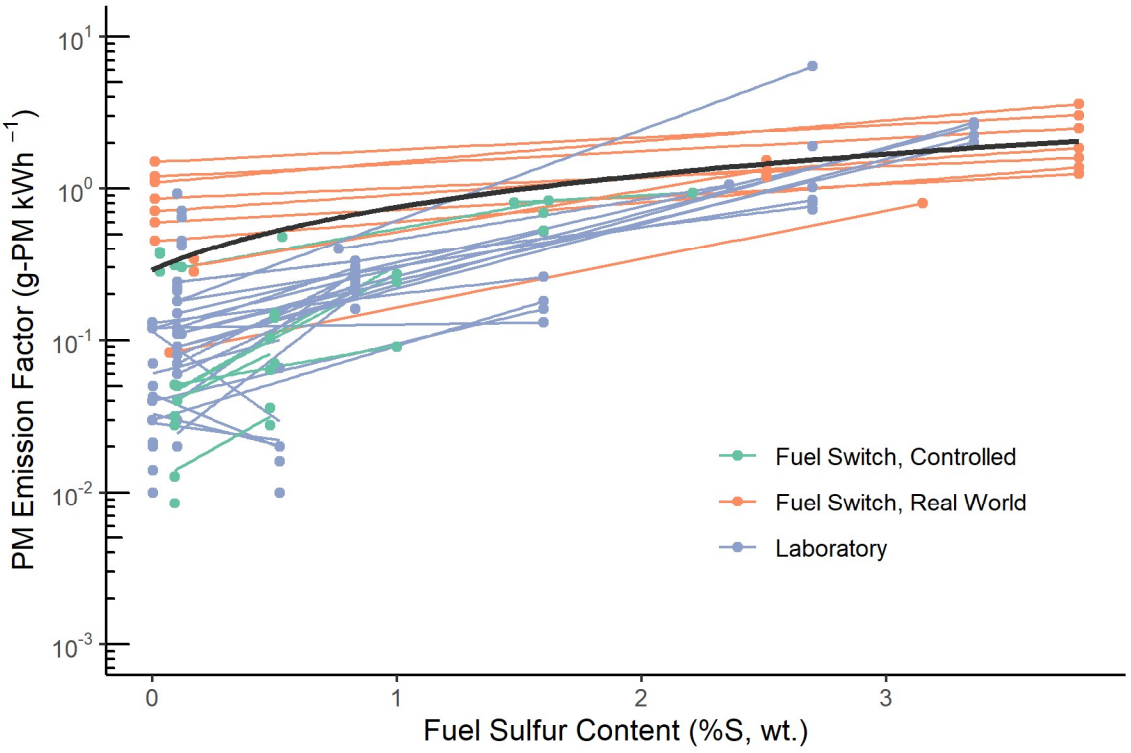

Figure 3-4 Linear regressions of EF_{PM} vs. FSC for individual measurements on the same engine with different fuel grades, colored by measurement methodology, and with the overall regression for all studies shown in black. All regressions displayed on a log-scale axis. Real world fuel switch studies involve onboard measurements as the fuel grade is switched under real-world conditions, which may result in temporary mixtures of the two fuel grades and concurrent ship speed changes. In these instances, a series may include different fuel grades and different engine loads. Controlled fuel switch studies involve onboard measurements of two fuel grades in the same engine with a stabilization period between measurements, sometimes achieved during two separate measurement campaigns. Laboratory studies involve measurements of different fuel grades on the same test engine under controlled laboratory settings. In these instances, series are separated by engine load.

In just one study, a lower-sulfur fuel was associated with a higher EF_{PM} than was a higher-sulfur fuel (Anderson et al., 2015). In this study, a low-sulfur (0.12%S) heavy fuel oil was compared to a diesel oil that had a higher sulfur content (0.52%S) but was otherwise more refined (e.g., lower viscosity and lower concentrations of carbon residue and trace metals).

3.3.1.2 Engine and Ship Type

Engine type (main vs. auxiliary) was reported for 95% of emission factors. Main engines were far more frequent in our dataset than auxiliary engines (86% of emission factors), making a

statistical comparison between these two groups difficult. Engine stroke was more evenly distributed between 2-stroke (23%) and 4-stroke (36%) but had a larger percentage of missing data (40%). Medium speed engines were the dominant speed class (Table 3-6, Figure 3-5). Engine maximum rated power ranged from 80 to 68,700 kW, but most of the engines were between 100 and 3,000 kW. The most frequently measured ship type was cargo, followed by passenger ships and then tankers. Tugboats, dredgers, naval vessels, and "special craft" were observed, but not with the same frequency as larger, ocean-going ship types (Table 3-7**Error! Reference source not found.**).

Table 3-6 Summary of engine speed classes in dataset

Speed Class	***Studies***	***Data Points***
SS	17	85
MS	25	163
HS	20	133
Missing	16	75

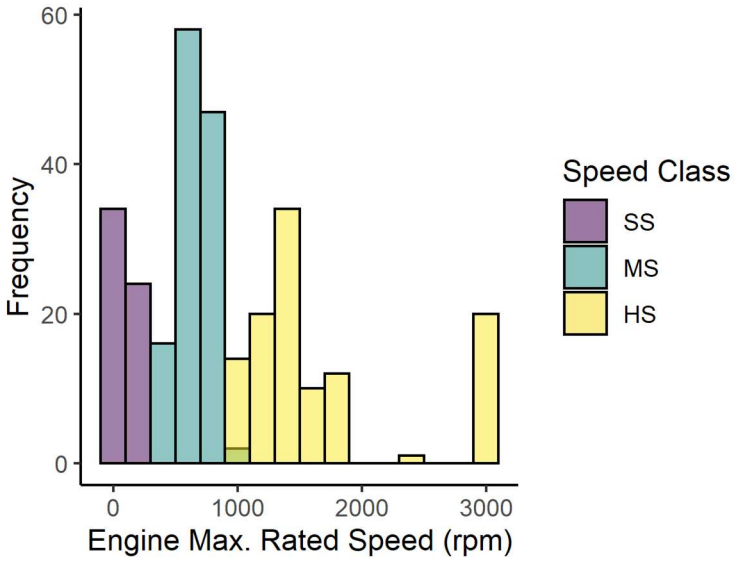

Figure 3-5 Histogram of engine speeds, separated by engine speed class.

Table 3-7: Summary of ship types in dataset

Ship Type	***Studies***	***Data Points***	***Specific Ship Types***
Cargo	25	135	Cargo, carrier, container, RoRo
Tanker	11	47	Chemical tanker, oil tanker, product tanker
Dredger	1	1	-
Fishing	1	26	Angling, trawler, gill net
Naval	2	20	Naval, frigate, surveillance
Passenger	18	68	Ferry, RoPax, passenger liner
Special Craft	4	29	Dock landing, research vessel, maintenance, engineering vessel
Tug	7	21	Tug, pilot tender, push tow
Laboratory Test Rig	6	49	-
Missing	12	68	-

Emission factors for slow-speed engines tend to be clustered around high FSC and those for high-speed engines tend to be clustered around low FSC, weakening the regressions for these two categories (Figure 3-6).

An ANOVA test showed that emission factors for 2-stroke engines were significantly ($p<0.001$) higher than those for 4-stroke engines (

Figure **3-7**). Similarly, an ANOVA test showed that adding a stroke term to the model of EF_{PM} as a function of FSC statistically significantly ($p<0.001$) reduces the residual sum of squares relative to a model with only FSC (Figure 3-8).

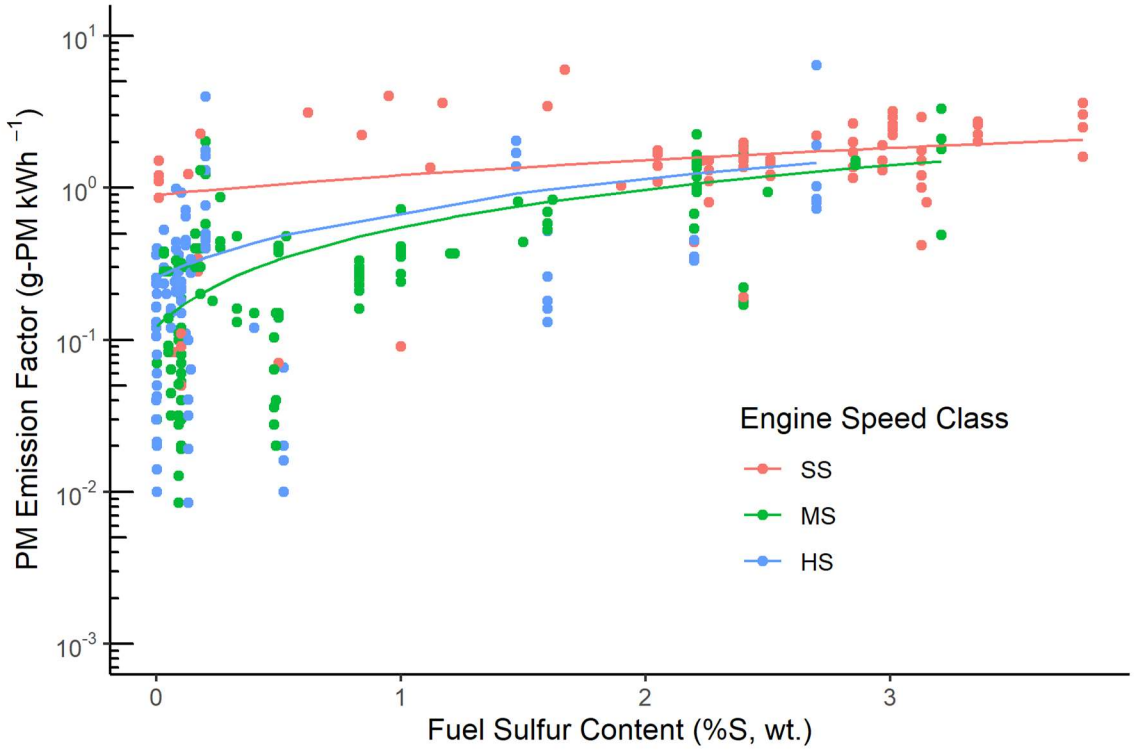

Figure 3-6 Regression of EF_{PM} vs. FSC, stratified by engine speed class and displayed on a log-scale axis.

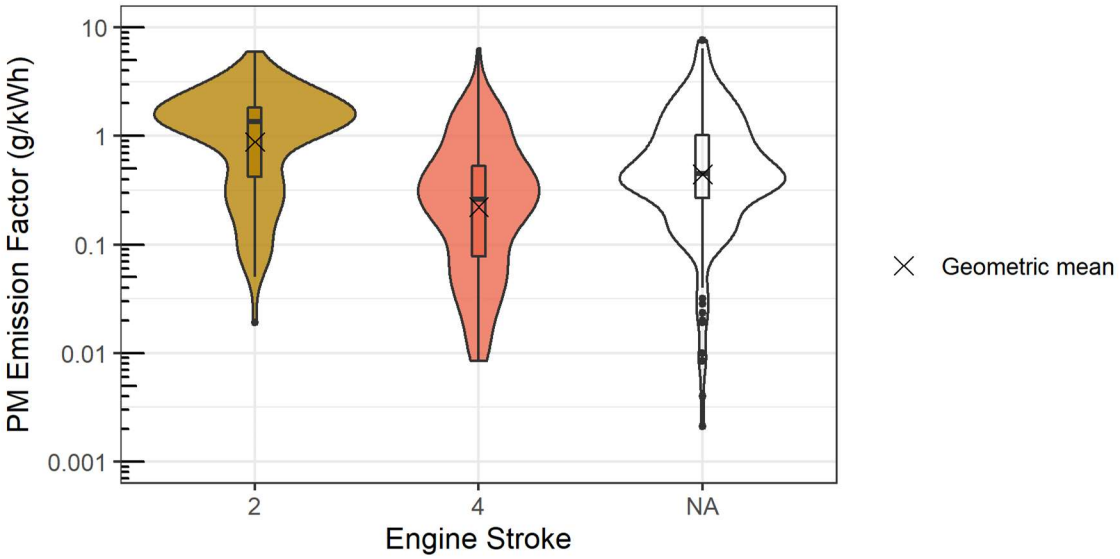

Figure 3-7 Violin plots showing EF_{PM} for 2-stroke vs. 4-stroke engines, with geometric mean emission factors indicated with an x.

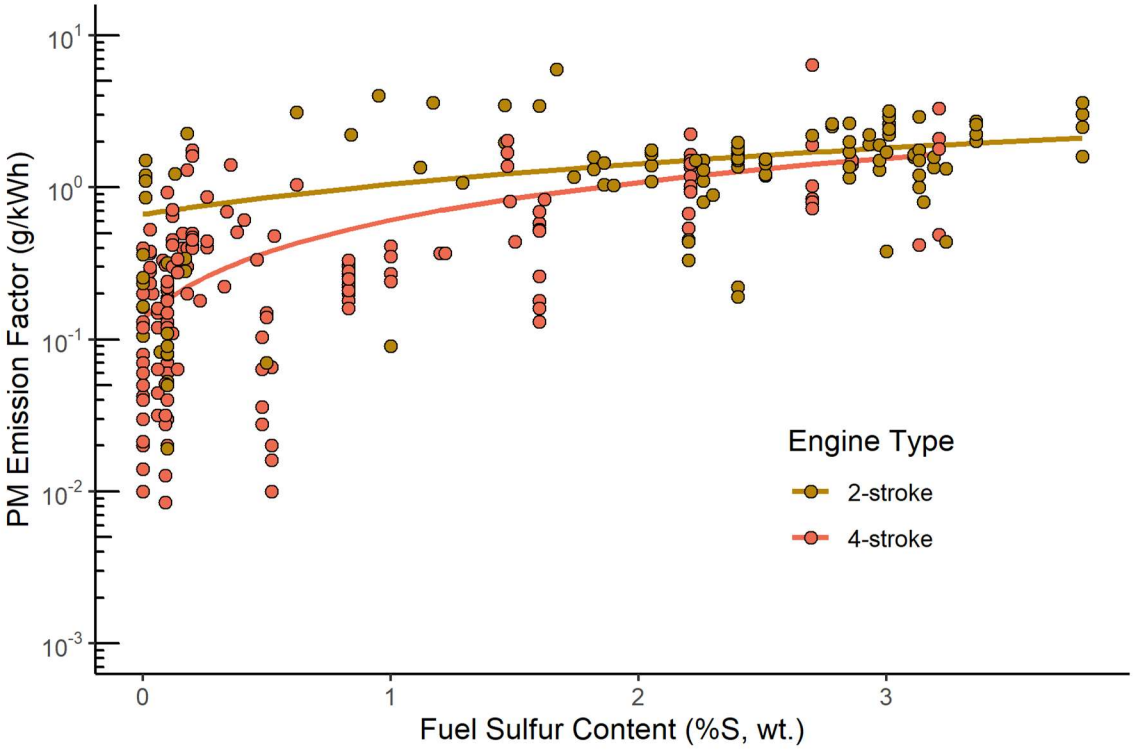

Figure 3-8 Regression of EF_{PM} vs. FSC, stratified by engine stroke and displayed on a log-scale axis.

Fishing boats, measured in only one study, had statistically significantly ($p < 0.001$) higher emission factors than other ships measured with similar fuel grades (Figure 3-9, Figure 3-10). Tugs also stand out from other ship types because of the relatively higher emission factors given the low FSC fuels measured (Figure 3-9, Figure 3-10). For this reason, these two ship types were excluded from the overall regression presented at the beginning of the results section. These two ship types should likely be considered uniquely in any emissions inventory.

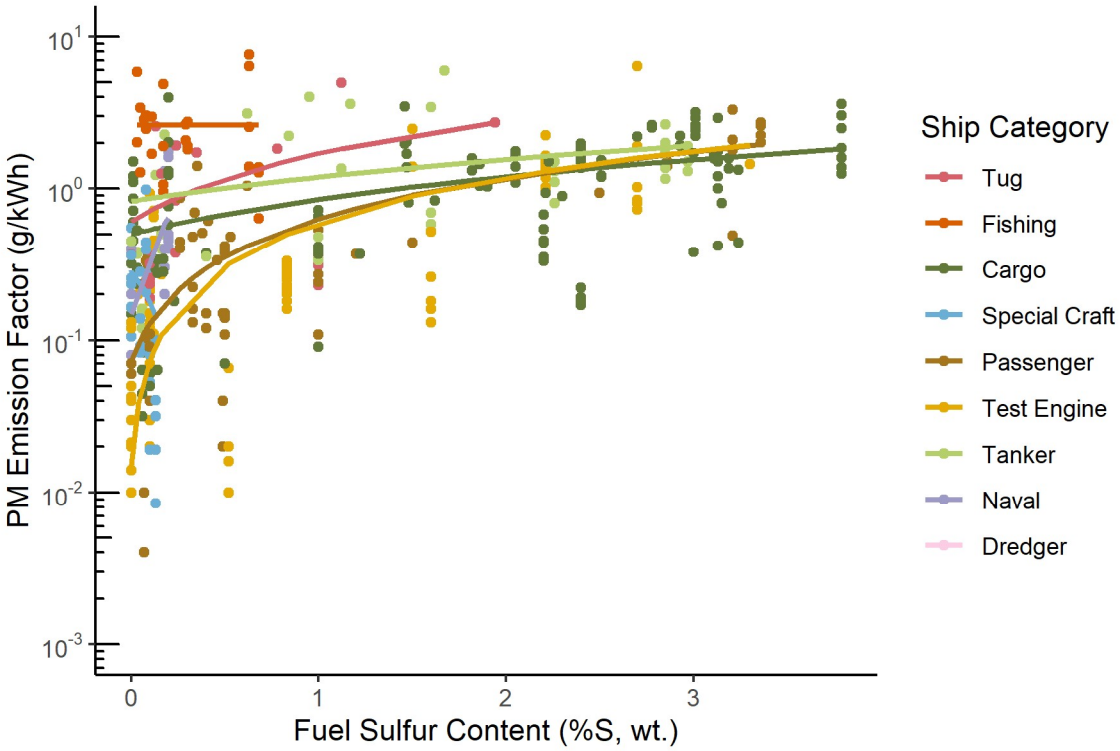

Figure 3-9 Regression of EF_{PM} vs. FSC, stratified by engine stroke and displayed on a log-scale axis.

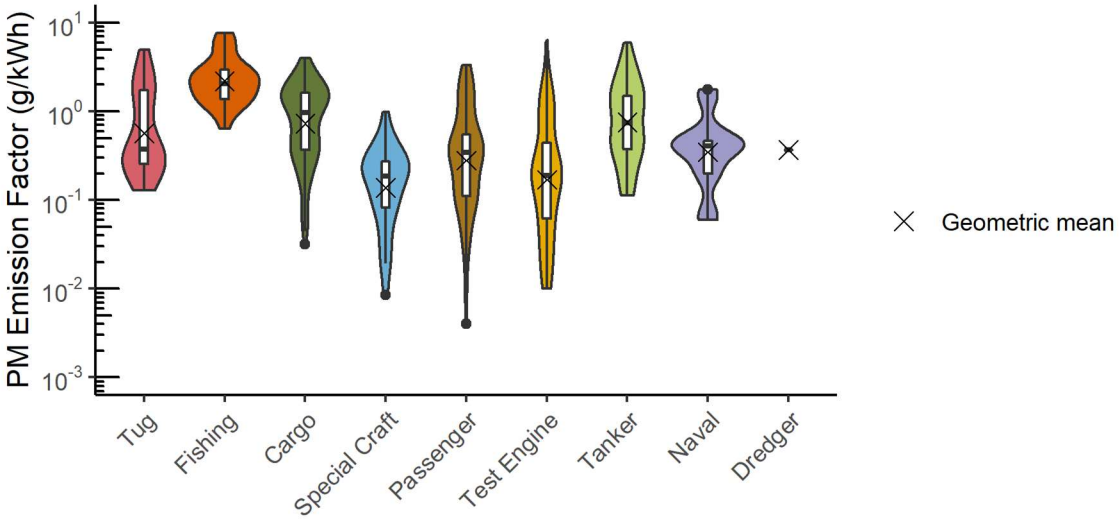

Figure 3-10 Violin plots showing EF_{PM} for different ship type categories, with geometric mean emission factors indicated with an x.

3.3.1.3 Operating Conditions

Variables describing an engine's operating mode were reported less consistently than engine characteristics: 47% of data points were not associated with an operating mode and 40% did not have an engine load. A majority (63%) of the emission factors with an associated operating mode fall into one of the two cruising categories. Only 22% of the reported emission factors characterize in-port maneuvering and at-berth/idling emissions (Table 3-5).

Table 3-8: Summary of operating modes in dataset

Operating Mode	*Studies*	*Data Points*
Cruising *(>15 knots, >40% load)*	29	107
Cruising, low speed *(12-15 knots, 20-40% load)*	11	43
Slow Steaming *(<12 knots, <20% load)*	8	37
Maneuvering	7	37
At-Berth/Idling	4	16
Unknown	27	216

Some authors investigating the effect of engine load found that EF_{PM} generally increases with engine load or vessel speed (Lack et al., 2011). However, other studies suggested that at very low loads, the brake-specific emission factor may increase relative to emission factors at medium loads due to direct emission of unburned residuals (Streibel et al. 2017) or inefficient combustion due to load fluctuations during low-load operations such as maneuvering (Peng et al., 2016). In studies where emission factors were measured at loads far below rated power (<25% MCR), a majority indicated that emission factors increased at least modestly at very low engine loads and idling conditions, although this pattern was not apparent in all such studies (Figure 3-11, Figure 3-12).

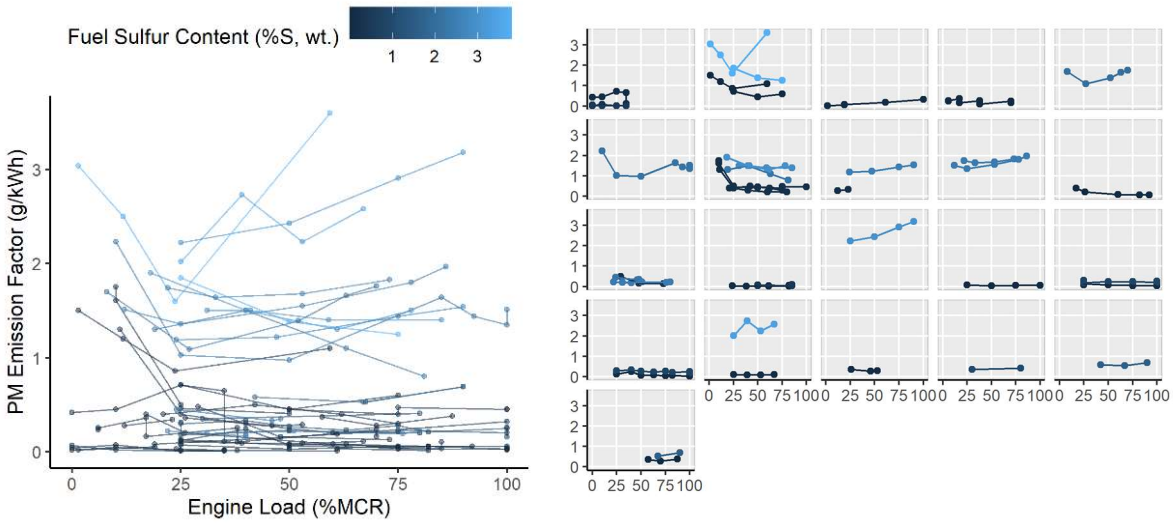

Figure 3-11 EF_{PM} vs. engine load for engines with measurements at multiple loads, colored by fuel sulfur content. Data are plotted together (left) and separated by paper (right). Source papers from top left are: Anderson, Salo et al. (2015); Browning, Hartley et al. (2012); Cappa, Williams et al. (2014); Betha, Russell et al. (2017); Agrawal, Malloy et al. (2008); Petzold, Weingartner et al. (2010); Wright (1997); Khan, Giordano et al. (2012); Man B&W (2004); Khan, Russell et al. (2012); Fridell, Steen et al. (2008); Zetterdahl, Moldanova et al. (2016); Agrawal, Welch et al. (2010); Kalender and Ergin (2017); Sarvi, Fogelholm et al. (2008); Sarvi, Lyyranen et al. (2011); Zhou, Zhou et al. (2017); Cooper (2001); Moldanova, Fridell et al. (2013); Winnes and Fridell (2010); Winnes and Fridell (2009)

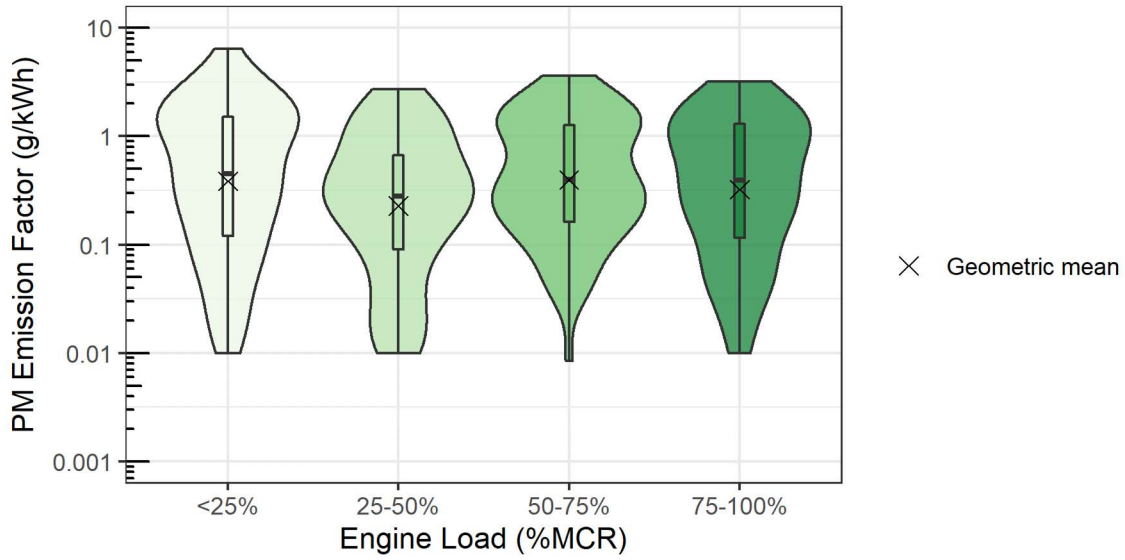

Figure 3-12 Violin plot of EF_{PM} for different engine load ranges, with geometric means indicated by an x.

3.3.2 Measurement Methodology and Reported Particle Size

3.3.2.1 Measurement Methodology

Plume intercept measurements yielded statistically significantly higher EF_{PM} values than in-stack or exhaust measurements measured onboard for FSC >1%, according to an ANOVA and Tukey's HSD test ($p = 0.001$), but there are relatively few plume measurements for FSC in this range and the difference is not statistically significant for other FSC tiers (Figure 3-13, Figure 3-14).

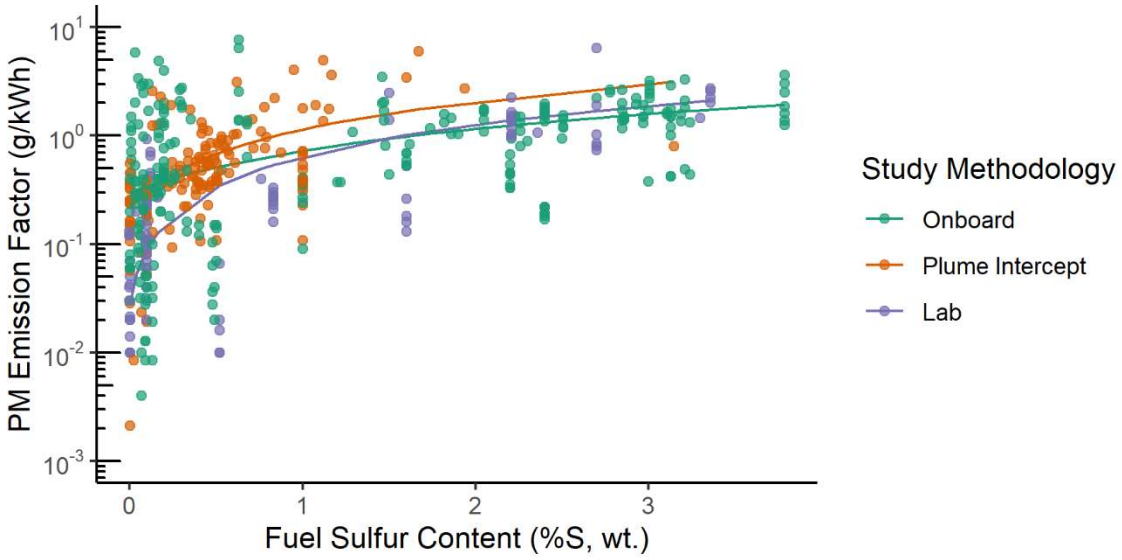

Figure 3-13 Linear regression of EF_{PM} vs. fuel sulfur content, stratified by measurement methodology and displayed on a log-scale axis.

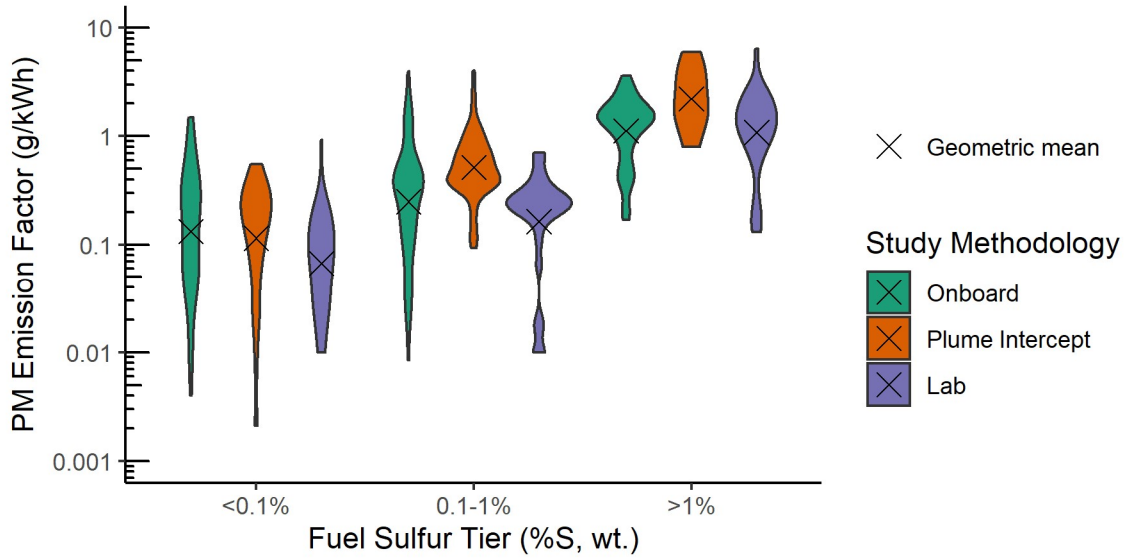

Figure 3-14 Violin plot of EF_{PM} by methodology and fuel sulfur content tier

3.3.2.2 Particle Size

The most frequently reported particle size fraction was $PM_{2.5}$ (28%), followed by TSP (23%). We could not determine the measured size fraction for 23% of emission factors. We assessed the size-dependent relative contribution to $PM_{2.5}$ from 31 cumulative mass size distributions reported in eleven papers. The distributions we included indicate that there may be significant particle mass contributed by particles with diameters larger than 1 micron, with 9 of the 31 distributions suggesting that PM_1 mass is less than 75% of $PM_{2.5}$ mass (Figure 3-3). This could lead to an underestimate if a measured PM_1 factor is assumed to be equivalent to a $PM_{2.5}$ factor. Similarly, size distributions from 6 of the 11 studies reach a cumulative mass percent greater than 125%, suggesting that using PM_{10} or TSP measurements as $PM_{2.5}$ emission factors could lead to overestimates of 25% or greater (Figure 3-3). However, the majority of measurements indicate that $PM_{2.5}$ comprises at least 50% of total measured particle mass. Therefore, emission factors for

size fractions larger than $PM_{2.5}$ should be considered an upper limit and emission factors for smaller fractions should be considered a lower limit for $PM_{2.5}$ emission factors.

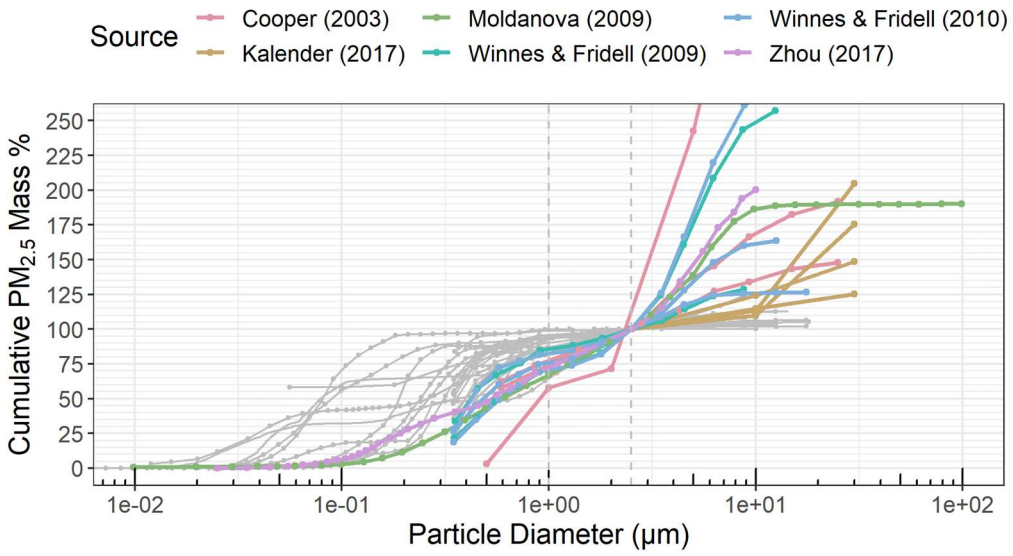

Figure 3-15 Cumulative mass size distributions of ship particulate emissions normalized to $PM_{2.5}$ mass, with source papers indicated by color for distributions where the mass of particles larger than 2.5 µm is greater than 25% of $PM_{2.5}$ mass. Vertical lines indicate 1 µm and 2.5 µm diameters. Two series extend beyond the bounds of the plot and reach maxima of 275% (Winnes & Fridell, 2010) and 750% (Cooper, 2003).

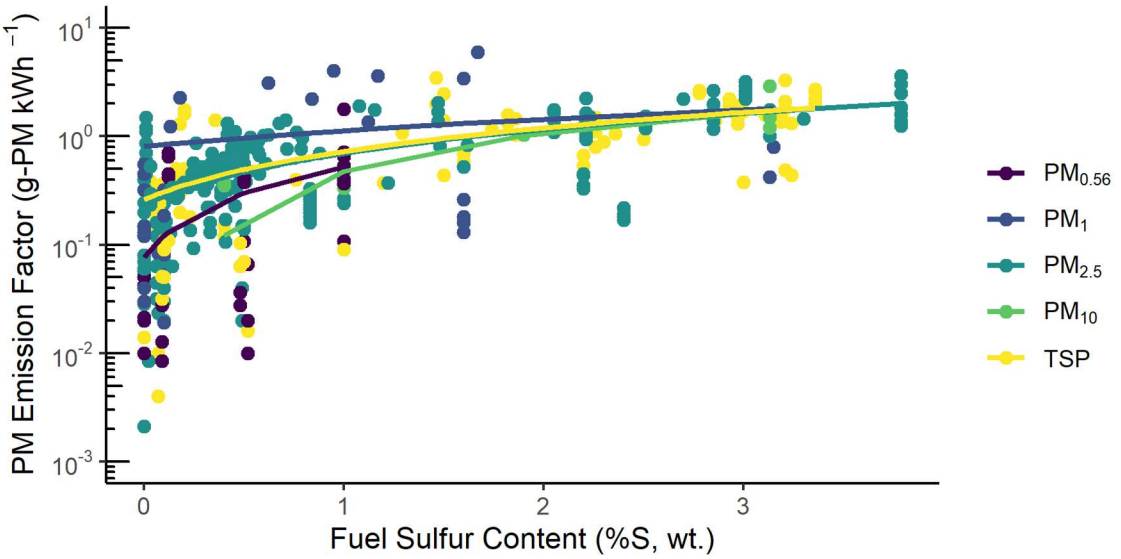

Figure 3-16 Linear regression of EFPM vs. FSC, stratified by measured particle size range, displayed on a log-scale axis.

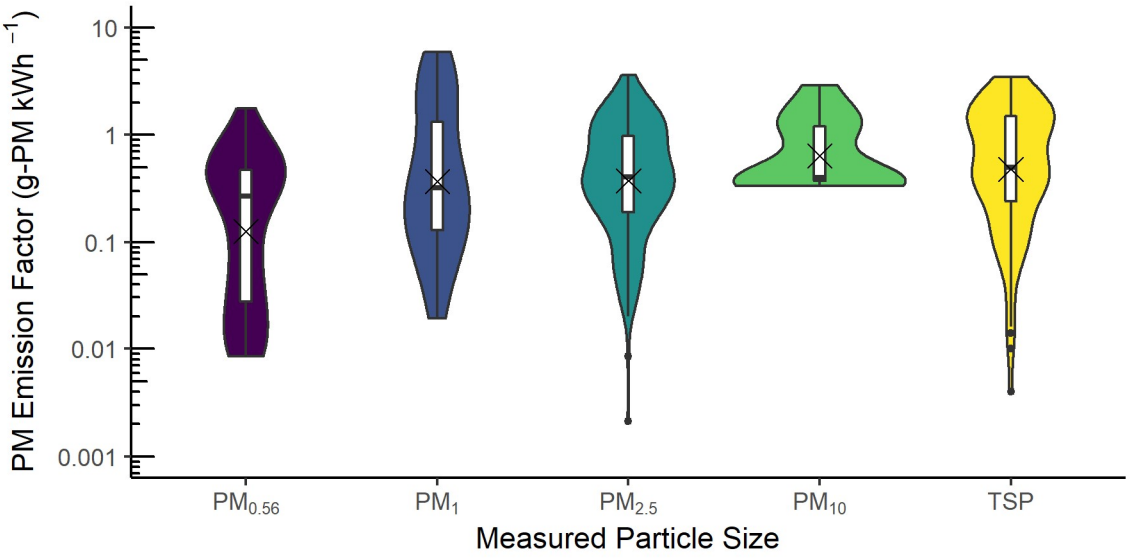

Figure 3-17 Violin plot of EF_{PM} by measured particle size range.

3.4 Discussion & Conclusions

Our analysis of individual studies and the overall dataset support the assumption that limiting FSC has the potential to significantly reduce EF_{PM}. In individual studies that tested multiple fuel grades on the same engine, a switch from high to low FSC consistently reduced $EF_{PM,}$ in most cases by more than 50% and in some cases up to 98%. In the overall dataset, EF_{PM} was statistically significantly positively correlated with FSC, and increasing tiers of FSC had increasing geometric mean EF_{PM}. The clearest mechanism for the correlation between FSC and EF_{PM} is that lower-sulfur fuels have less sulfur to transfer to the particulate phase, but the reduction may also be explained by the fact that that these fuels typically have lower concentrations of transition metals, more efficient combustion resulting in less emission of unburned residuals, and diminished need for lubricating oils that can be emitted as particulate matter (Sippula et al., 2014).

Switching from a high-sulfur residual (unrefined) fuel to a low-sulfur residual fuel may reduce EF_{PM} less than switching to a low-sulfur distillate (more refined) fuel. For fuels with <1%S,

other fuel components and properties may in some cases cause a higher-sulfur fuel to have a lower EF_{PM} than a lower-sulfur fuel as in Anderson, Salo et al. (2015), but this effect is difficult to capture at the meta-analysis scale due to inconsistent reporting of fuel grade between papers. A review of the effect of fuel components other than sulfur on EF_{PM} was beyond the scope of this study, but merits further investigation, especially if some ship operators choose to comply with new fuel sulfur regulations by switching to low-sulfur residual fuels.

The geometric mean EF_{PM} values presented here are similar to those reported in Sax and Alexis (2007), as well as those currently recommended by the California Air Resources Board. The linear regression model fit to the updated data was nearly identical to the one presented in the previous critical review, although the relationship was even weaker with the additional data (Sax & Alexis, 2007). This may be due in part to the greater variety of methodologies and conditions of the more recent measurements allowed for a more detailed investigation of additional factors that might influence EF_{PM}.

Across all studies, larger two-stroke engines were found to have statistically significantly higher EF_{PM}. A comparison of different engine speed classes (slow, medium, high) was difficult due to the confounding relationship with FSC. In analyzing the potential effects of ship type, we found that tugs were underrepresented and had higher emission factors than other ships with similar fuel grades. A more thorough characterization of emissions from tugs and other pilot craft would be useful for near-port modeling as they are abundant in the coastal setting.

We found that EF_{PM} generally increased with engine load, but that in most cases at very low loads (<25% MCR), EF_{PM} increased relative to medium loads. This is likely a reflection of combustion efficiency, which is greatest at medium loads. It is important to note that a higher brake-specific EF_{PM} does not result in higher absolute emissions in a low-load scenario, because

the power consumed will be lower. Still, there is evidence that ship-emitted particles can be high near areas where ships berth, so we recommend further measurements of the low-engine-load conditions that dominate ship emissions in port.

Measurement methodologies were diverse and, in some cases, vaguely described. This difference could be explained in part due to the extra time for chemical transformations that produce secondary PM to occur between leaving the stack and interception. Plume intercept studies also tended to rely more on integrated number size distributions to calculate $EF_{PM,}$ which have been shown to introduce both positive and negative biases due to the necessary assumptions about particle morphology and density (Moldanova et al., 2013).

The variety of measurement methodologies and the greater dependence of EF_{PM} on combustion conditions (as compared to, e.g., NO_x or CO_2), necessitate further measurements of underrepresented ship/engine types, operating modes, and fuel types. Additional studies characterizing conditions and vessel types typical of near-shore emissions (e.g., maneuvering, auxiliary engines, fuel switching, and pilot craft) have the potential to improve the spatiotemporal resolution of ship emission inventories in support of emissions modeling in port communities and near-shore ecosystems. We also recommend that future studies reporting PM emission factors clearly specify sampling instrumentation and its configuration, sampling conditions, and the size of particles measured. Finally, as understanding of the human health impacts of specific air pollutants advances, it will be increasingly important to measure other parameters in addition to $PM_{2.5}$, such as ultrafine particles, chemical composition of particles, and emission factors of air toxics.

3.5 Supplementary Information

Table S3-1 Table indicating papers that were reviewed, but ultimately excluded from the meta-analysis. The point in the review process at which the study was excluded and the reason for exclusion are also noted.

Paper Title	Year	First Author	Point of Exclusion	Reason excluded
Marine Exhaust Emissions Research Programme	1995	Lloyd's Register	Full Text	Duplicate data with Wright (1997)
Air Pollutant Emissions from ships: high Tyrrhenian Sea ports case study	1998	Trozzi	Abstract	No original measurements of EF_{PM}
Emissions from ships with respect to their effects on clouds	2000	Hobbs	Full Text	Study reports only number emission factors, integration of size distributions yielded unphysical results
Airship measurements of ship's exhaust plumes and their effect on marine boundary layer clouds	2000	Frick & Hoppel	Full Text	Duplicate data with Hobbs 2000
A study on exhaust gas emissions from ships in Turkish Straits	2001	Kesgin & Vardar	Abstract	No original measurements of EF_{PM}
Emission from international sea transportation and environmental impact	2003	Endresen	Abstract	No original measurements of EF_{PM}
Emissions of trace gases and particles from two ships in the southern Atlantic Ocean	2003	Sinha	Full Text	Study reports only number emission factors, integration of size distributions yielded unphysical results
Particle Emissions from Ship Engines	2004	Petzold	Full Text	Not enough data included to calculate EF_{PM}
Particle Emissions from Ship Engines: Emission Properties and Transformation in the Marine Boundary Layer	2006	Petzold	Full Text	Not enough data included to calculate EF_{PM}
An investigation of the chemistry of ship emission plumes during ITCT 2002	2005	Chen	Full Text	Study reports only number emission factors; no straightforward way to convert to mass emission factors.
Identification and characterization of inland ship plumes over Vancouver, BC	2006	Lu	Full Text	Study reports ambient measurements, not measurements of ship emissions

Positive matrix factorization and trajectory modelling for source identification: A new look at Indian Ocean Experiment ship observations	2008	Bhanuprasad	Abstract	No original measurements of EF_{PM}
Light absorbing carbon emissions from commercial shipping	2008	Lack	Abstract	Emission factors are reported for light-absorbing carbon, not total PM.
Emission Factors in Inland Navigation	2008	Matei	Full Text	No original measurements of EF_{PM}
Emissions from large-scale medium-speed diesel engines: 1. Influence of engine operation mode and turbocharger	2008	Sarvi	Full Text	Duplicate data with Sarvi et al. (2008), Part 2
Emissions from large-scale medium-speed diesel engines: 2. Influence of fuel type and operating mode	2011	Sarvi	Full Text	Duplicate data with Sarvi et al. (2011), Part 1
Primary Particulate Matter from Ocean-Going Engines in the Southern California Air Basin	2009	Agrawal, H.	Full Text	Duplicate data with Agrawal (2008), Parts A & B
Update on emissions and environmental impacts from the international fleet of ships: the contribution from major ship types and ports	2009	Dalsoren	Abstract	No original measurements of EF_{PM}
Characterisation of single particles from in-port ship emissions	2009	Healy	Full Text	Measurements are of totals at receptor, not apportioned to ship activity
Emissions of maritime transport: A European reference	2009	Schrooten	Abstract	No original measurements of EF_{PM} (review)
Final Report The Cleanest Ship Project	2009	Schweighofer & Blaauw	Full Text	No original measurements of EF_{PM}
Emissions of NOx, SO2, CO, and HCHO from commercial marine shipping during Texas Air Quality Study (TexAQS) 2006	2009	Williams	Abstract	No original measurements of EF_{PM}
Air quality impact assessment of at-berth ship emissions. Case-study for the project of a new freight port.	2010	Lonati	Abstract	No original measurements of EF_{PM}

The direct influence of ship traffic on atmospheric PM2.5, PM10 and PAH in Venice	2011	Contini	Abstract	No original measurements of EF_{PM}
Real-Time Gaseous, PM and Ultrafine Particle Emissions from a Modern Marine Engine Operating on Biodiesel	2011	Jayaram	Abstract	Measurements are specific to biofuels; beyond scope
Particulate emissions from large-scale medium-speed diesel engines: 2. Chemical composition	2011	Sarvi	Full Text	Measurements are of chemical composition, not particle mass
Black carbon from ships: a review of the effects of ship speed, fuel quality and exhaust gas scrubbing	2012	Lack & Corbett	Title	No original measurements of EF_{PM} (review, and measurements are of black carbon, not total PM)
Microscopic characterization of individual particles from multicomponent ship exhaust	2012	Popovicheva	Title	Measurements are of particle chemical composition and morphology rather than mass concentration
Estimation of exhaust emission from ocean-going vessels in Hong Kong	2012	Yau	Abstract	No original measurements of EF_{PM}
Measurements of air pollution emission factors for marine transportation in SECA	2013	Alfoldy	Full Text	Emission factors are reported for total particle number and particulate sulfur mass, not total PM mass.
Particulate Matter Emissions Reductions due to Adoption of Clean Diesel Technology at a Major Shipping Port	2013	Kuwayama	Abstract	No original measurements of EF_{PM} (source apportionment study)
Effects of Switching to Lower Sulfur Marine Fuel Oil on Air Quality in the San Francisco Bay Area	2013	Tao	Abstract	No original measurements of EF_{PM}
Black carbon emissions from in-use ships: a California regional assessment	2014	Buffaloe	Title	BC not PM
Contribution of harbour activities and ship traffic to PM2.5, particle number concentrations and PAHs in a port city of the Mediterranean Sea (Italy)	2014	Donateo	Abstract	No original measurements of EF_{PM}

Impact of inland shipping emissions on elemental carbon concentrations near waterways in The Netherlands	2014	Keuken	Full Text	Emission factors are reported for elemental carbon mass and particle number, but not total particle mass, and on a per time or per velocity basis.
Mobile measurements of ship emissions in two harbour areas in Finland	2014	Kivekas	Full Text	No original measurements of EF_{PM}
Identification and quantification of shipping emissions in Bohai Rim, China	2014	Zhang	Abstract	No original measurements of EF_{PM} (source apportionment study)
An AIS-based approach to calculate atmospheric emissions from the UK fishing fleet	2015	Coello	Abstract	No original measurements of EF_{PM}
Particulate matter in marine diesel engines exhausts: Emissions and control strategies	2015	Di Natale	Abstract	No original measurements of EF_{PM} (review)
Multi-catalytic soot filtration in automotive and marine applications	2015	Johansen	Abstract	No original measurements of EF_{PM}
Monitoring compliance with sulfur content regulations of shipping fuel by in situ measurements of ship emissions	2015	Kattner	Abstract	No original measurements of EF_{PM}
Evaluating the social cost of cruise ships air emissions in major ports of Greece	2015	Maragkogianni	Abstract	No original measurements of EF_{PM}
Characterization of black carbon-containing particles from soot particle aerosol mass spectrometer measurements on the R/V Atlantis during CalNex 2010	2015	Massoli	Abstract	No original measurements of EF_{PM}
Characterisation of ship diesel primary particulate matter at the molecular level by means of ultra-high-resolution mass spectrometry coupled to laser desorption ionisation-comparison of feed fuel, filter extracts and direct particle measurements	2015	Ruger	Abstract	No original measurements of EF_{PM}

Characterisation of ship diesel primary particulate matter at the molecular level by means of ultra-high-resolution mass spectrometry coupled to laser desorption ionisation-comparison of feed fuel, filter extracts and direct particle measurements	2015	Sorooshian	Full Text	No original measurements of EF_{PM}
Comparison of fuel consumption and emission characteristics of various marine heavy fuel additives	2016	Jang	Full Text	Emissions reported as mass concentrations rather than emission factors (no fuel or power consumption data).
Influence of in-port ships emissions to gaseous atmospheric pollutants and to particulate matter of different sizes in a Mediterranean harbour in Italy	2016	Merico	Full Text	No original measurements of EF_{PM}
Impact of harbour emissions on ambient PM10 and PM2.5 in Barcelona (Spain): Evidences of secondary aerosol formation within the urban area	2016	Perez	Abstract	No original measurements of EF_{PM} (source apportionment study)
Performance and emission characteristics of additives-enhanced heavy fuel oil in large two-stroke marine diesel engine	2016	Ryu	Full Text	Emissions reported as mass concentrations rather than emission factors (no fuel or power consumption data).
Uneven distribution of inorganic pollutants in marine air originating from ocean-going ships	2017	Bencs	Full Text	Measurements not linked to specific ships
Experimental Study on the Effects of Ultra-low Sulfur Diesel Fuel to the Exhaust Emissions of a Ferry	2017	Durmaz	Full Text	Duplicate data with Kalender & Ergin (2012)
The effects of marine vessel fuel sulfur regulations on ambient PM2.5 at coastal and near coastal monitoring sites in the US	2017	Kotchenruther	Abstract	No original measurements of EF_{PM}

Hydroxyl radical formation and soluble trace metal content in particulate matter from renewable diesel and ultra low sulfur diesel in at-sea operations of a research vessel	2017	Kuang	Full Text	Companion paper Betha (2017) has PM emission data
Influence of Ship Emissions on Urban Air Quality: A Comprehensive Study Using Highly Time-Resolved Online Measurements and Numerical Simulation in Shanghai	2017	Liu	Abstract	No original measurements of EF_{PM}
Shipping emissions in a Nordic port: Assessment of mitigation strategies	2017	Lopez-Aparicio	Abstract	No original measurements of EF_{PM}
Atmospheric impact of ship traffic in four Adriatic-Ionian port-cities: Comparison and harmonization of different approaches	2017	Merico	Abstract	No original measurements of EF_{PM}
Atmospheric impact of ship traffic in four Adriatic-Ionian port-cities: Comparison and harmonization of different approaches	2017	Steffens	Abstract	No original measurements of EF_{PM}
Chemical characterization and toxicity assessment of fine particulate matters emitted from the combustion of petrol and diesel fuels	2017	Wu	Abstract	No original measurements of EF_{PM}
Characteristics of marine shipping emissions at berth: profiles for PM and VOCs	2018	Xiao	Full Text	Grab sampling of particles and mass spectrometry not conducive to emission factor calculation

3.6 References

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