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Geospatial frameworks for differential health impacts of heterogeneous particulate matter in arid and semi-arid regions

A Dissertation submitted in partial satisfaction of the requirements
for the degree Doctor of Philosophy

in

Earth Sciences

by

Emmet Daler Norris

Committee in charge:

Professor Sarah Aarons, Co-Chair
Professor Tarik Benmarhnia, Co-Chair
Professor Manuel Shvartzberg Carrió
Professor BT Werner

2025

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The Dissertation of Emmet Daler Norris is approved, and it is acceptable in quality and form for publication on microfilm and electronically.

University of California San Diego

2025

DEDICATION

To childhoods.

All beings seek to create a safe space for the next generation. May our nests be protected, and may we tend to intrinsic wonder and wellbeing.

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LIST OF ABBREVIATIONS

SJP	San Jacinto Peak
PM	Particulate matter
$PM_{2.5}$	Particulate matter of less than 2.5 micrometers in size
PM_{10}	Particulate matter of less than 10 micrometers in size
PMF	Positive Matrix Factorization
ICD	International Classification of Diseases
RR	Relative Risk
CI	Confidence Interval
EJ	Environmental Justice
CEJ	Critical Environmental Justice

ACKNOWLEDGEMENTS

In acknowledgment of how little I have been aware of all the grace and love, patience and kindness that has been shared with me, I would like to express my gratitude to several specific forces that brought this dissertation and my current life into being.

To begin, I thank Sarah Aarons for opening the door to Scripps Institution of Oceanography and teaching me what it meant to be there. With attention to my process, methods, and how I ask questions, I am grateful for her patience with my learning, and for giving me the space to explore.

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To the boundless graciousness of bt werner, creating expansive spaces to think, learn and feel belonging, and for sharing their attention and time at critical moments. To Manuel Shvartzberg Carrió; I sincerely feel our interactions have altered the course of my life for the better. I admire his process, grace, and the way he greets people.

To our home Earth and our Sun for being so fantastically coordinated, and making a wonderful place to live. To the trees that have shaded and nourished me throughout my life, and especially during the last 5 years while working on this dissertation. I am grateful to all who have come before us.

To my sister for being the light that I follow. To my mother for giving birth to me and being such a uniquely enjoyable person to live life alongside. I could not have completed this dissertation without her attention and love at the final stages. To my father for exposing me to the largest questions, and for asking them together. To Nadina for dancing with me in all worlds. To Anza, my miraculous daughter and teacher; it would make my wildest dreams come true if you read this someday and say, "*I understand being a child of Earth so much better than you.*" I dedicate my life in service to your world. To my teacher of this life. To Rej for joyful rejuvenation, and to all in my community who raised me. There are so many magnificent people that I have been blessed to exist alongside.

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Chapter 3, in part, is currently being prepared for submission for publication of the material. Norris, Emmet; Ma, Yiqun; Benmarhnia, Tarik. The dissertation author was the primary researcher and author of this material.

Chapter 4 contains unpublished material coauthored with Norris, Emmet; Werner, BT, and Singh, Shruti. The dissertation author was the primary author of this chapter.

PROLOUGE

I perceive the atmospheric transport of earthly particles to be one of the more exquisite forces that our planet has perfected over the last 4.5 billion years. As one of many vantage points from which to view Earth and human societies, I find the language of dust well suited to articulate the astonishing inter-relationships between geographically disparate parts of the Earth, and how industrial societies have transformed them. As dust is a communicator between events that happen on the Earth's surface and what occurs in the sky, it is quite adept at characterizing how stable, or unstable, surfaces are, and what that means for other parts of the world. For a mountain stream, an upwind copper mine means something entirely different than the grassland it used to be. The life of a coal miner in Mongolia, might seem removed from a farmer in California, but they are connected through life on the Earth's surface, and we can learn about how they interact through dust.

In 2024, parts of India and Pakistan had declared emergencies, closing schools and issuing stay-home advisories due to severe air pollution, while wildfires in California burned more than 1 million acres (CAL FIRE, 2024), followed by devastating wildfires in the Los Angeles region in January 2025, exacerbated by drought and up to 100 mile per hour winds, killing at least 27 people, burning more than 400,000 acres and more than 12,000 structures (CAL FIRE, 2025). 2024 was the hottest year ever recorded (with average surface temperatures 2.30 ˚F or 1.28 ˚C above the 20th-century baseline; NASA, 2025), and had the highest number of global conflicts since World War II (Institute for Economics & Peace, 2024), including the immeasurable human suffering from genocide in Gaza, and violent conflicts or wars in Sudan, Syria, Myanmar, Democratic Republic of the Congo, Haiti, Columbia, and Ukraine. Globally, 90 million displaced people are living in countries with high-to-extreme exposure to climate-related hazards (UNHCR, 2024). Each event is connected by the atmospheric burden of violence and modification of global climate, either creating or receiving the impact of unhealthy concentrations and compositions of airborne particles and greenhouse gases. Dust doesn't abide by borders, and it carries the weight of actions to consequence others.

This text tracks a chronological body of work, and the arc of my learning as a person and scientist while a PhD student at Scripps Institution of Oceanography. It is heavily informed by events that took place outside of the laboratory or academia, including living and working with communities on the US-Mexico borderlands; accompanying refugees from climate change and violence on their journeys for self-determination amidst hostile conditions; and becoming a parent. Entering Scripps, I found myself questioning how to be a geologist engaged in and supporting ongoing movements for justice, especially related to the disproportionate exposure of some communities to toxic particulate matter and its impact on the more-than-human world. This led to the core intent of the following dissertation: exploring geological and epidemiological disciplines and critiquing their functions in perpetuating colonial aims and knowledge, or conversely, confronting them. The study of dust has taught me sobering facts about the impact of my life and society on Earth. I believe the interwoven stories that are held in microscopic particles transiting the sky have a lot to teach us.

VITA

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PUBLICATIONS

1. **Norris, E.**, Ma, Y., Benhmarhia, T. *Spatial and Temporal Variance in Source Apportioned PM2.5 Health Risks Across California: Population and Climate Effect Modifiers* (in preparation)
2. **Norris, E.**, Bailey, J., Ma, Y., Benhmarhia, T., Aarons, S., Goswami, A. *Source Apportionment and Differential Health Impacts of Short-term Particulate Matter Exposure in Arid and Semi-Arid Regions: Scoping Review*, GeoHealth (in preparation)
3. **Norris, E.**, Aarons, S., Hu, K., Ferrier, K., Blankenship, R., Goswami. A., Han, J. *Disentangling Natural and Anthropogenic Sources of Dust to a Montane Ecosystem at San Jacinto Peak*, *Southern California*, JGR: Atmospheres (in review)
1. Huang, L., Aarons, S., Koffman, B., Cheng, W., Hanschka, L., Munk. L., Jenckes. J., **Norris, E.**, Arendt. C. *Role of Source, Mineralogy, and Organic Complexation on Lability and Fe Isotopic Composition of Terrestrial Fe sources to the Gulf of Alaska*, ACS Earth and Space Chemistry, 2024
2. Light, T., **Norris, E.**, Zhai, D., Varner, R., Lanpher, K., Capone, D., Erazo, N., Norris, R. *All Aboard! Behind the Scenes of a Scientific Research Cruise*, Frontiers for Young Minds, 2024
3. Erazo, N., Light, T., Capone, D., Effinger, A., Erazo, P., Huang, L., Kannad, A., Lanpher, K., **Norris, E.**, Perry, S., Romero, E., Russell, T., Varner, R., Wicker, L., Aoming Yu, A., Zhai, D., Norris, R. *The Ocean as a Classroom: Considering the Roles of Equity, Diversity, and Justice in Oceanographic Knowledge Production to Promote Accessibility for Future Generations*, Oceanography 36(4), 2023
4. Munroe, J., **Norris E.**, Olson, P., Ryan, P, Tappa, M., Beard, B. *Quantifying the contribution of dust to alpine soils in the periglacial zone of the Uinta Mountains*, Utah, USA, Geoderma, Volume 378, 2020
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FIELD OF STUDY

Major Field: Geochemistry

Studies in mineral dust transport and composition
Professor Sarah Aarons

Major Field: Epidemiology

Studies in particulate matter exposure and related health impacts
Professor Tarik Benmarhnia

ABSTRACT OF THE DISSERTATION

Geospatial frameworks for differential health impacts of heterogeneous particulate matter in arid and semi-arid regions

by

Emmet Daler Norris

Doctor of Philosophy in Earth Sciences

University of California San Diego, 2025

Professor Sarah Aarons, Co-Chair
Professor Tarik Benmarhnia, Co-Chair

The atmospheric transport of mineral dust or particulate matter (PM) interacts with global Earth systems at multiple scales and is significantly modified by human activities. Different academic disciplines address unique facets of the environmental and public health significance of dust. However, given the complex composition and emission dynamics of dust sources, especially in arid and semi-arid regions, there is a need for further multiscale and trans-disciplinary research to develop new tools to consider their differential health impacts. In this dissertation, I use multiple geospatial frameworks to study the generation, transport and health impacts of dust,

focusing on case studies in California. In chapter 1, I investigate the spatial and temporal variance of mineral dust deposition through a multi-year field study on San Jacinto Peak in Southern California, using low-temperature geochemical tools including flux, major and trace element enrichment, and Positive Matrix Factorization (PMF). I highlight the significant presence of anthropogenically created particles, and the impact of wildfires on deposition rates and metal content. In chapter 2, I perform a scoping review and meta-analysis of relative risks from PM exposure to assess the extent of global knowledge of PM-related cardiorespiratory hospitalizations and mortality in arid and semi-arid environments, investigating how climate plays a role in modifying health outcomes from exposure to PM. In chapter 3, I combine geochemical source apportionment tools (PMF) and an epidemiological two-way fixed effects model to study the association between source-specific PM concentrations and cardiorespiratory hospitalization across California. This study spans a more than 10-year period, and identifies effect modification by climate, air basin, and season. In chapter 4, I summarize environmental justice and anti-colonial frameworks to investigate multi-timescale interactions between human land uses, dust emission, and intergenerational health impacts, with a case study of the Imperial Valley, California, to explore the potential use of this approach. This discussion examines a quantitative accounting method to trace dust impacts on health, connecting specific sources and short-term health impacts to long-time-scale contexts that create land use related dust sources and disproportionate burdens on specific populations.

INTRODUCTION

The transport of earthly particles in the atmosphere is part of a global cycle that shapes Earth's surface and distribution of life. These particles have many labels, including mineral dust, particulate matter (PM), aerosols, pollen, smoke, and air pollution. The range of terminology indicates the extreme heterogeneity of these particles, and their multi-faceted roles in complex ecological webs. In this dissertation, I primarily use the **term *dust*** to include *all solid particles that are emitted from the earth and transit the atmosphere, from a range of origins*.

At a global scale, dust plays a significant role in regulating climate and ecosystems. In the atmosphere, particles either scatter or inversely absorb solar radiation, thereby having either a cooling or heating effect, determined by particle mineralogy and size (Forster et al., 2007; Kok et al., 2023, and references therein). Dust also plays an important role in precipitation, acting as cloud condensation nuclei (Karydis et al., 2011). The impact of dust on ecology occurs primarily through supplying nutrients to a wide range of marine and terrestrial organisms. A climatologically relevant example is the contribution of biologically-limiting nutrients to phytoplankton, which are thought to consume 40% of annually fixed atmospheric CO_2, and are often limited by the abundance of dust-supplied Fe and other metals in the ocean (Falkowski, 1994; Schroth et al., 2009; Shoenfelt et al., 2017; Wang et al., 2017). The Amazon Rainforest in South America relies on a continuous supply of nutrient-rich dust from the Sahara Desert in northern Africa to support the world's largest concentration of life (Swap et al., 1992). Additionally, dust from both the Gobi Desert and local deserts is the most important source of nutrients to soil in many highland ecosystems in the western United States (Aciego et al., 2017; Hirmas & Graham, 2011; Munroe et al., 2024; Reynolds et al., 2006). These are examples of transcontinental relationships between dust sources and receiving ecosystems (Bullard, 2013; Goudie, 2014).

Inhalation of dust can be harmful to human health, especially if new dust sources activate on short timescales and the receiving system is unable to adjust to the influx of new chemicals. In 2022, environmental pollution was estimated to be the largest single risk factor for death that humanity faces, of which airborne particulate matter is the largest contributor (Fuller et al. 2022). The emergence of new dust sources—especially in the last 300 years, associated with human land use change which largely coincided with the industrial revolution—has considerably altered the quantity of globally transported dust, its composition, and its effect on climate and public health (Chen et al., 2018; Kim et al., 2015; Lambin & Geist, 2008; Shao et al., 2011). Notably, emerging dust sources from human activity have increased the total mass of transported dust (including that from the Sahara Desert) by 19%, which is especially significant in semi-arid environments where croplands and urban environments emit 43% of global anthropogenic emissions (Chen et al., 2018).

The many ways that human activities modify ecosystems on Earth's surface are commonly referred to as land-cover changes as a result of land use activities. The term "land-cover change" refers to biophysical alterations of the attributes of Earth's surface (e.g., removing trees in a forest and replacing them with soybeans, or flooding a valley upstream of a dam), while "land use" refers to the intent to alter the Earth's surface for a particular purpose (e.g., making a soybean farm in the Amazon, or damming the Colorado river to bring water to Las Vegas to produce hydroelectric energy) (Lambin et al., 2001). The complexity of land-cover changes and land use impact on dust emissions is significant, because of how the dust cycle interacts with other cycles such as carbon, nitrogen, energy, water, human respiration, and soil pedogenesis (Shao et al., 2011).

It is projected that dust, and specifically $PM_{2.5}$ (PM of less than 2.5 micrometers in size), emissions will increase in the future (Park et al., 2020). Dust production relies on the availability of unstable particles to be suspended in the atmosphere by the force of wind (Kok et al., 2012). The destabilization of land surfaces through a combination of human disturbance and short-timescale shifts in precipitation and temperatures—as is occurring due to climate change—are thought to increase global emissions, and increase the frequency of dust storms and subsequent impacts on human health (Lababpour, 2020; Li et al., 2021). Potential reductions in anthropogenic $PM_{2.5}$ emissions—such as from improvements in vehicle emission technology and emission regulations (Yan et al., 2011)—is thought to be entirely cancelled out by the increased likelihood of wildfire smoke from wildfires fueled by higher ambient temperatures (Burke et al., 2023; Goss et al., 2020). By 2100, it is projected that wildfire smoke will account for more than 50% of $PM_{2.5}$ in the United States by 2100 (Ford et al., 2018). Overall, in light of current and potential future dust and PM concentrations and global impacts, considerable global action—supported by increased understanding of the role of human choices—is needed to address multi-scale human and environmental health burdens.

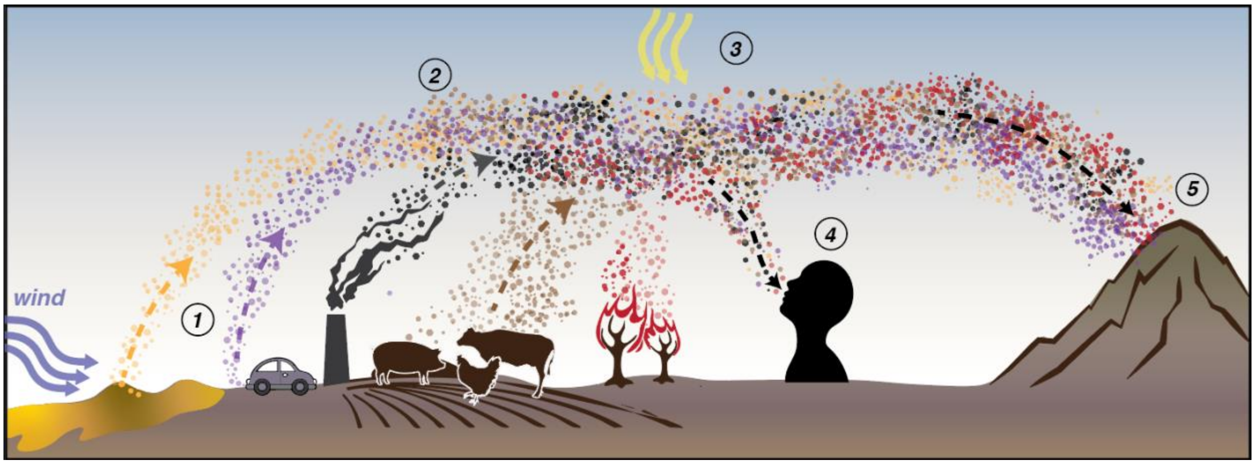

Figure 1.1. Components of the dust cycle discussed in this dissertation. **1)** Particles are emitted from sources including arid regions (e.g., deserts) (yellow), vehicle exhaust (purple), smokestacks and factory emissions (grey), agricultural activities (brown), and biomass burning (red). **2)** Particles are then transported in the atmosphere, and **3)** can undergo transformation due to sunlight or chemical reactions with other particles or gases. **4)** Humans are exposed to the mixture of particles in ambient air with a range of consequences. Finally, **5)** particles are deposited across the Earth's surface and enter ecosystems.

1.1 Multiple approaches to studying dust

Due to the complex nature and origins of modern social and environmental issues, trans-disciplinary research has proven to be effective in addressing environmental health (Sauvé et al., 2016). The study of dust and air pollution, which interacts with multiple fields of research due to its multi-scale impacts on Earth-systems and facets of human society, is often trans-disciplinary or interdisciplinary research. As different disciplines focus on specific aspects of dust throughout its cycle (Figure 1), they produce complementary types of information that can contribute to understanding the behavior of dust at multiple scales or create altogether new themes of research. In this dissertation, I draw from both geochemistry and epidemiology, using multiple geospatial techniques. Further, I discuss the benefit of including additional research frameworks in my final chapter (Chapter 4).

Epidemiological research has unequivocally shown that dust, referred to as particulate matter, negatively impacts human health (Du et al., 2016; Kim et al., 2015; Pope, 2000). Current estimates suggest that 4.7 million deaths globally are attributed to exposure of outdoor, fine particulate matter ($PM_{2.5}$; Health Effects Institute, 2024). Mortality however does not capture the full scope of the health burden on humans, which includes a wide range of diseases such as: asthma, chronic obstructive pulmonary disease, Alzheimer's disease, cancer, anxiety/depression, memory loss, infertility (Li et al., 2022; Sharma et al., 2023), and the increased frequency of transport-related accidents (Baryshnikova & Wesselbaum, 2023; Deng et al., 2024). As a core discipline of public health, epidemiology deals with the analytical or descriptive study of the distributions and determinants of health outcomes in populations (Szklo & Nieto, 2014), with an established history of identifying population level, statistically meaningful, impacts of PM on humans. Research using these tools cover a wide terrain, from infectious diseases (Munroe et al., 2019) to heatwave vulnerability (Benmarhnia et al., 2018), or wildfire smoke impacts on respiratory health (Marlier et al., 2023), using multiple types of study designs to identify causes or risk factors in a population or individuals in response to a health-related event or disease (Song & Chung, 2010). Specifically, the use of association measures such as risk ratios (RR) to assess the magnitude of adverse health outcomes between populations from differential long-term or short-term exposure, or differential susceptibility, is widely used to show the associations between PM and health outcomes, using hospitalization or mortality data (Burnett et al., 2014).

Study of dust by Earth science includes multiple tools at different stages in the dust cycle (Shao et al., 2011). These include: the emission of particles (Kok et al., 2012); transcontinental dust transport (Kellogg & Griffin, 2006); morphology and chemical composition evolution during transport (Krueger et al., 2004); interaction with global climate (Kok et al., 2023); the role of human activities in introducing new types of particles into ecosystems (e.g., microplastics, Brahney et al., 2020); and the role of dust as a fertilizer, providing biologically relevant nutrients (e.g., K, Ca, P, Mg, Fe, Cu) to diverse terrestrial and marine ecosystems (Arvin et al., 2017b; Crusius et al., 2011; Munroe et al., 2020; Reynolds et al., 2001). Understanding the role of dust deposition in terrestrial ecosystems is an ongoing field of research, primarily through the direct geochemical measurement of dust samples. Specifically, the use of isotopic variations in some elements (e.g., Sr, Nd, Hf, Pb, Fe, U) or the enrichment and depletions of rare earth elements (REEs) and metals enables researchers to identify the origins of dust—or dust provenance—from a range of geological materials and anthropogenic sources (Aciego et al., 2009; Gabrielli et al., 2010; Grousset & Biscaye, 2005; Reimann & de Caritat, 2005).

In this dissertation, I join a community of researchers who consider how to apply geochemical analysis of PM to the epidemiological study of population-level health risk from short-term exposure. Several studies have demonstrated that the chemical composition of some PM is more harmful than others (Stanek et al., 2011). In particular, organic compounds and metals are thought to be uniquely toxic, causing enhanced oxidative stress (i.e., an imbalance in the production of reactive oxygen species in the body's cells and tissues where the antioxidant system is not able to neutralize them) and inflammation, which is the origin of most cardiorespiratory diseases from PM exposure (Ghio et al., 2012; Hussain et al., 2016). PM from a range of crustal and fossil fuel combustion sources are composed of different materials, meaning the presence of these or other toxic compounds is highly variable. One approach to addressing the impact of differing PM compositions and toxicities on the public is to investigate their source. This can be achieved through remote sensing (Aguilera et al., 2023) or source apportionment modeling (such as Positive Matrix Factorization) of chemically speciated PM data (Kim & Hopke, 2007). Then, the relative concentrations of different sources in ambient air can be assessed in relation to spatiotemporal variations in health outcomes from exposure (Berger et al., 2018). The intent of this analysis, and the reason to employ geochemical techniques in epidemiological research overall, is to not only identify the magnitude of risk from PM exposure but identify what sources—and PM compositions—are most harmful to human health. This information in turn can then be used to enable communities and governments to target and reduce or eliminate the emission source.

It has been widely established that the spatial distribution of PM concentrations and toxicity, and health impacts from exposure, is highly unequal. Specifically in the United States, structural racism and classism has led to some communities—namely people of color and lower socioeconomic groups—to face higher air pollution burdens. For example, non-Hispanic Black people were 3.47 times more likely to die from $PM_{2.5}$ associated cardiovascular diseases than non-Hispanic white people (Ma et al., 2023), while Latino and black Californians are exposed to 39% and 43% higher $PM_{2.5}$ concentrations than white people (The Union of Concerned Scientists, 2019). This is partially due to the systematic and deliberate placement of toxic industries in communities of color (Tessum et al., 2021). These structural issues are classified as an environmental injustice, as the interaction between social injustices and environmental pollution are highly correlated and continue a legacy of slavery, racial marginalization, and the treatment of some people as dispensable (Pellow, 2017). These inequalities in particle exposure are only one aspect of the multi-scale harms created by technologies and land use arising in the Industrial Revolution, such as large scale mining (McNeill & Vrtis, 2017; Schneider et al., 2022) and monocrop agriculture (Holleman, 2017), which include intensive climate and Earth system function alteration, and the continuation of colonial projects and the extreme exploitation of the Earth's surface. Addressing structural and longer timescale dynamics in the creation of toxic dust and its impact on populations require a longer timescale view, as is addressed in environmental justice literature and resistance movements (Pellow, 2017). Further, considering that the causes of widescale land use changes and disruption of ecosystems that have intensively modified the dust cycle are largely a product of colonialism, study of historical and modern colonial land uses and impacts on the environment highlights the origins of many modern environmental pollution issues (Liboiron, 2021). In my fourth chapter, I discuss how research that considers multiple frameworks, including geochemistry, epidemiology, environmental justice, and anti-colonial research methods can provide a more comprehensive understanding of—and potentially more useful conclusions about—the differential impacts of heterogenous PM and dust.

1.3 Dissertation Overview

The following four chapters of this dissertation explore the nature of dust or particulate matter, each at a different scale and using different tools. The first project, **Chapter 1**, is a multi-year study of the spatiotemporal variation in dust deposition on a mountain in Southern California, using the enrichment of rare earth elements and metals to identify different dust sources. This project contributes to ongoing scholarship to understand the importance of dust in montane environments, and how human activities and climate modify the composition and amount of dust that enter those ecosystems. Field work, sample processing, and analysis spanned more than three years, and supported invaluable learning. In the second, **Chapter 2**, I explore unanswered questions about the impact of climate on cardiorespiratory health from PM exposure. I conducted a review of epidemiological studies in arid and semi-arid climate zones to assess the extent of knowledge about the risks faced in these regions, which are uniquely vulnerable to land-surface disturbance and dust emission. Following this work, **Chapter 3** combines tools from both geochemical and epidemiological disciplines to reveal the source-specific impact of short-term exposure to $PM_{2.5}$ on communities throughout California. I combined two modeling approaches to first create a >10-year dataset of source-specific PM concentrations at thirty-five $PM_{2.5}$ monitoring stations and then estimated the association between sources and cardiovascular hospitalization rates and modification by geographic, temporal and climate variables. In **Chapter 4,** I build off research questions from the previous three chapters and takes a wider perspective of the interactions between human health, dust emissions, and societal actions. I consider additional research frameworks to investigate the longer-time-scale and systemic changes to dust emissions that arise from global land use. Using complex systems science, Critical Environmental Justice, and anti-colonial methodologies, I explore how modern short-term harms from PM exposure from certain land uses are connected to long-time-scale systems of exploitation—arising from European colonialism—that sever existing relationships between geologic and ecological forces with a variety of impacts.

Each chapter is written in the format of a manuscript, as they were or will be submitted for publication. Some introductory sections may contain similar background information regarding the significance of particulate matter or dust, because I believe it is relevant to restate how overarching dynamics require specific attention to different scales of study.

Chapter 1 Disentangling Natural and Anthropogenic Sources of Dust to a Montane Ecosystem at San Jacinto Peak, Southern California

1 Introduction

1.1 Dust in the Earth system

The entrainment, transport, and deposition of fine-grained particles in the atmosphere, hereafter referred to as dust, can influence landscape ecology, climate, and human health (Field et al., 2010; Goudie, 2014; Grantz et al., 2003; Marx et al., 2018). The dust cycle is recognized for its role of providing biologically limiting nutrients to marine and terrestrial ecosystems (Belnap et al., 2011; Grantz et al., 2003; Johnson & Lindberg, 2013). The composition and supply of dust is particularly relevant in montane nutrient-limited systems where erosion outpaces the conversion of rock to soil (Aciego et al., 2017), and thus where aeolian mineral dust is a vital nutrient source. The source and formation process of dust controls its mineralogy, particle size, and geochemical composition, which in turn influences dust's impact at the deposition site. Therefore, the ability to track and identify the provenance of dust supplied to ecosystems with respect to time and space is important for gauging its long-term impact on the Earth's surface.

Arid regions—characterized by low or highly seasonal rainfall—are vulnerable to changes in soil moisture and land use changes, and therefore are highly susceptible to wind erosion and are common areas for dust emission (Nafarzadegan et al., 2021). The most common dust sources are arid deserts, floodplains, and proglacial valleys that are devoid of vegetation and especially susceptible to wind erosion of surficial sediment (Shao et al., 2011). However, urban and agricultural regions heavily impacted by soil disturbance and rapid land cover change have become significant components of the global dust cycle on shorter time scales (Webb & Pierre, 2018). Wildfires and agricultural activity can increase dust emissions by orders of magnitude (Duniway et al., 2019). Anthropogenically emitted dust—sourced from activities such as mining, agriculture, and urban development (Belnap et al., 2011; Duniway et al., 2019; Philip et al., 2017)—is increasingly recognized as a major global source of dust, especially in semi-arid and sub-humid regions not characterized as significant dust sources in the past (Chen et al., 2018). Increased dust emissions from both natural and anthropogenic sources may pose serious threats to public health (Achakulwisut et al., 2018; Goudie, 2014; Morman & Plumlee, 2013), particularly for vulnerable populations such as children (Johnston et al., 2019). Despite differences in the formation mechanisms of natural and anthropogenic dust in complex land-use regions like southern California (Fig. 1), it remains challenging to disentangle dust emissions driven by anthropogenic land-use from natural dust emissions driven by modern climate variability. In areas highly impacted by anthropogenic land use, dust geochemistry can reveal the impact of human activity on regional dust cycles (Frie et al., 2019).

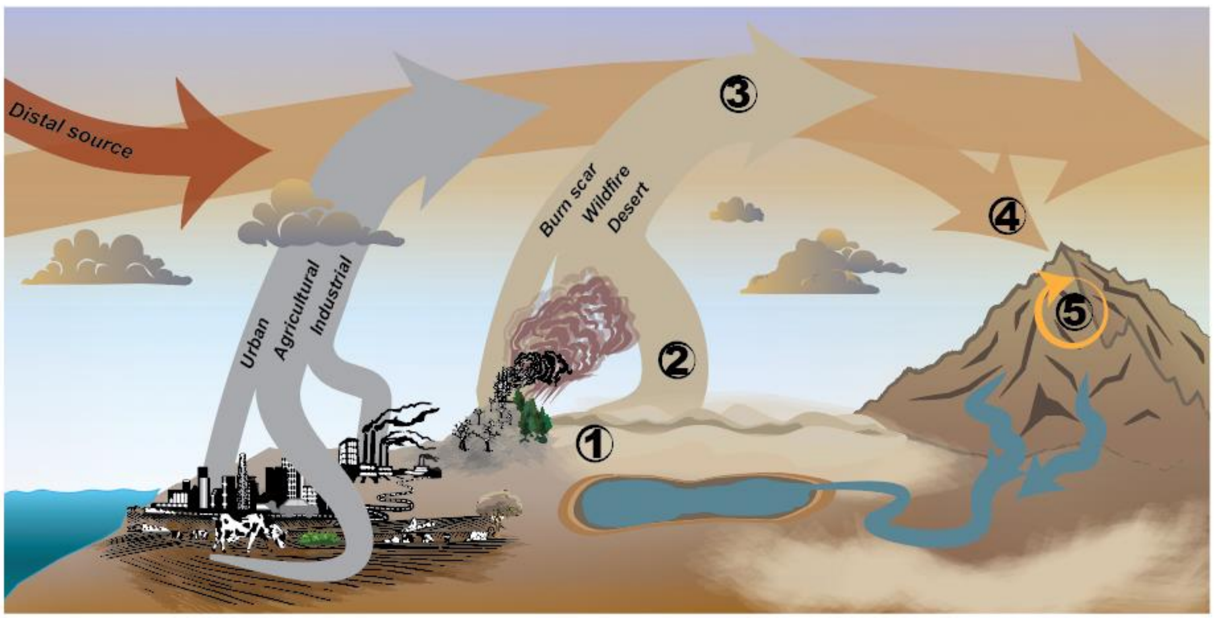

Figure 1.1. A schematic representation of the dust cycle in the context of our study, showing dust sources, mixing, and deposition on San Jacinto Peak. 1) Dust source: dust particles may originate as soil derived from the exhumation and weathering of rocks, or secondary products of manufacturing or waste in urban environments. 2) Emission: occurs through a variety of processes including a decrease of soil moisture or land surface disturbances, which interact with wind to suspend and entrain dust in an air mass. 3) Transport: once entrained, dust can be transported locally or globally, including across oceans. 4) Deposition: changes in topography, precipitation events or loss of energy in the air parcels cause dust deposition. 5) Stabilization: deposited dust is incorporated into terrestrial ecosystems.

1.2 Probing the influences of climate and anthropogenic forcing on dust flux

On long timescales, physical and chemical compositions of particulate matter in western North America have been linked to large scale climate variations such as glacial-interglacial cycles and sea surface temperature (Arcusa et al., 2020; Glover et al., 2020; Kirby et al., 2019; Kirby et al., 2015; Tau et al., 2021). Previous studies in the Southwest and Rocky Mountain West regions have documented the ecological significance of modern dust deposition to montane systems (Aciego et al., 2017; Arvin et al., 2017; Heindel et al., 2020; Lawrence et al., 2010; Munroe et al., 2020; Neff et al., 2008; Reynolds et al., 2001). Probing the role of aridity or land use change upon the regional dust cycle and whether dust composition of flux varies spatially across a mountain range is less well explored. This study reports the results of a multi-year seasonal record of dust deposition on two altitudinal transects at San Jacinto Peak in Southern California, surrounded by dense urban areas, agricultural producing valleys, and Southwest United States (US) deserts (Fig. 2). Our approach uses direct measurement of dust flux, dust geochemistry, and climatic factors including soil moisture and temperature from a more than three-year field campaign. This research aims to address how dust flux varies seasonally, how dust sources vary by location and elevation, and the roles of modern climate variability and anthropogenic land use on the composition, emission, and transport of dust in this region.

1.3 Site Background

San Jacinto Peak (hereafter referred to as SJP) is the most prominent peak within the San Jacinto Mountains in Riverside County, California (Fig. 2), and is dominated by Cretaceous plutonic rocks with a restricted lithological composition of tonalite and granodiorite (Hill, 1988). Rapid uplift and erosion beginning in the late Pliocene or early Pleistocene associated with the active San Jacinto fault zone has created steep and rough fault-controlled ridges and deeply incised canyons (Dorsey & Roering, 2006), resulting in much steeper topography on the north side of SJP compared to the south side. Climate conditions vary significantly with elevation and across the north and south faces of SJP (see Figure S1 in Supplementary Information S1 for more detailed information).

Meteorological records and atmospheric back trajectory modeling using the National Oceanic and Atmospheric Administration (NOAA) Hybrid Single-Particle Lagrangian Integrated Trajectory (HYSPLIT) model for 2021 show that the dominant wind direction reaching the north side of SJP and the

Coachella Valley is predominantly from the north for three months from November to February and from the west for the remaining nine months (Text S1 in Supplementary Information S1). Southerly winds increase in July to November and November to March because of the Santa Ana winds (Global Modeling and Assimilation Office (GMAO), 2015; Raphael, 2003; Stein et al., 2015) which carry material from inland deserts because of high-pressure systems over the Great Basin and southeastern California (Muhs et al., 2007). The HYSPLIT modeling indicates some air parcels reaching SJP originate across the Pacific Ocean (Figure S2 in Supporting Information S1), suggesting that SJP may receive airborne particulate matter from distal sources, which is consistent with the presence of Asian dust observed in the Sierra Nevada in central California (Aciego et al., 2017). The dominant wind direction on the south side of SJP is distinct from the north side (Fig. 2), characterized by southwesterly winds and potentially greater interaction with coastal air parcels originating across the Pacific Ocean (Global Modeling and Assimilation Office (GMAO), 2015).

Considering the potential mixture of distal and local dust sources, San Jacinto Peak is an ideal natural laboratory to study the effects of regional land surface, temperature, and elevation on dust deposition in a coastal montane environment. San Jacinto Peak is adjacent to dust emitting regions to the north, east, and south (Fig. 2A). The Coachella Valley to the east and northeast is an arid rift valley, approximately 72 km long and 24 km wide, which is undergoing rapid urbanization with the net impact of increasing aeolian sand and dust activity and transport (Katra et al., 2009). The Imperial Valley and Salton Sea Basin to the southeast are deserts with expanding playas from lake level decreases linked to agricultural land use and irrigation over the last several decades (Jones & Fleck, 2020). The 75 km^2 of exposed playa in the Salton Sea Basin contributes up to 89 g m^{-2} yr^{-1} of sediment lofted into the air as measured by monitoring locations nearby (Frie et al., 2019). Local alluvial sediment and particulate matter from agricultural burning and urban activity are also atmospherically transported, resulting in deteriorated air quality in this region (Johnston et al., 2019). The Mojave and Sonoran deserts to the east are well established dust sources formed by alternating wet and dry playa conditions (Reynolds et al., 2007; Shoop et al., 2018), and long-term emissions from these sparsely vegetated drylands are strongly linked to soil moisture and plant cover (Reynolds et al., 2007). Climate models anticipate warmer and drier conditions throughout the southwestern US with increasing effects of climate change, which is predicted to increase fine dust emission rates (Achakulwisut et al., 2018). Predicting future shifts in the dust cycle in Southern California therefore requires understanding the roles of climate and anthropogenic activity on dust flux and composition.

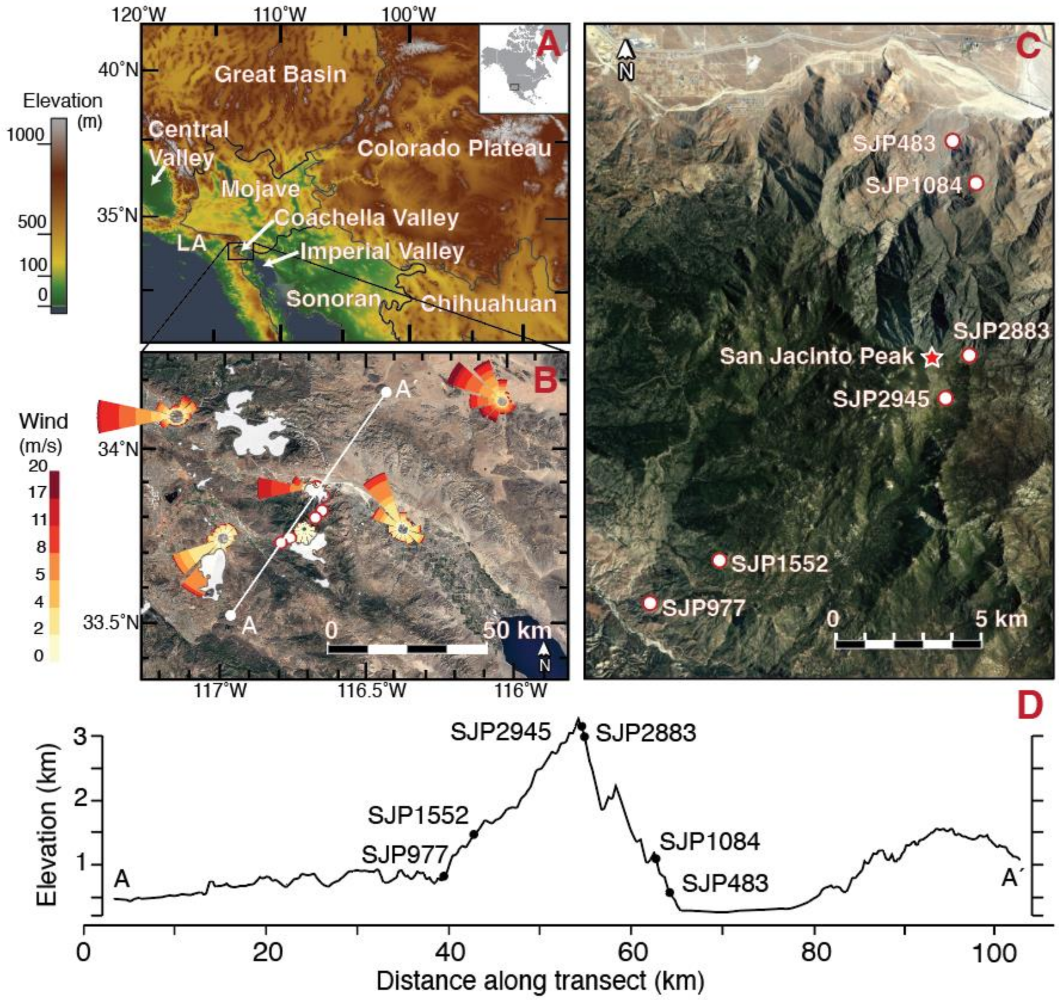

Figure 1.2. A) Topographic map of the southwest United States showing major southwestern North American deserts, the urban Los Angeles (LA) region and the predominantly agricultural Central Valley (Danielson & Gesch, 2011). B) Regional site map showing the location of study sites. Wind roses (MesoWest) represent hourly wind speed and direction from January 2019-March 2022 and dominant wind directions. Colors in wind roses represent wind speed, while petal orientation signifies the direction of wind, and its size represents the frequency of wind from that direction. The white line between A-A' is the location of the transect shown in panel D. White shaded areas are the extent of fires between 2018 and 2021 (Fire and Resource Assessment Program - FRAP). C) A map of the field area showing the location of the dust collectors and the summit of San Jacinto Peak (Image: USCSD-FPAC/GEO, 2009). D) SW-NE topographic profile across the A-A' profile shown in panel B of San Jacinto Peak highlighting the significant variation in elevation across our study sites.

2 Material and Methods

2.1 Sampling design and collection

We collected samples along two transects: The north face of SJP, which is a steep escarpment rising approximately 3000 m over 11 horizontal km from the Coachella Valley, and the south face, which is less steep and rises out of San Jacinto Valley. The transects cover a significant range in elevation from the base to peak: 2400 m on the north slope, and 1968 m on the south. In total the study spans 2462 m in elevation and more than 20 km in horizontal distance between the lowest elevation sites on either transect. The slope of the north transect in the Coachella Valley starts at <400 m in a desert environment of sparse creosote bushes and reaches to subalpine pine forest (>2600 m), spanning a mean annual air temperature difference of 18.7°C (Hanawalt & Whittaker, 1976; Western Regional Climate Center, 2023). The south transect is notably wetter, ranging from chaparral to subalpine pine forest, with a mean annual air temperature difference of 17.0°C (Western Regional Climate Center, 2023). Both transects receive air parcels from multiple directions (Fig. 2B and Fig. S2).

Between January 2019 and March 2022, we deployed passive dust collectors together with climate monitoring stations at six sites on the southwest and northeast transects of SJP to collect representative samples of dust at low, medium, and high elevations (Table 1). Most samples were collected every four months: March containing November-March deposition; July containing March-July deposition; and November containing July-November deposition. A four-month collection interval was selected to ensure the mass of collected dust in each sampler was sufficient for the intended chemical analysis. On three occasions the scheduled collection was not possible due to the COVID-19 pandemic or other extenuating circumstances, leading to several samples that do not follow the same temporal schedule (Table S1). In total thirty-eight (38) samples were collected and used for analysis. Sites were selected in exposed and remote areas to minimize the effects of local topography on dust deposition and to avoid human disturbance while simultaneously being accessible to researchers.

All sites use passive marble dust collectors (modeled after Reheis & Kihl, 1995). Collectors consist of a circular Teflon coated metal pan with a surface area of 0.0458 m^2 filled with quartz marbles on top of a PTFE mesh, which is mounted ~2 meters off the ground (Figure S3 in Supporting Information S1). Below the PTFE mesh, a spacer was added to create a chamber for the dust to settle and prevent resuspension. Zip tie bird spikes were mounted on the collectors to discourage roosting birds and possible sample contamination. Prior to deployment, all dust collector equipment was washed sequentially in 2M HCl, three times in Milli-Q water, 3M HNO_3, three times in Milli-Q water, 2M HCl, and three times with Milli-Q water in an ISO Class 7 clean lab at Scripps Institution of Oceanography (SIO) in La Jolla, CA. Dust sample collection occurred at four-month intervals, except for March 2020, due to a COVID-19 pandemic-related break in collection (Table 1). Three samples that spanned a yearlong period (483 01-20F, 2883 08-20F, 2945 07-20F) experienced abnormally long sample deployment due to COVID-19 restrictions and/or sample loss possibly due to rain/snow accumulation and are not compared with the typical four-month long sample period or used in analysis of dust flux or temporal variability in chemical composition. During collection, Milli-Q water stored in pre-cleaned 1L LDPE Nalgene bottles was poured over the pan to rinse the marbles. This mixture was then poured into a funnel over a cleaned 1L Nalgene with a 400 µm polyethylene filter to remove large non-dust items (e.g., sticks, pollen) collected in the pan. The pan and marbles were further rinsed three times and the dust-water mixture was stored in Nalgene bottles until brought to SIO and kept frozen until further processing. After sample collection a new set of cleaned marbles, a spacer, and mesh were deployed to begin the next collection period.

2.2 Sample processing

Samples frozen in water were melted and filtered within 24 hours in an ISO Class 5 exhausted laminar flow hood at SIO. The filter set up consists of an acid pre-cleaned 150 mL PFA Savillex column with PFA closures and two 47 mm filters, a first-stage 30 µm polycarbonate filter and secondary PTFE 0.2 µm filter to separate out the coarse (C: particles between 30-400 µm) and fine (F: particles between 0.2-30 µm) fractions, following methods described in Aarons et al. (2019). Filtered pressurized air was used to push the dust-water mixture through both filters. Once the water-sediment mixture passed through the column, the filters were dried in a laminar flow hood for a period of 24 hours before weighing and storage in separate pre-weighed Savillex PFA jars. Samples were weighed on a Mettler Toledo analytical balance with a precision of 0.01 mg and repeatability of 0.015 mg. The weight, sampling duration (percent of year), and surface area (size of pan opening in m^2) were used to calculate the dust flux (Table 1) which is measured as mass collected per unit area per unit time.

Collected particles consisted of a wide range of coarse material, including large (~400 μm) rock fragments and organic debris (e.g., pinecones). Large particles >30 μm are unlikely to have traveled far from their origin, as particles 70-100 μm in diameter tend to be transported through saltation and particles 20-70 μm tend to be transported via short-term suspension (Kok et al., 2012). Therefore, we consider particles >30 μm not to be atmospherically entrained dust at these sites. For the remainder of this study, we report measurements on the fine (0.2-30 μm) dust fraction.

2.3 Sample digestion

Aliquots for chemical analysis were dissolved with concentrated Optima grade acid using an Anton Paar Multiwave 5000 microwave. Each pre-weighed sample on a 0.2 μm PTFE filter was cut in half using a light board prior to digestion directly off the filter using 4 mL concentrated HNO_3, 1 mL concentrated HCl and 2 mL concentrated HF. Following initial sample digestion, the filter was removed, and the remaining solution was transferred into an acid-cleaned PTFE microwave vessel. The vessel was heated to 150˚C for a 20-minute interval, then to 185˚C for 10 minutes, and held at 185˚C for 3 hours. This sequence was repeated 8 times to ensure adequate dissolution of high field strength elements such as Zi. The sample was then transferred into a 15 mL PFA jar and dried then fluxed with 1 mL concentrated HNO_3. For trace and major elemental analysis, 100 µL, or 10%, was taken for analysis on a Thermo Scientific iCap Qc ICP MS at the UCI TEMPR lab. Each sample was loaded in 2% HNO_3, and five bracketing standards (0.1 to 1000 ppb) were used from the Inorganic Ventures IV-ICPMS-71A multi-element calibration standard. Samples were run in standard mode with acid blanks run every 8 samples. A total of 28 elements with sample concentrations at least 50 times greater than measured acid blanks were analyzed. United States Geological Survey geochemical reference materials (GSP-2 and BHVO-2) were also analyzed, with measured values in agreement with published values (Table S2 in Supporting Information S2).

2.4 Elemental enrichment relative to the continental crust

We use enrichment factors (Equation 1) to distinguish between natural and anthropogenic dust sources using our elemental concentration data.

$$EF_{X,i} = \frac{X_{sample}/i_{sample}}{X_{UCC}/i_{UCC}} \qquad \textbf{Eq. 1}$$

Here, X_{sample} and i_{sample} are the concentrations (ppm) of an element *X* and an index element *i* respectively, while X_{UCC} and i_{UCC} are the concentrations of *X* and *i* in the average upper continental crust (UCC), respectively (Wedepohl, 1995). We follow previous studies in normalizing concentrations of rare earth elements (REE) by the index element Ce, which has been shown to be useful in distinguishing dust sources (Gabrielli et al., 2010; Aarons et al., 2017). The rationale for using REE enrichment patterns (i.e., values of $EF_{X,\,Ce}$ calculated with Equation 1) to discern dust source hinges on the fact that the 14 REEs range in atomic number and radius but keep the same external electronic configuration. As such, REE chemical behavior and fractionation patterns are similar to those of isotopes and are therefore useful geochemical tracers (Gabrielli et al., 2010; Henderson, 1984).

For all other elements, we use a low-mobility element (Ti, which we assumed to be immobile) to calculate $EF_{X,\,Ti}$. Previous studies have identified $EF_{X,\,Ti}$ as an effective indicator of anthropogenic sources of heavy metals in water, dust, and soils (Goldberg, 1972; Reimann and de Caritat, 2005). Enrichment factors close to unity indicate little differentiation from the average upper continental crust, and thus are typically interpreted to be of crustal origin. By contrast, enrichment factors greater than 10 indicate a large deviation from the average upper continental crust, and thus are typically interpreted as anthropogenic in origin. The goal of using $EF_{X,\,Ti}$ in this study is to gauge the presence of anthropogenically sourced heavy metals in our dust samples. The UCC is not assumed to be representative of the local dust parent material composition but used as a reference point of geologic material for comparison with anthropogenic material.

2.5 Positive Matrix Factorization

Positive matrix factorization (PMF) is a mathematical receptor model, useful in identifying contributing sources to dust by reducing chemical data to a set of unique factors (Paatero and Tapper 1994). PMF works by analyzing a range of variables from measured sample data and outputting a set of factors and the degree to which each factor contributes to analyzed samples of unknown origin. The PMF model does not include potential source chemical signatures as an input, relying on the user to identify the material types or sources represented as unique factors, through comparison to previously measured source samples. Because our chemical concentration and $EF_{X,\ Ti}$ results (section 4.1.3) are not able to identify the degree to which regional sources contributed to our samples, we selected PMF as a tool to quantify source contributions, given its successful application in related studies (e.g., Frie et al., 2019).

In this study we used the EPA PMF 5.0 model to identify the likely sources of our dust samples using 14 measured chemical variables and 7 factors (Fig. 7). The number of factors was chosen using the sum of the squares of the scaled residuals (Q) divided by the expected Q (Qexp) based on the number of chemical variables. This value (Q/Qexp) is used to locate the number of factors which meaningfully explains the variance in our chemical dataset (see Text S3 in Supplementary Information S1 for more detailed information).

Using published dust source chemical compositions from the region (Watson & Chow 2001; Chow et al. 2003; Reheis et al. 2009; Oroumiyeh et al., 2022; Alshehri et al., 2023) and PMF analysis conducted for nearby dust samples (Frie et al., 2019), we attributed a single dust source type to each factor (Fig. 7A). Factor 1 is composed of all the elements included in the PMF analysis and shows strong similarity in elemental abundance to granitic rocks, including an outcrop (SJP2883-X2) measured in this study. We assign this the name *Bedrock*. Factor 2 is predominantly composed of Sr, Ca, Pb, and Na, which corresponds to the PMF source signature identified by Frie et al. (2019) as *Playa*, due to these elements' presence in evaporite minerals common in playas (Frie et al. 2017). Factor 3 includes a mixture of 12 out of the 14 included elements, resembling samples from the Mojave Desert (Aarons et al., 2017) and local alluvium (Frie et al. 2019). We assign this the name *Southwest Alluvium*. High K, Ca, and Cl in Factor 4 resembles the diagnostic characteristics of a sampled *Fly Ash* from a manure-fueled plant in Imperial County are (Watson & Chow 2001). Factor 5 is 55% Mn, 21% Co, 16% Cu, and 13% Ni, all of which are often associated with metallurgy or vehicular emissions (Farahani et al., 2021). To include several anthropogenic sources with fitting emission profiles, Factor 5 will be referred to as *Anthropogenic Metals*. Factor 6 is 41% Cu, in addition to Fe, V, and Sr. This factor is identified as *Vehicular Emissions* as brake wear and direct vehicle emissions are known sources of Cu and Sr—Fe to a lesser extent—while V is generally thought to come from diesel combustion (Schauer et al., 2006). Factor 7 is overwhelmingly composed of Ni. Cement, brick and glass manufacturing (Farahani et al., 2021) as well as power plant stack emissions (Watson & Chow 2001) are major sources of atmospheric Ni, so Factor 7 will be referred to as *Ni-Industrial* emissions. While these attributions are interpretations, they allow us to attribute dust from geologic or natural sources (Factors 1-3) from anthropogenic sources (Factors 4-7).

3 Results

3.1 Dust flux

Our measurements at the six study stations reveal disparate trends in fine dust flux (0.2-30 μm) with respect to elevation and sites (Fig. 3a, Table 1). Dust fluxes averaged 0.71 g m^{-2} y^{-1} across all stations and all seasons, with a maximum dust flux of 4.07 g m^{-2} y^{-1} at SJP2945 during March-July 2021 directly following a local wildfire, and a minimum of 0.08 g m^{-2} y^{-1} at SJP483 during January 2019-January 2020. The average total dust flux including both transects was higher during the July-November sampling interval (1.15 g m^{-2} y^{-1}) than the March-July (1.00 g m^{-2} y^{-1}) and November-March (0.63 g m^{-2} y^{-1}) sampling intervals. The largest dust fluxes occurred between July-November 2020 at all sites except SJP977. Notably, the July-November 2020 dust flux at SJP483 was more than two standard deviations greater than the elevation and seasonal average (Fig. 3b).

Table 1-1. The measured dust flux for each site (in g $m^{-2}y^{-1}$) by sampling period. Reported flux is for the fine (0.2-30 µm) fraction of dust only. Values in the "Average" column are averages over all sampling periods. Additional information about each collection is in Table S1 in Supporting Information S2.

Fine Dust Flux (g m^{-2} y^{-1})

Site ID	*Mean*	*Jan-Aug 2019*	*Aug 2019 - Jan 2020*	*Jan-Jul 2020*	*Jul-Nov 2020*	*Nov 2020 - Mar 2021*	*Mar-Jul 2021*	*Jul-Nov 2021*	*Nov 2021 - Mar 2022*
SJP483	1.097			0.359	3.496	0.477	0.348	1.763	0.565
SJP1084	0.486					0.444	0.238	0.989	0.287
SJP2883	0.514				0.792	0.581	0.630	0.299	0.317
SJP977	0.691		0.453	1.526	0.252	0.326	0.273	0.880	0.702
SJP1552	0.363	0.209	0.231	0.547	0.633	0.457	0.431	0.140	0.287
SJP2945	2.109				1.543	2.764	4.070	1.876	0.337

The overall trend in dust fluxes on the north transect were lower than those on the south transect, with the means and standard deviations of 0.7 ± 0.35 and 1.05 ± 0.93 g m^{-2} y^{-1} respectively. On the north transect, the site with the highest annual average dust flux was at the lowest elevation (1.1 g m^{-2} y^{-1} at 483 m), more than twice that at the other two northern stations (avg. 0.49 g m^{-2} y^{-1} at 1084 m and 0.51 g m^{-2} y^{-1} at 2883 m, respectively). On the south transect, the highest average dust flux was at the highest elevation (2.11 g m^{-2} y^{-1} at 2945 m), roughly 3-5 times higher than those at the other two southern stations (0.69 g m^{-2} y^{-1} at 977 m and 0.36 g m^{-2} y^{-1} at 1552 m, respectively). Dust fluxes at the low-elevation site SJP483 varied by a factor of 10 among seasons, ranging from 0.36 to 3.49 g m^{-2} y^{-1} (Fig. 3b) highlighting a wide seasonal variation in dust flux. For the stations on the north transect (SJP483, SJP1084, and SJP2883), dust fluxes tended to be highest during the July-November sampling interval.

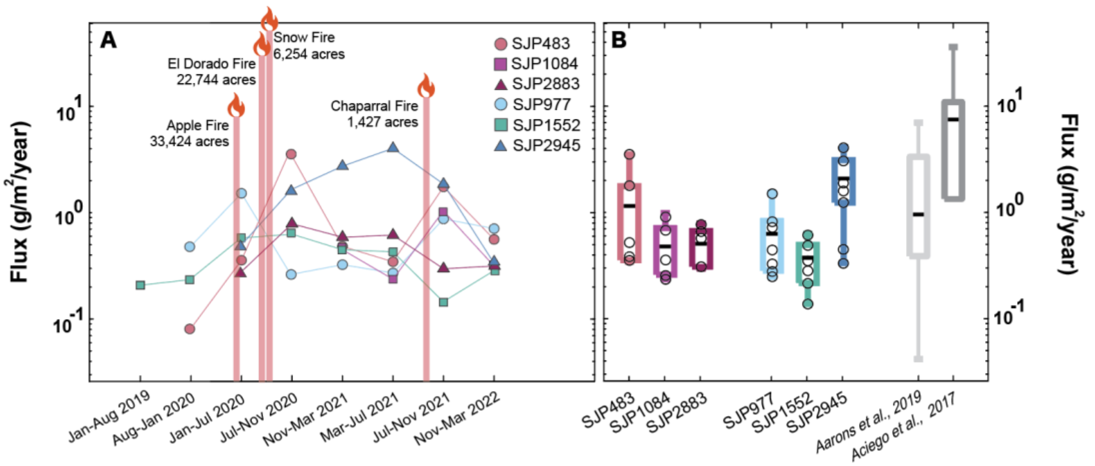

Figure 1.3. **A)** Measured dust fluxes over time at San Jacinto Peak. Vertical red lines represent wildfires that occurred in the region, along with the number of acres they burned. **B)** Box plot showing seasonal dust fluxes by site, with the north transect sites in ascending order on the left side, and the south transect sites on the right. Boxes contain the 50% of the sample values, with the top of the box representing the 75th percentile and the bottom representing the 25th percentile. The interior line represents the sample mean value. The whiskers show the range of remaining sample values. Dust deposition data from two previous studies of the Sierra Nevada Mountains, CA is plotted in gray box plots to show a comparison of our results with others in the region (Aciego et al., 2017; Aarons et al., 2019).

The standard deviations of the seasonal dust flux measurements indicate that dust fluxes varied over time more at some stations than others. The standard deviations of the dust fluxes were higher at the highest and lowest elevation sites (1.39 and 1.13 g m^{-2} y^{-1} at SJP2945 and SJP483, respectively) than at the other stations (0.51, 0.34, 0.18, and 0.21 g m^{-2} y^{-1} at SJP977, SJP1084, SJP1552, and SJP2883, respectively).

3.2 Trace elements

We measured a total of 28 trace elements (Table S2), and we follow previous methods which focus on elemental enrichments of six metals (Mn, Fe, Co, Ni, Cu, Pb) to differentiate between anthropogenic particulate matter and desert dust sources (Oroumiyeh et al., 2022). For these heavy metals, the average $EF_{X, Ti}$ (Eq. 1) tends to increase with elevation along both the north and south transects, except for Pb (Figure 4, Table S3). Most of these metals tend to be enriched in our dust samples relative to the UCC. The median enrichment factor was > 1 at all six sites for Ni, Cu, and Pb, at five sites for Mn, and at four sites for Co (Figure 4a). Only Fe was systematically depleted in our dust samples relative to the UCC, with a median $EF_{Fe, Ti} < 1$ at five of the six study sites (Figure 4a). All these metals were more enriched on the north transect than on the south transect.

Figure 4b indicates no systemic seasonal variation in metal enrichment factors, although there is significant variation in the mean for some elements (e.g., Ni) and total range of enrichment values (e.g., Fe, Ni, Pb). For all sites and intervals, the element with the highest enrichment factor was Cu, with an average $EF_{Cu, Ti}$ of 11.8 across all sites (14.6 on the south sites and 8.6 on the north sites). Site SJP2945 had the highest enrichment factors for all metals with average values of $EF_{Mn, Ti} = 4.3$, $EF_{Fe, Ti} = 1.7$, $EF_{Co, Ti} = 3.0$, $EF_{Ni, Ti} = 19.0$, $EF_{Cu, Ti} = 25.7$, and $EF_{Pb, Ti} = 6.0$. Sampling intervals during 2020 had the highest average $EF_{X, Ti}$ for the majority of metals, and with the March-July 2020 interval being the highest. These data show that metal $EF_{X, Ti}$ varies seasonally, the high elevation sites are more enriched, and Ni, Pb, and Cu are enriched relative to UCC for all samples, whereas Mn, Fe and Co are closer to the UCC composition.

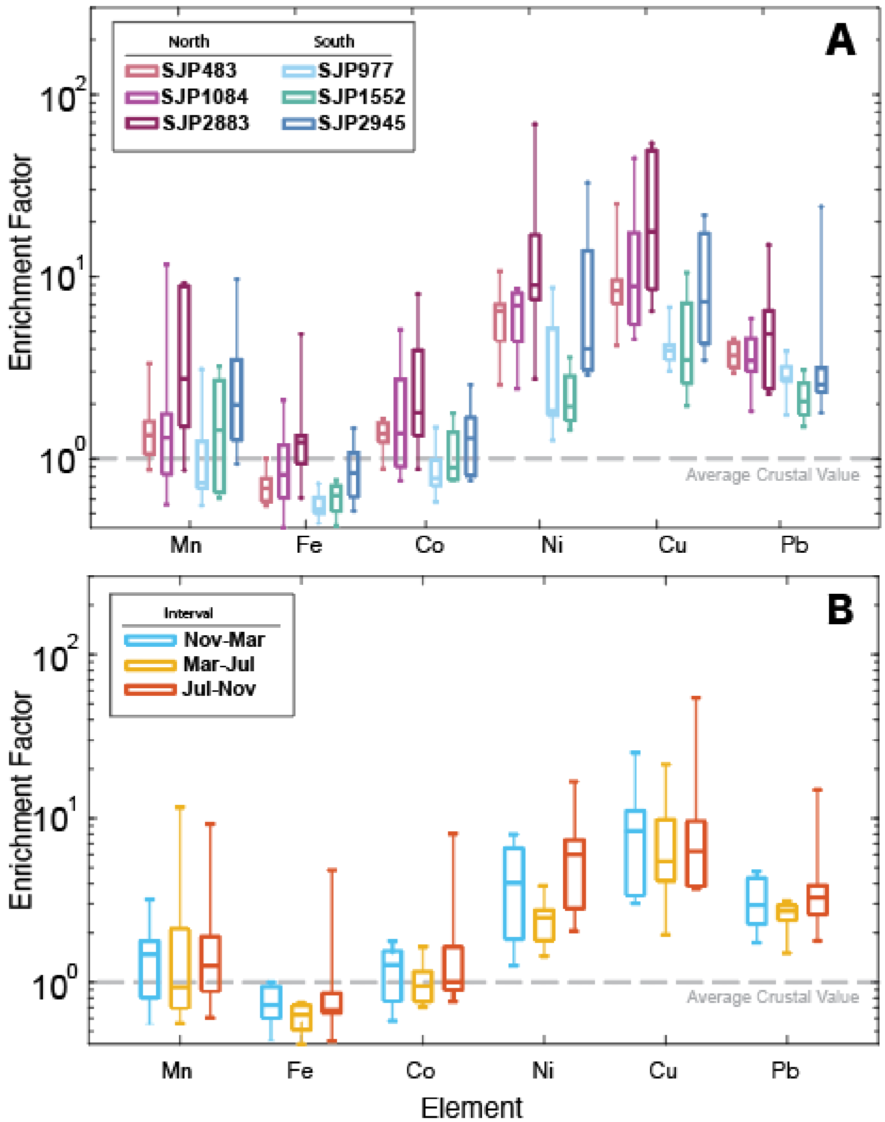

Figure 1.4. Enrichment factors $EF_{X, Ti}$ (Equation 1) for the six metals of interest (Section 3.2). **A)** Enrichment factors for each element at each study site. **(B)** Enrichment factors for each element by seasonal interval between 2019 and 2022, with intervals defined as: Nov-Mar, Mar-Jul, and Jul-Nov. The gray horizontal line at $EF_{X, Ti} = 1$ represents no enrichment or depletion relative to the average upper continental crust. The bottom and top of each box represent the first and third quartiles, while the horizontal line inside each box represents the median and the whiskers show the range.

3.3 REE variability

REE concentrations normalized to UCC ($EF_{X, Ce}$) are used to determine dust composition variability and gauge the contribution of multiple dust sources for each site (Fig. 5, Table S4). Each site is characterized by unique seasonal trends and distinct chemical compositions between seasons. Rare earth element patterns are grouped into light REEs (LREE: La, Pr, Nd), medium REEs (MREE: Sm, Eu, Gd, Dy, Ho) and heavy REEs (HREE: Tm, Yb). Average $EF_{X, Ce}$ values for LREE, MREE and HREEs were similarly enriched during the March-July intervals, and notably lower in the November-March interval. Enrichment factors were highest at site SJP977, and lowest at site SJP1084 for all REEs except La (Table S4).

REE enrichment factors for site SJP2883 reveal significant REE depletion in dust compared to the underlying bedrock samples (gray line in Fig. 5C) for all elements except La (Fig. 5). SJP2883 and SJP2945 samples are characterized by consistently low and significantly depleted REE enrichment factors in comparison to the SJP28833 bedrock outcrop composition, with all samples more than 3 standard deviations from the outcrop REE enrichment factor. Dust sample REE concentrations at SJP483, SJP1084, and SJP1552 are similar to the SJP2883 outcrop. At these sites, March-July and July-November values of $EF_{X, Ce}$ are greater than those in November-March samples, with a greater increase in MREEs relative to LREEs and HREEs. At SJP483 March-July and July-November samples for all years are enriched in all REEs relative to November-March samples.

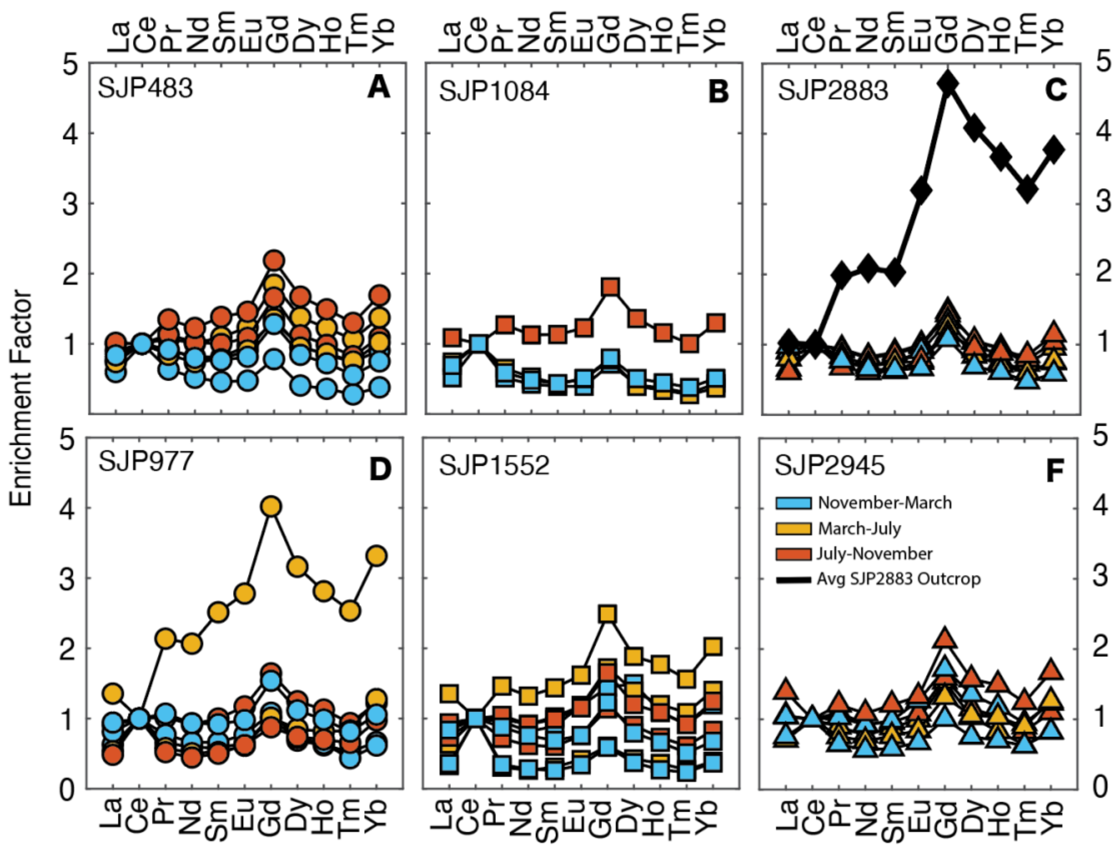

Figure 1.5. REE enrichment factors $EF_{X, Ce}$ (Equation 1) at dust collector site **A)** SJP483, **B)** SJP1084, **C)** SJP2883, **D)** SJP977, **E)** SJP1552, and **F)** SJP2945. Icon colors represent samples from different intervals. For comparison, panel C also shows REE enrichment factors for rock outcrop samples at SJP2883.

4 Discussion

4.1 Patterns in dust flux and composition

Fine dust deposition rates at SJP1084, SJP1552, and SJP2883 are similar, with relatively low means and ranges (0.486 (0.238-0.989), 0.363 (0.140-0.633), and 0.416 (0.269-0.792) g m^{-2} y^{-1} respectively). In comparison, two sites (SJP2945 and SJP483) have higher means and ranges (1.526 (0.337-4.070) and 0.778 (0.078-3.496) g m^{-2} y^{-1} respectively). Two sites on separate transects (SJP1552 and SJP2883) are characterized by similar seasonal patterns in dust deposition rate, whereas sites SJP483, SJP977 and SJP2945 vary significantly from each other. Sites located in close proximity do not have similar seasonal patterns in dust deposition rates, demonstrating that dust deposition—and potentially dust sources—are influenced by multiple factors such as proximity to dust sources and elevation above the valley floor (Fig. 3). Previous research on dust deposition in a rocky desert found the highest dust accumulation zones were at the base of concave windward slopes, followed by slopes parallel to the dominant wind direction (Goossens, 2000). Site SJP483 on the north transect is located at the base of a slope parallel to the dominant wind direction and has the highest average flux compared to higher sites located on more convex parts of the slope (Fig. 3), a pattern of dust deposition with respect to slope direction in agreement with previous observations (Goossens, 2000). However, the north transect sites—parallel to the dominant wind direction—have lower accumulation rates compared to the south transect which is at odds with observations in Goossens (2000). The highest elevation site on the south transect (SJP2945) is located on a convex slope and has the highest average dust flux which is also at odds with previous observations (Goossens, 2000). Our results indicate that changes in local topography are not the likely driver of dust deposition rates observed at SJP.

Previous dust measurements in the Coachella Valley and Salton Sea Basin indicate that dust fluxes are seasonally variable and are typically highest in the late spring and summer and lowest in the winter (Frie et al., 2019). Dust emissions from playas in the California Mojave Desert also vary seasonally, primarily driven by high velocity winds during winter storms, rainfall, and ephemeral streams, which recharge dust supply and destabilize surface crusts leading to differential temporal emission patterns (Reynolds et al., 2007). Visibility data from regional meteorological stations indicate that dust emissions are greatest during late winter and spring, and that interannual cycles such as El Niño-Southern Oscillation increase global dust production from sources including Central Asia and California playas (Reynolds et al., 2007). Large spikes in dust flux are not observed during the winter in this study, suggesting no significant wintertime increase in dust supply. Our observation of seasonally variable dust fluxes on the north SJP transect are consistent with previous studies (Frie et al., 2019; Hand et al., 2017), highest in the March-July and July-November collection periods and lowest in the November-March collection period.

Modeled estimates of dust flux in southern California range from 2 to 20 g m^{-2} y^{-1} at present (Mahowald et al., 2006; Kok et al., 2021), and a lesser 0.79-1.99 g m^{-2} y^{-1} averaged over the Holocene (Lambert et al., 2015). These Holocene dust flux estimates fall within the lower range of our measured dust fluxes (0.08 to 4.07 g m^{-2} y^{-1}, Fig. 3), indicating agreement between direct observations and model results. Our measured dust flux is within the range of previous dust flux measurement in the Sierra Nevada Mountains (Fig. 3, Aciego et al., 2017; Aarons et al., 2019). Previous studies focused on dust collection spanning 30 to 60-day intervals (Aciego et al., 2017) whereas others spanned a total sampling time period of 84 to 105 days (Aarons et al., 2019), both of which are significantly shorter than the roughly 2.5-year sampling period of this study. The ranges in dust flux observed in previous studies (0.04 to 6.98 g m^{-2} y^{-1}, Aarons et al., 2019; and 2.47 to 36.00 g m^{-2} y^{-1}, Aciego et al., 2017) are significantly greater than this study (0.08 to 4.07 g m^{-2} y^{-1}), even though the range in site elevation is comparable (2300 m for Sierra Nevada studies, and 2462 m for this study). One possible explanation for the higher fluxes in the Sierra Nevada is the proximity to the Central Valley, which is a hotspot of dust emission due to substantial agricultural activity (Ying & Kleeman 2006). Other potential explanations include strong interannual variability, particularly since the sampling in Aciego et al. (2017) and Aarons et al. (2019) occurred during and directly after a prolonged multiyear regional drought. Regional events such as drought (Aarons et al., 2019; Achakulwisut et al., 2018) have a strong impact on measured dust fluxes and thus need to be considered in extrapolating dust flux data retrieved during short-term dust collection campaigns to longer time periods.

In the case of SJP483, the large temporal variability in flux is largely driven by spikes during July-November 2020 and July-November 2021, which occurred after local wildfires (Fig. 3b). At site SJP2945, dust fluxes in samples collected from July-December 2020 through July-November 2021 are significantly elevated compared to dust fluxes just before this period (August-July 2020) and just after (November-March 2022), resulting in a high standard deviation for all intervals at SJP2945. If the post-wildfire samples during the July-December 2020 and July-November 2021 sampling interval are excluded, the average dust fluxes at SJP483 and SJP2945 are similar to those at the other four stations.

Dust fluxes at the mid-elevation sites SJP1552 and SJP1084 were less variable throughout the sampling campaign, with all measured dust fluxes falling within one standard deviation except for July-November 2021 at both sites (coefficient of variation 0.482 and 0.705 respectively). This collection period occurred immediately after multiple local wildfires (Fig. 3b) further discussed in section 4.1.2. While SJP1552 records a slight increase in flux following 2020 fires, this site does not record the same pattern as others after 2021 fires. The standard deviation of dust flux across all sites and instances is highest in the March-July samples (1.51 g m^{-2} y^{-1}) followed by July-November (1.08 g m^{-2} y^{-1}) and then November-March (0.14 g m^{-2} y^{-1}).

4.1.1 Compositional variation in rare earth elements

Rare earth element patterns vary as a function of sediment source (Hoskin & Ireland 2000) and are therefore useful in dust provenance studies (Yang et al., 2007). Seasonal trends in dust REE abundance patterns at SJP indicate that the dominant dust source(s) varies among our study sites and by sampling interval at each site. At the mid-elevation sites on both transects (SJP1084 and SJP1552), the lowest $EF_{X, Ce}$ for LREEs, MREEs, and HREEs occur during November-March (Figure 5). At SJP977, there is substantial REE (particularly MREE) enrichment during one March-July sampling period (2020) with a REE enrichment pattern similar to rock outcrops sampled at high elevation site SJP2883. At SJP1084, LREEs and several MREEs (Pr, Eu, and Gd) are enriched relative to other samples during July-November.

At the low-elevation sites (SJP483 and SJP977), no distinguishable seasonal trends in dust $EF_{X, Ce}$ are observed. At SJP977, some of the highest REE enrichments for LREEs, MREEs and HREEs occurred during one March-July interval, whereas one of the lowest enrichments also occurred during another March-July interval. The variability in REE concentrations at these sites may reflect their proximity to the respective valley floors and the influence of local emissions from sediment saltation, which would result in greater local input in a sporadic pattern not following seasonal cycles. Dust delivered to the low-elevation sites is partially derived from local bedrock with distinct REE enrichment patterns from those measured at SJP2883, and possibly another unidentified source with low REE enrichment.

The temporal variation in the range of $EF_{X, Ce}$ at the high-elevation sites SJP2883 and SJP2945 is smaller than those at the other sites (Fig. 5). At SJP2883, $EF_{X, Ce}$ exhibited less seasonal variability than at any other site, with no clear seasonal variation. These characteristics are consistent with dust sources that contribute throughout multiple seasons (Zhang & Liu, 2004). The dust delivered to high elevation sites on SJP (SJP2945 and SJP2883) is less seasonally variable than the dust delivered to low elevations. This is consistent with smaller inputs of locally derived dust at high elevation than low elevation and proportionally larger inputs from distal or anthropogenic dust sources. We interpret the variations in $EF_{X, Ce}$ across seasons and among sites as an indicator of variations in dust sources to the sites, modulated by San Jacinto Peak's complex topography (Fig. 1b) and meteorology (Figure S2 in Supplemental Information S1).

4.1.2 REE of dust in comparison to local rock outcrop and anthropogenic sources

Several dust samples share similar $EF_{X, Ce}$ patterns in rock outcrops at low and mid-elevation sites, particularly in July-November and March-July, with the outcrop characterized by enriched LREEs and HREEs and highly enriched MREEs. Differences between the outcrop and dust $EF_{X, Ce}$ patterns are observable by generally lower enrichment in the dust and an absence of the high MREE relative to LREEs and HREEs pattern in many dust samples (Fig. 5). The lack of a clear seasonal pattern in REE abundances or similarity to SJP bedrock outcrop at sites indicates that seasonality alone cannot alone account for REE variability.

The presence of anthropogenic dust within the dust samples analyzed could explain the large range in our measured dust $EF_{X, Ce}$. Variability between samples could reflect seasonality of source emission magnitude or atmospheric transport patterns. For example, sites with little seasonal variation in $EF_{X, Ce}$ (e.g., SJP2833 and SJP2945) are interpreted to receive dust from uniform sources year-round, whereas site SJP1552 receives varying proportions of dust from multiple sources on a seasonal basis.

Consistently low $EF_{X, Ce}$ at site SJP2833 (Fig. 5) indicates input from a dust source distinct from natural desert dust or SJP2833 rock outcrops. Input from urban particulate matter may also drive low $EF_{X, Ce}$ at other sites and is either overprinted with increases in natural material or decreases emission strength during intervals where $EF_{X, Ce}$ values are similar to SJP2883 rock samples.

4.1.3 Metal enrichment patterns relative to geologic sources

The north transect has higher average metal $EF_{X, Ti}$ values than the south transect at comparable elevations. Ni, Cu, and Pb have enrichment factors > 1 at all sites, implying significant enrichment of these elements relative to UCC. By contrast, Mn, Fe and Co show a mixture of enrichment ($EF_{X, Ti} > 1$) and depletion ($EF_{X, Ti} < 1$) between transects. High Cu enrichment in the dust samples could be associated with addition of material from the local rock, consistent with high Cu enrichment in the SJP2883 outcrop samples. This explanation does not account for the full extent of the Cu enrichment, as the $EF_{Cu, Ti}$ of the measured outcrop is less than 10% of that in some dust samples. Similarly, high Pb and Ni dust enrichment factors may be partially derived from bedrock composition, although for both elements the measured outcrop is depleted ($EF_{X, Ti} < 1$) relative to the UCC.

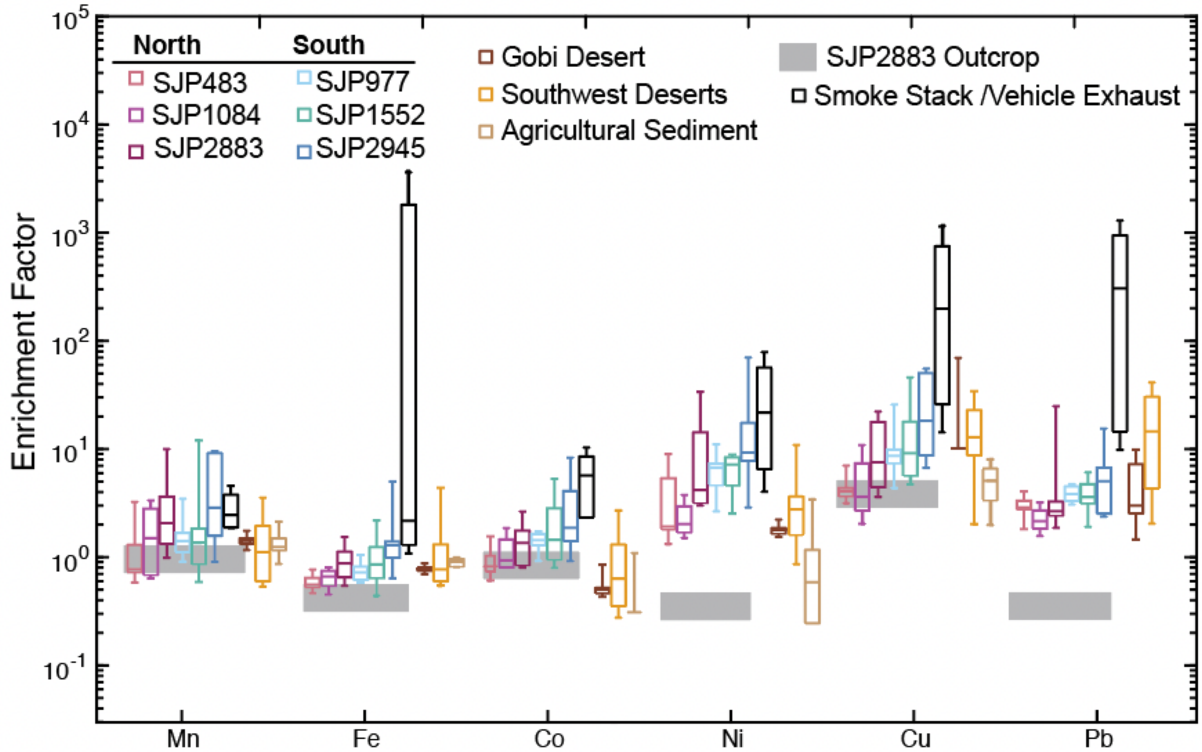

Figure 1.6. Metal enrichment factor ($EF_{X,Ti}$) for our SJP dust samples along with other potential sources of dust. The gray horizontal bar indicates the average enrichment factors from SJP2833 outcrops for each element (Wedepohl, 1995). Potential source samples include the Coachella Valley (Frie et al., 2019), Gobi Desert, GY (Jeong, 2020), Southwest Deserts (Reheis et al., 2009), agricultural sediment from the Central Valley (Chow et al., 2003), and smokestack and vehicle emission from the Imperial Valley (Watson and Chow 2001).

The combination of $EF_{X,\,Ti}$ data for metals and $EF_{X,\,Ce}$ data for REE indicates that local outcrops or desert sediment contribution does not account for the full range in composition observed in the collected dust samples analyzed here. The difference in dust metal enrichment relative to the UCC suggests that high $EF_{Ni,Ti}$ and $EF_{Pb,Ti}$ values must come from a unique source not consistently dominating the dust compositional signal throughout the year. Sites such as SJP483 and SJP1084 on the north transect have $EF_{X,\,Ce}$ values close to measured SJP2883 outcrops. At the highest sites SJP2833 and SJP2945, all REE signatures in dust are distinct from the SJP2883 outcrops, which could be attributed to input from regional anthropogenic dust sources, transpacific dust sources, or a combination of both. Variation in enrichment factors across all sites further indicates that there are several primary sources of dust in the region including a source characterized by low REE concentrations and high metal enrichment.

To probe the impact of meteorology on dust transport and deposition, we used HYSPLIT back-trajectory modeling, which estimates the path of air parcels reaching a receptor location over time (Stein et al., 2015). We conducted back-trajectory frequency plots each month for the entirety of 2021, finding large seasonal variations in air masses reaching the six study sites, as well as differences among these six sites (Text S1 and Figure S2 in supplemental information S1). The low elevation site on the north transect (SJP483) shows a clear dominance of incoming air from the Pacific Coast during the summer months, and a greater variation in the winter and spring to include air parcels from the south and east. In contrast, the highest elevation site (SJP2945) receives air masses from a wider range of locations year-round including across the western US and Mexico, with highest input from southern air masses during the summer.

4.2 Sources of dust to SJP

4.2.1 Metal enrichment

Our SJP dust samples have higher Pb and Ni enrichment factors and comparable Cu enrichment factors to dust sources in Coachella Valley (Frie et al., 2019). Natural dust sources elsewhere in Southern California show significantly lower enrichment of these metals than in our dust samples (Aarons et al., 2017; Reheis et al., 2002). This is consistent with enrichment of these metals in our dust samples by regional anthropogenic input. High concentrations of Fe, Ni, Cu and Pb have been observed in samples representative of crop burning, car exhaust, and fly ash from the Imperial Valley (Watson & Chow, 2001). Fire ash in recent California wildfires is also a source of heavy metals (Alshehri et al., 2023). Vehicle ash (soot), and to a lesser degree vegetation ash, are enriched in Ni, whereas Pb is overwhelmingly associated with urban building and vehicle ash (Alshehri et al., 2023). For the low-elevation site SJP483, the dust samples with the highest Fe, Mn, and Ni $EF_{X,\,Ti}$ were in March-July and July-November 2020, following three nearby fires. This is consistent with observations of increased Fe and Pb transport in other post-wildfire regions in California (California Air Resources Board, 2021).

Smokestacks and vehicle exhaust from the Imperial Valley have higher $EF_{X,\,Ti}$ for Fe, Co, Ni, Cu and Pb than the other potential sources considered (Gobi Desert, Southwest Deserts, agricultural sediment; Figure 6). For several elements such as Ni, $EF_{X, Ti}$ values for our SJP dust samples plot between the smokestack and natural desert $EF_{X, Ti}$ values. Notably, the $EF_{X, Ti}$ range of natural dust sources (Gobi Desert, Mojave Desert, Coachella Valley sediment) do not reach the upper limits of the SJP $EF_{X, Ti}$ values (Fig. 6). $EF_{X, Ti}$ in the SJP samples span the range of $EF_{X, Ti}$ values for crustal sources (such as Southwest Deserts) and anthropogenic sources, suggesting that natural material composes a significant portion of the SJP dust and that other sources such as wildfire ash or smokestack emissions have driven the high metal enrichments observed in dust at SJP.

4.2.2 Positive matrix factorization

PMF results show that each site contains a temporally variable amount of input from each source (Table S5). On both transects, the contribution from natural sources decreases with elevation, replaced by a mixture of the four anthropogenic sources. On average, the south transect receives more inputs from anthropogenic sources (average of 67%) than the north does (average of 50%). Similar to $EF_{X, Ti}$ and $EF_{X, Ce}$ results, PMF shows that SJP1552 receives the highest contribution from anthropogenic sources (average of 70% by PMF), with *Fly Ash* the greatest contributor (average of 25%). SJP483 and SJP1084 on the north transect have the largest inputs from *Southwest Alluvium*, consistent with their proximity to the arid Coachella Valley.

Seasonal fluctuations of source inputs occur at the sample sites (Figure S4 in Supplementary Information S1). *Industrial Ni* is relatively constant during all intervals for both transects, while *Fly Ash* is greatest in November-March for both transects. On the north transect, *Bedrock* and *Southwest Alluvium* are greatest during July-November, which may indicate an increase in emissions during the dry and windy autumn season. Together, metal $EF_{X, Ti}$, $EF_{X, Ce}$, and PMF analysis indicate that there is a significant contribution of anthropogenic sources to our dust samples.

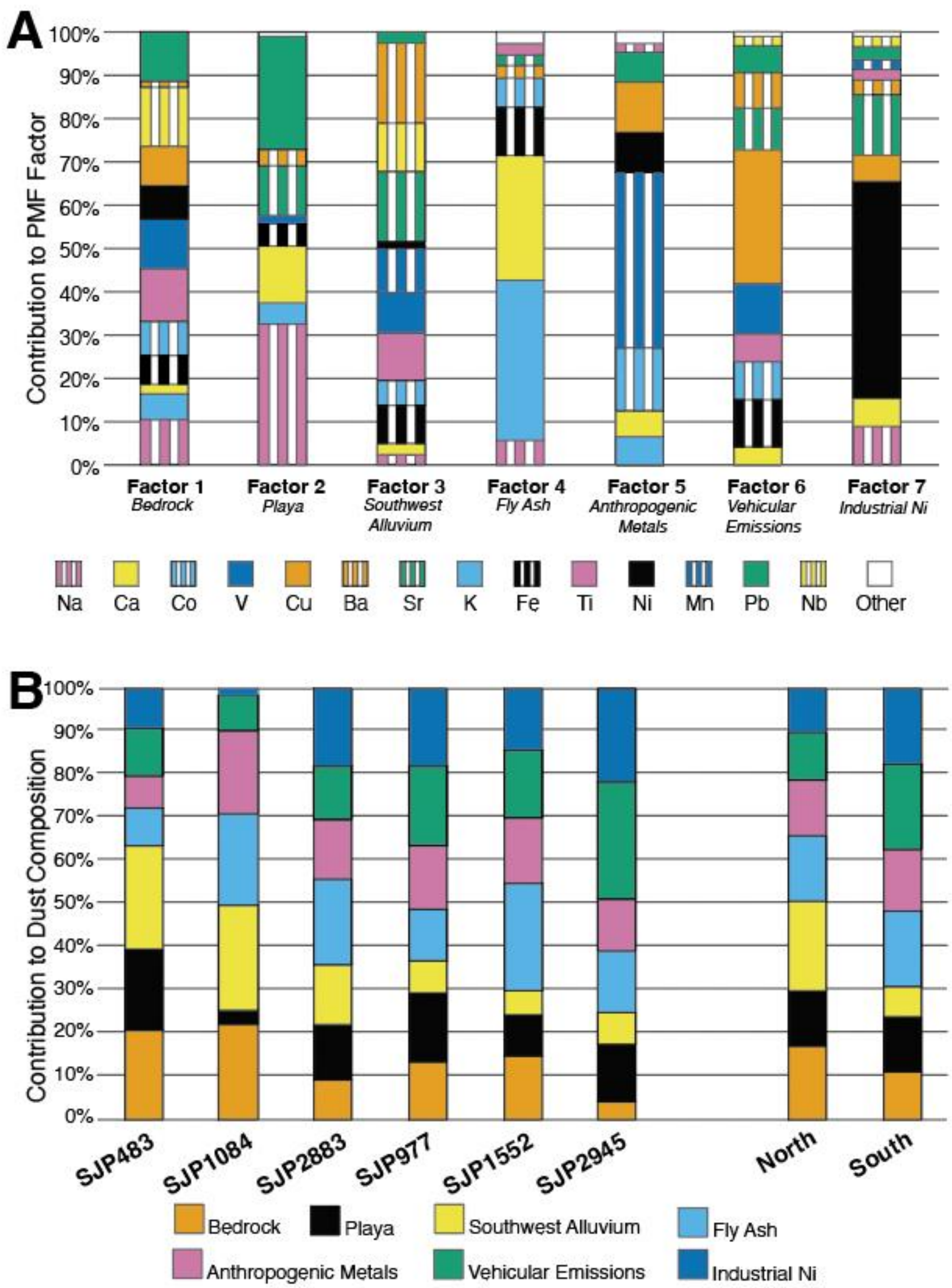

Figure 1.7. **A)** Elemental contributions to the seven modeled PMF factors. The italicized name under each factor is the source attribution (Section 4.2.2). Elements composing < 3% of a given factor are plotted as “Other”. **B)** Contributions of each factor to the average dust composition at each site. On the right, the contribution of each factor to the average composition of north sites (SJP483, SJP1084, and SJP2883) and south (SJP977, SJP1552, and SJP2945) transects are shown.

4.2.3 Human influence on dust flux and composition

In the Southwest US human induced climate change (Achakulwisut et al., 2019; Munson et al., 2011) and changes in land use (Frie et al., 2019) both actively impact modern dust transport. Anthropogenic sources of dust—estimated at ~19% of global dust emissions (Chen et al., 2018)—are highly variable in composition and location, whereas natural dust is predominantly emitted from hyper-arid to arid desert belts around the world (Choobari et al., 2014; Prospero et al., 2002). As the Southwest US is projected to become warmer and drier with increasing climate change (Ficklin & Novick, 2017), decreases in biomass and increasing drought vulnerability are also expected (Bradford et al., 2020). Models of projected changes in biomass and land use show increases in regional dust emissions with potentially large implications on ecosystems and human health (Li et al., 2021). Understanding how dust sources vary in complex topography with respect to season, climate, land use, and extreme events such as wildfire is therefore a primary goal of this research.

The San Jacinto Valley (SJV) to the west, containing the cities of Hemet and San Jacinto, is a significant dairy producing area with upwards of 200,000 cattle and agricultural fields (Committee of Experts on Dairy Manure Management, 2008). In the SJV, agricultural activity contributes 40–60% of local ambient particulate matter of 10 µm or less (Rogge et al., 2006). Further north in Central California at a series of sites along an elevation transect in the Sierra Nevada, ~18–45% of deposited dust originates from Central Asian deserts (Aciego et al., 2017), but this proportion varies significantly on a year-to-year basis (Aarons et al., 2019).

PMF analysis of our dust samples indicates a mixture of anthropogenic and natural dust sources, with anthropogenic sources contributing between 15% (SJP483, Jul-Nov 2021) and 96% (SJP2945 Nov-Mar 2021) of our sampled dust composition (n = 38). While our findings cover only a small geographic area, they provide insights into dust sources and influences of anthropogenic activity on dust deposition in this region. A multi-year investigation of particulate matter in the Imperial Valley 150 km to the southwest found that average annual PM2.5 composition is dominated by organic carbon (combustion), elemental carbon (soot), sulfate (industry), nitrate and ammonia (agriculture), and only 20% from geologic material, with up to 60% of the total PM2.5 attributable to sources in Mexicali, Mexico (Imperial County Air Pollution Control District, 2018). A study in the nearby Salton Sea Basin similarly found that anthropogenic sources account for 55 to 80% of particulate matter across five locations, with a site 25 km from SJP notably containing the highest percentage of anthropogenic trace metals peaking in May (Frie et al. 2019). We interpret the wide variation in inputs of anthropogenic sources to our samples as a function of distance from the sources and the sites' proximity to unconsolidated alluvium and (e.g., SJP483).

Estimates of the contribution of agricultural activity to global dust transport range from 25% (Ginoux et al., 2012) to between 0 and 50% (Mahowald et al., 2004). which is likely greater in highly intensive agricultural regions such as southern California. Emissions are highest during machine intensive periods, notably the summer, when soil moisture is low (Clausnitzer & Singer, 2000). Agricultural samples are geochemically similar to desert dust in contrast to the geochemically distinct signatures in anthropogenic sources like urban aerosols and smokestack emissions (Upadhyay et al., 2015), which makes it difficult to identify agricultural contributions to our sample sites.

The seasonal flux and chemical compositions of collected dust samples are influenced by wildfire ash, anthropogenic sources, and natural sediment. Using co-located environmental sensors, we found no meaningful correlation between hyperlocal environmental characteristics (precipitation, soil moisture, wind, and air temperature) and our measured dust fluxes (Text S2 and Figure S5 in Supplementary Information S1). This suggests that rather than local meteorology, dust transport and deposition to our sample sites are more strongly influenced by variations in anthropogenic land use, and strongly modulated by discrete events such as wildfires. Projected trends due to climate change including increased wildfire frequency and intensity (Bryant & Westerling, 2014) along with the continued disturbance of natural surface sediments through development and land use in the region (Smith et al., 2023) imply that dust fluxes are likely to increase, and its composition will be increasingly modulated by anthropogenic sources.

The co-occurrence of California statewide stay-at-home orders in response to the COVID-19 pandemic and our sampling campaign must be acknowledged, as it may have led to an alteration in regional dust sources. Ambient $PM_{2.5}$ concentrations were found to decrease in response to stay-at-home orders starting in California on 19 March, attributed to the reduction of anthropogenic emissions from sources such as vehicle traffic and industrial activities (Jiang et al., 2021). Overall, although our sampling intervals do not correspond with specific lockdown periods or policies, we may assume that COVID-19 pandemic related changes to human activities may have caused our study to underestimate the predominance of anthropogenic dust deposition on SJP regarding pre-pandemic and post-lockdown periods, as seen in Environmental Protection Agency (EPA) monitoring sites (Pan et al., 2019). The extent of reduction of anthropogenic dust flux in montane environments such as SJP is beyond the scope of this study and would require additional data collection.

4.3 Wildfire impact on dust emissions and heavy metal enrichment

Climate change throughout Southern California over the past four decades has led to an increase in July-November temperatures and decreases in July-November precipitation (~30%), which will likely increase seasonal dust fluxes and increase the likelihood of wildfires (Wagenbrenner et al., 2013; Goss et al., 2020). During the July-November 2020 sampling period, wildfires burned more than 29,000 acres next to the north transect in the Coachella Valley (CAL FIRE, 2021) (Fig. 2). The two north transect samplers that were active at the time show above average dust flux (percent of annual mean flux: 450% for SJP483, and 191% for SJP2883). At SJP483, the July-November 2020 spike in dust flux represented 48% of the total dust flux in the 2020-2022 sampling interval despite representing only 14% of the sampling time. These disproportionate inputs may be explained by wildfire-related increases in regional dust emissions, including ash and sediment from burn scars, loss of vegetation, decreased soil moisture, and fire-induced weathering (Yu & Ginoux, 2022). Higher metal enrichments accompanied the high fluxes following fires (Fig. 4).

Increased metal transport due to fire has been attributed to the mobilization of elements previously sequestered in soil (Alshehri et al., 2023), as decreased biomass and surface stability interact with high wind speeds during fires (Wagner et al., 2018). Post-wildfire land surfaces, known as burn scars, can transform a stable land surface into a dust source for multiple years following a fire event, due to loss of vegetation, decreased soil moisture, and destabilized surface sediment (Duniway et al., 2019). A decade-long study of Western US wildfire plumes revealed significant enrichments of dust (coarse and fine) in both the local fire region and downwind, including changes in trace element composition and a 22% increase in fine soil transport in California (Schlosser et al., 2017). Concentrations of Ni and Pb (enriched in our samples) from aerosols following the December 2017 Thomas Fire in Ventura and Santa Barbara Counties were comparable to or exceeded metal concentrations of fly ash and Los Angeles urban aerosols (Kelly et al., 2021). Autumn 2020 fires burned through mostly natural and some anthropogenic land surfaces (CAL FIRE, 2021). The resulting ash—which included biomass and manufactured materials—is strongly enriched in metals, particularly ash from anthropogenic materials (Alshehri et al., 2023). Our measurements show that wildfires significantly increased dust fluxes and metal abundances at our study sites.

To the extent that wildfires and anthropogenic dust sources grow more pronounced in the future, our results suggest that dust fluxes and metal abundances are likely to increase in the future (Bryant & Westerling, 2014, Zhong et al., 2021). Increased dust flux may impact ecosystems and human health (Jayarathne et al., 2018), which is already a concern in the region (Johnston et al., 2019). Our results demonstrate the significance of non-natural (desert) dust sources and composition in the current regional dust cycle and the importance of continued long-term monitoring and identification of regional dust sources.

4.4 Mechanisms of dust deposition and comparisons to ambient PM concentration

Our data show a positive correlation between elevation and anthropogenic dust deposition (defined as Factors 4-7 in our positive matrix factorization; Section 2.5). On the north transect, anthropogenic dust contributed on average 36% of the fine dust at the low-elevation site SJP483, 51% at the mid-elevation site SJP1084, and 64% at the high-elevation SJP2883. On the south transect, anthropogenic dust contributed on average 63% of the fine dust at the low-elevation site SJP977, 70% at the mid-elevation site SJP1552, and 75% at the high-elevation SJP2945.

We interpret the increased proportions of anthropogenic dust at higher elevations as a reflection of the geographic location and physical and chemical characteristics of the dust source, and its ability to be lofted into high altitudes in the atmosphere. Mixed emissions from factories, smokestacks and regional urban centers (e.g., Los Angeles) as well as far travelled dust from Asia are likely to be well integrated in air parcels capable of global transport (Chen et al., 2018) and hence likely to be transported to all sites at SJP. In comparison, hyper-local sediment in the Coachella Valley is less likely to reach higher elevations due to orographic entrapment of surface level air parcels. This was observed in a study of dust storms just south of SJP in the same valley, where dust plumes were confined to < 2000 m above ground level (AGL), with extinction or concentration maximums much lower (e.g. 100 and 500 m AGL; Evan et al., 2023). In this mountain-bounded valley, surface level air containing newly deflated dust is unable to move upwards to the mountain slopes and is confined to a relatively shallow layer, referred to as trapped lee waves (Evan et al., 2023). Similar conditions limiting the vertical mixing of dust plumes was found in similar environments such as Owens Valley in eastern California (Grubišić et al., 2008) indicating that our findings highlight a mechanism which may limit the deposition of local deflated sediment in mountainous regions. At lower elevations, where the input of local desert dust is higher, the proportion of far traveled dust is smaller.

In our study we used multi-month sampling intervals, following the approach in previous studies of dust in montane environments (Aciego et al., 2017; Aarons et al., 2019; Heindel et al., 2020; Munroe et al., 2020). This results in measurements of dust flux and composition that are averaged over several months, which is beneficial for our study because it provides a longer-term perspective on dust chemical variability in montane environments than the measurement of ambient air dust concentrations or specific depositional events. Because this approach yields rates and compositions that are averaged over several months, it cannot isolate short-term events such as dust storms or record ambient air dust concentration, quantities that are more relevant for studies focused on dust's impact on human health (Bell et al., 2007). Our research probes the intersection of long-term dust deposition and short-term dust source variability.

To explore the relationship between the dust fluxes measured in our passive dust samplers to those measured in modern active air samplers, we compared our measured dust fluxes to the fluxes of particulate matter less than 10 µm in size (PM10) measured at high-volume air samplers (US Environmental Protection Agency) within 20 km of our sites, during the same time intervals (Figure S6; text S4). Active air samplers differ from passive samplers in that they draw in air rather than allowing particulate matter to gravitationally settle out of the atmosphere.

We found a range of moderate negative (-0.47) to moderate positive (0.56) Pearson's (r) correlations between the mass of fine dust collected in our passive dust samplers and the mass of PM10 collected in the active dust samplers, indicating the absence of a strong correlation between our measured dust fluxes and the PM fluxes at those sites (Figure S6). The negative correlations at the higher elevation sites SJP1084 and SJP2883, along with the moderate positive correlation with at the low-elevation site SJP483, support our interpretation of poor vertical mixing of surface level dust in the Coachella Valley. This analysis highlights that our dust measurements at high elevations are not well correlated with the air quality characteristics that affect public health at low elevations, where most people live in the study area. Instead, our passive dust collection method is effective in distinguishing longer term spatial and temporal variations in dust deposition, which are relevant for nutrient supply and cycling in montane soils.

5 Conclusion

We present a new multi-year record of dust deposition that quantifies spatial and temporal variability of trace element composition and flux across two elevation gradients on a prominent peak in Southern California. The measured dust compositions indicate similarity to regional anthropogenic dust sources and showed that dust fluxes tend to increase after nearby wildfires. PMF analysis supports this interpretation and provides estimates of the contribution from anthropogenic and natural sources to measured dust samples. Metal enrichments in regional anthropogenic dust sources and local wildfire ash are higher than those in local rock outcrops and regional deserts. High Ni and Pb enrichment in our dust samples are consistent with high concentrations of Ni and Pb measured in previous studies of fire ash, urban PM, and industrial emissions elsewhere.

Seasonal patterns in dust flux and composition show that dust sources are not consistent year-round and are likely controlled by climate patterns (e.g., precipitation and wind), wildfire seasons, and anthropogenic land use (e.g., industrial activity). Dust deposition rates in several sites reach a maximum in the July-November interval and a minimum in the November-March interval. Variable trends in dust composition and dust flux at our sample sites on both transects demonstrate the influence of topography, elevation, and seasonality on dust deposition. Understanding the role of human impact on dust emissions would benefit from developing long-term trends in the pre-colonial Holocene dust cycle in the region.

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Chapter 1 Appendix

Text S1

To determine the relative contribution of local versus globally transported dust, we conducted month-long individual atmospheric back trajectory modeling for two dust collection sites for the entirety of 2021 (Figure S2 in Supporting Information S1) using NOAA HYSPLIT (Stein et al., 2015). For both sites, meteorological inputs were from the Global Data Assimilation System (GDAS) with a grid resolution of 0.5˚ x 0.5˚, a trajectory duration of 120 hours, an endpoint output frequency of 60 per hour, and the air mass altitude set to 1 m above the respective site elevation. While not diagnostic of dust transport from these regions, the back trajectory modeling can yield insight into pathways that air parcels passed over prior to reaching the samplers and therefore potential dust source regions. Calculation and summation of multiple trajectories over a set period of time (a calendar month in this case) yields a heatmap which indicates the number, or frequency, of air parcels intersecting any location within the studied area. Modal incoming wind direction coming down the coast from the northwest is observed for all sites, reflecting wind-driven seasonal coastal upwelling (Checkley & Barth, 2009), and agrees with local meteorological windrose data (Fig. 2). From April to September, mid and low elevation sites on the north transect receive air parcels predominantly from the northwest through the San Gorgonio Pass, a region less than 800 m in elevation which separates the San Bernardino Mountains (>3,500 m) to the north and the San Jacinto Mountains to the south. Site SJP483 receives substantial incoming parcels from the south and inland only in October-January, which may pass over Mexico and Central America before coming up the coast and making landfall once again. The mid-elevation site SJP1084, adjacent to SJP483, displays a similar pattern. The high-elevation site SJP2945 is least impacted by regional topography receives more direct air parcels from over the Pacific and show greater input from air masses to the south and inland. Trajectories at SJP2945 do not show a modal direction, but in general receive more air masses from the south during the summer and from the Pacific in the winter. Differences in HYSPLIT models for SJP483 and SJP2945 can be seen both in seasonal directionality and degree of uniformity.

Text S2: *Assessing the correlation of flux and local environmental data*

Our measured dust fluxes show little to no correlation with local environmental conditions (Fig. S5). While air temperature (Fig. S5c), soil moisture (Fig. S5b), total rainfall (Fig. S5d), and the total number of days with rainfall (Fig. 5a) are all impacted by seasonal climate, they do not vary with the seasonal variations in measured dust flux. It is theoretically possible that dust fluxes are related to local environmental conditions at the scale of a single dust deposition event, but, if any such relationships exist, they are not apparent in our data. Moreover, they would be difficult to resolve given the sampling intervals over which the dust samples were collected (at least four months, determined by the need to collect enough dust for geochemical analyses and the feasibility of accessing the sites in all seasons). These sampling intervals are far longer than the duration of a single dust deposition event (minutes to hours), which implies that the dust collected in any one sampling interval may be a mixture of dust deposited during multiple deposition events. If dust flux depends on local environmental conditions differently during different dust deposition events, then any relationships between dust flux and local environmental conditions may be obscured by the total mass of dust accumulated during the sampling interval and by temporal variations in local environmental conditions.

To probe shorter timescale relationships, local weather stations wind speed data are used to see if the number of high wind events correlate with flux. We applied a commonly used average wind speed threshold value to emit dust from a surface is 6.5 m s^{-1} (Tegen and Fung 1995). For samples collected during 2021, the relationship between our recorded dust flux and a wind speed threshold—the number of 10-minute measurements during the respective sampling interval with wind gusts greater than 6.5 m s^{-1}—from nearby stations is plotted in Fig. S5e. Three stations were used: SE577 in Snow Creek, CA (for in SJP483 and SJP1084), KNWC1 near Idyllwild, CA (for SJP483 and SJP1084), and MSJC1 at the top of San Jacinto Peak (for SJP2883 and SJP2945) (Horel et al., 2002). The season with the highest number of wind gusts capable of emitting dust was the winter (Nov-March) interval, while the highest flux occurs during the summer or autumn collection. However, at SJP2883, SJP1552, and SJP2945 there is a correlation between higher flux and number of high wind gusts in the summer months, and lower in the autumn. Low-elevation sites do not share the same correlation, nor does SJP1084. We conclude that while there might be a local correlation between higher wind speed events and flux, it does not explain our findings which are more likely related to elevation or topography.

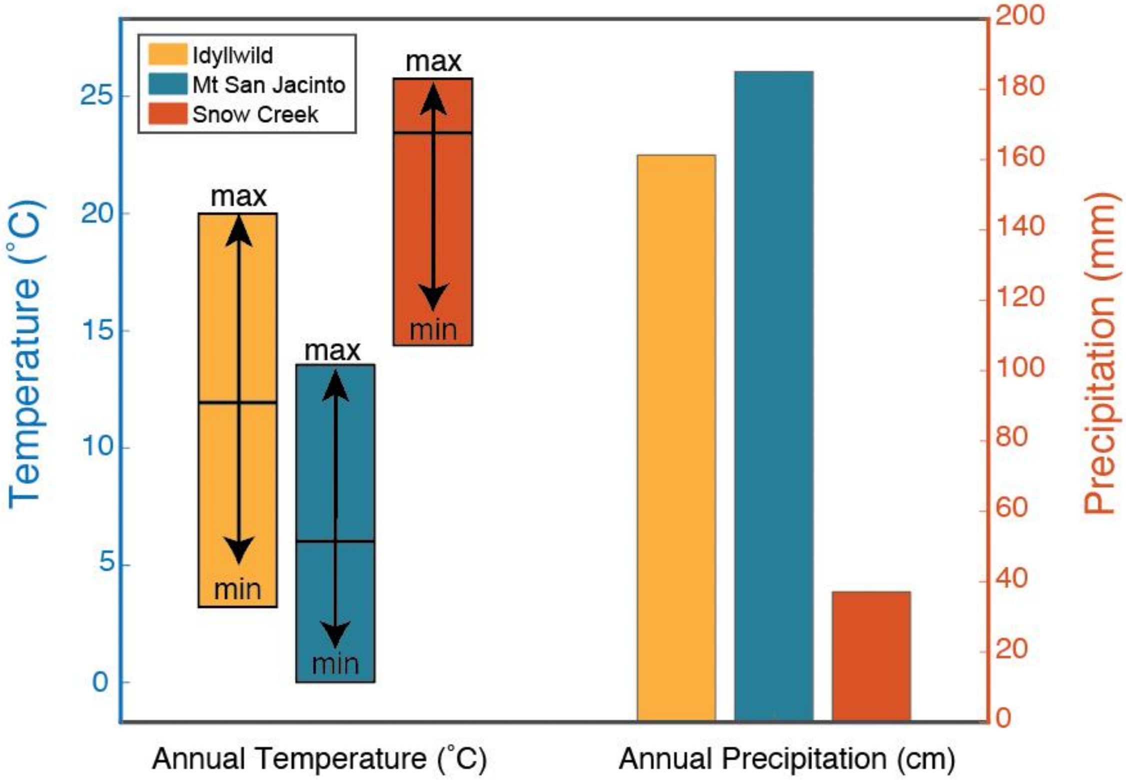
The low- and mid-elevation sites on the south transect are ~3 horizontal km apart from one another, more than the than the 1.6 km separating the low and mid-elevation sites on the north side. While SJP977 is 575 m lower in elevation than the mid-elevation site SJP1552, is it not significantly closer to local dust sources (Fig. 2). Average air temperature for each interval at each site was calculated by taking the average of each 15-minute interval reading for the entire period, while total precipitation during each interval is the sum of all readings and measured in millimeters collected. During each respective interval, SJP1552 was 0-5˚C colder than SJP977, while the low site received more rainfall (161-829 mm per year) than the mid-elevation (26-611 mm per year). The two high-elevation sites receive most of their precipitation as snowfall, which may be underestimated by the rain gauges. SJP2945 is the coldest with an average November-March temperature of -0.3˚C, March-July temperatures of 10.4˚C and average July-November temperature of 7.8˚C. The high-elevation sites located in subalpine forests are significantly different from the arid desert climate of the low and mid-elevation north side, and the arid sparse oak vegetation on the southside. Microclimate factors alone may lead to significant differences in the physics of dust deposition (Kok et al., 2012) and the mechanism (e.g., wet vs. dry deposition).

Text S3

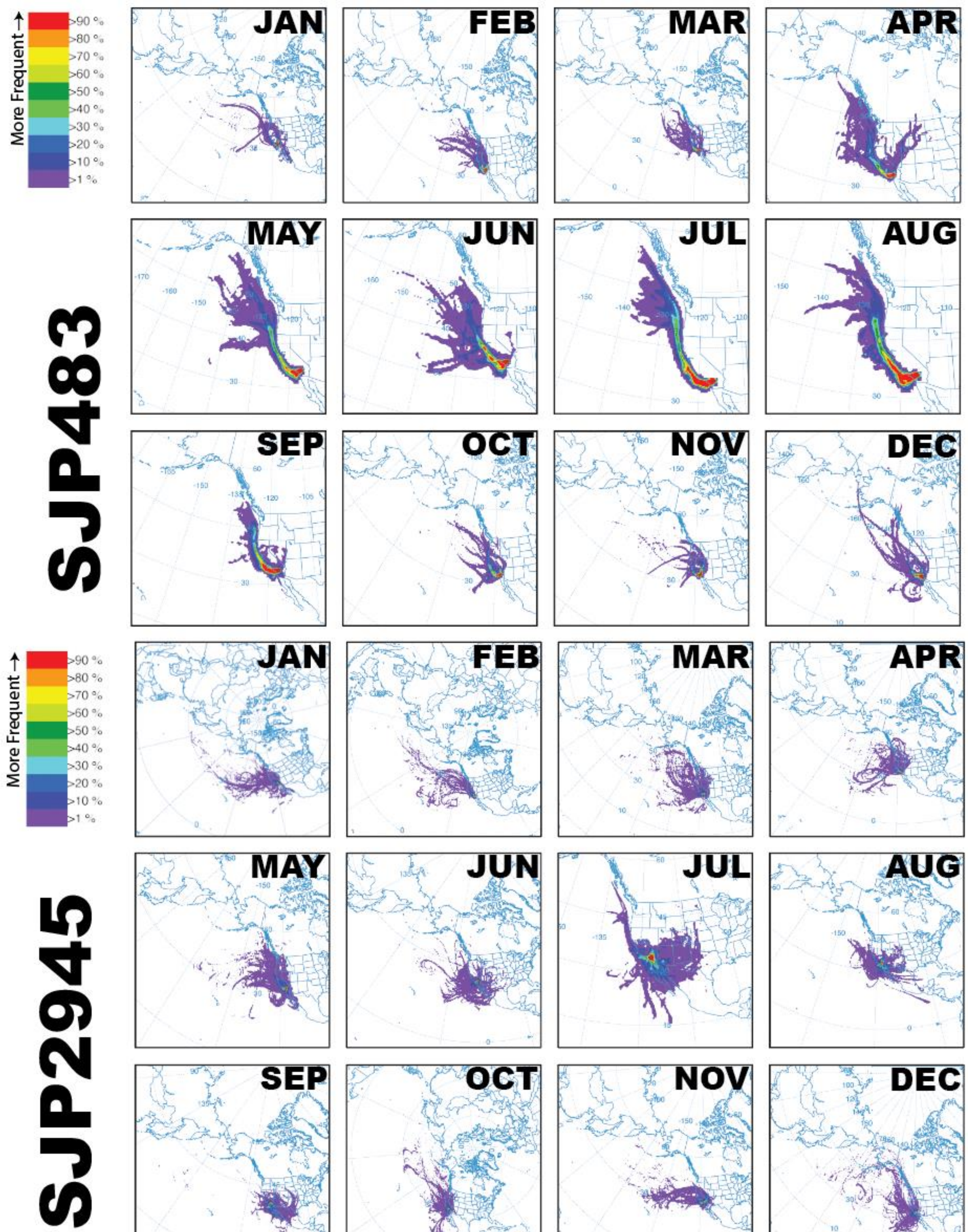
Positive Matrix Factorization (PMF) was performed using the EPA PMF 5.0 (P. Hopke, 2000). We selected fourteen elements from our chemical dataset to include in the model: Na, Ca, Co, V, Cu, Ba, Sr, K, Fe, Ti, Ni, Mn, Pb, and Nb. The raw data was initialized to ensure there are no negative values and the file was in an appropriate format. EPA PMF 5.0 requires an uncertainty matrix with the same dimensions as the concentration matrix, which provides an estimation of uncertainty or errors from sampling or measurement for each species to be modelled. Our uncertainty matrix was created using the analytical uncertainty from the ICP-MS measurement following Reff et al. (2007). The model is optimized using the sum of the squares of the scaled residuals (Q) and the expected Q (Qexp) based on the number of species (elements) (Paatero & Tapper, 1994). A low Q/Qexp (~1) indicates that errors in the input data have been appropriately estimated by the model, and the number of factors is suitable. The addition of factors will continue to decrease the value of Q/Qexp, so the amount of decrease when adding a factor indicates if additional factors meaningfully explain the data variance and source, or if it splits a real sources into two non-existing factors (Guha et al., 2015). The model was run numerous times with a different number of output factors (2-9) to locate the optimum number with a significant decrease in Q/Qexp value (Paatero & Tapper, 1994). In our case the Q/Qexp value for 7 factors was 0.8, which was markedly lowered than for 6 factors, while 8 and 9 factors showed only a slight improvement, thus 7 a factor was selected. All 20 PMF model runs converged, and all species categories were "strong", determined based on a high signal-to-noise ratio (S/N). Then, the bootstrap method with 100 runs yielded the g matrix (factor fingerprints) and the f matrix (abundance of each factor in samples) which was used for analysis and interpretation.

Text S4

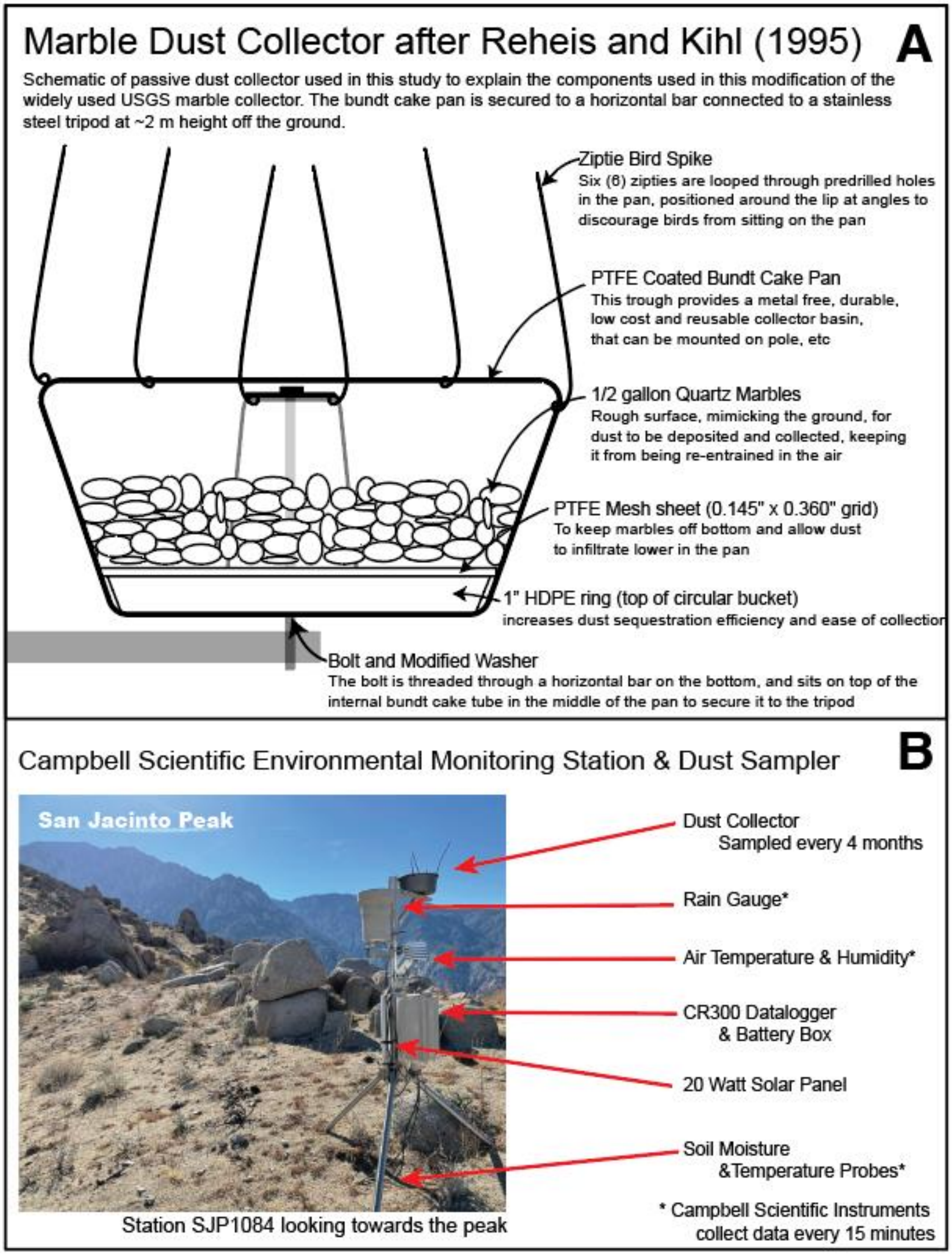
Of interest is the correlation of dust deposition, as measured in this study, with ambient particulate matter (PM) concentration and mass, such as measured an active air sampler. To compare our passive dust collectors to nearby ambient air samplers, we accessed daily average PM10 concentration data at two US Environmental Protection Agency monitoring stations in the Coachella Valley from the Air Quality System Data Mart (US Environmental Protection Agency). The first station in Palm springs is 13 km from SJP 483 (ID: 06-065-5001), and the second at the Banning Airport is 18 km (ID: 06-065-0012). The total average daily PM10 mass (g) sampled was calculated by multiplying the average daily concentration (μg/m^3) by the lower range of the monitor's air flow rate (36 CFM) as conservative estimate of the volume of air sampled during each 24-hour period. For the same time interval as our samples, the total mass measured was calculated by summing the calculated daily PM10 mass (Figure S6). The correlation between the resulting PM10 mass records and our dust mass from the three collectors on the north transect of SJP in the Coachella Valley was assessed using the Pearson's (r) correlation coefficient (Freedman et al., 2007). As shown in Figure S6, the greatest correlation was between the Banning Airport site and SJP843, which was moderate (.56, p = .249), and the least was between Palm Springs and SJP2883 with a moderate inverse correlation (-0.52, p =.522). Overall, we conclude there is no statistically significant correlation between our dust deposition samples and ambient PM10 in the Coachella Valley.



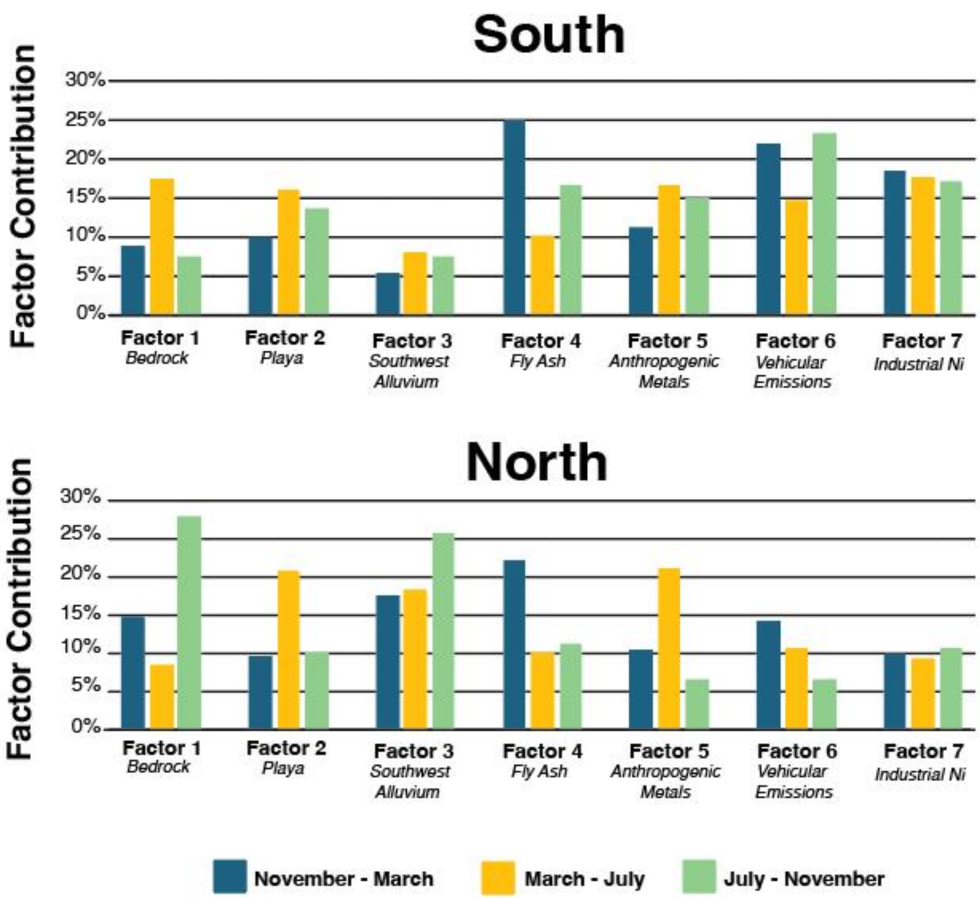
Supplemental Figure 1.1. The differences in climate from the base of each transect to the summit of San Jacinto Peak. At left, bars indicate the maximum and minimum average monthly temperatures between 1943 and 2016 for Idyllwild, CA (Idyllwild) on the south transect close to SJP1552, at Mt San Jacinto Weather Station (Mt San Jacinto) close to SJP2883 between 1991 and 2020, and Snow Creek (Snow Creek) at the base of the north transect close to SJP483 between 1939 and 2007. Horizontal black lines in the middle of each bar indicate the mean average monthly temperature over the monitoring periods. On the right, average annual precipitation in cm for each site for the same period of record is plotted for the same stations. Data is from NOAA and the Western Regional Climate Center (Vose et al. 2014; Western Regional Climate Center, 2023).



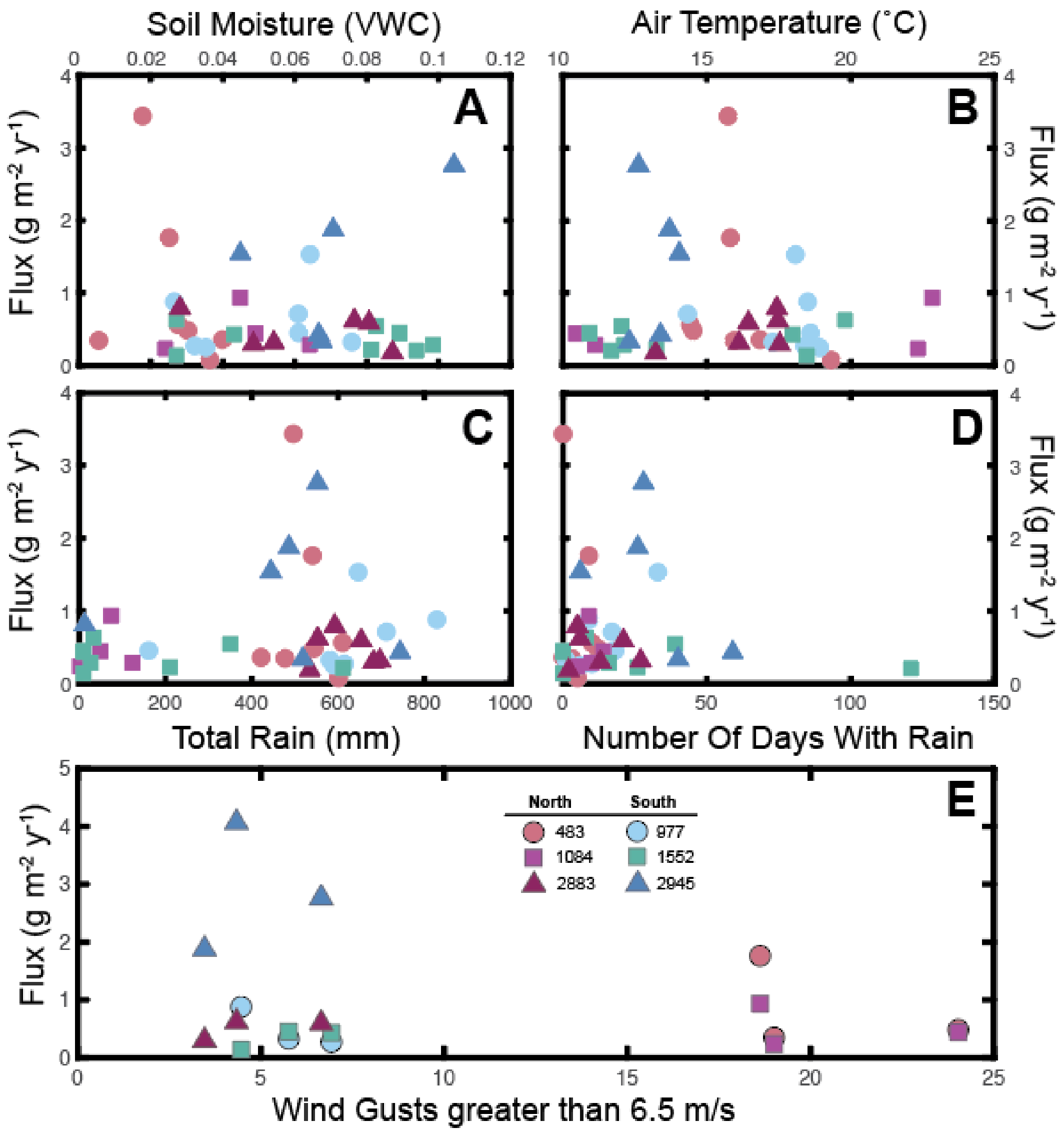
Supplemental Figure 1.2. Monthlong HYSPLIT back trajectory frequency plots for the entirety of 2021 for two sites: high-elevation southside SJP2945 and low-elevation northside SJP483. Colors signify the frequency of air parcel trajectories that intersected with each 1 km gridded cell, with warm colors (red) indicating air parcels often (>90% of the time) came over the cell, and cool colors (purple) indicating air parcels seldom (>1%) intersected with the cell.



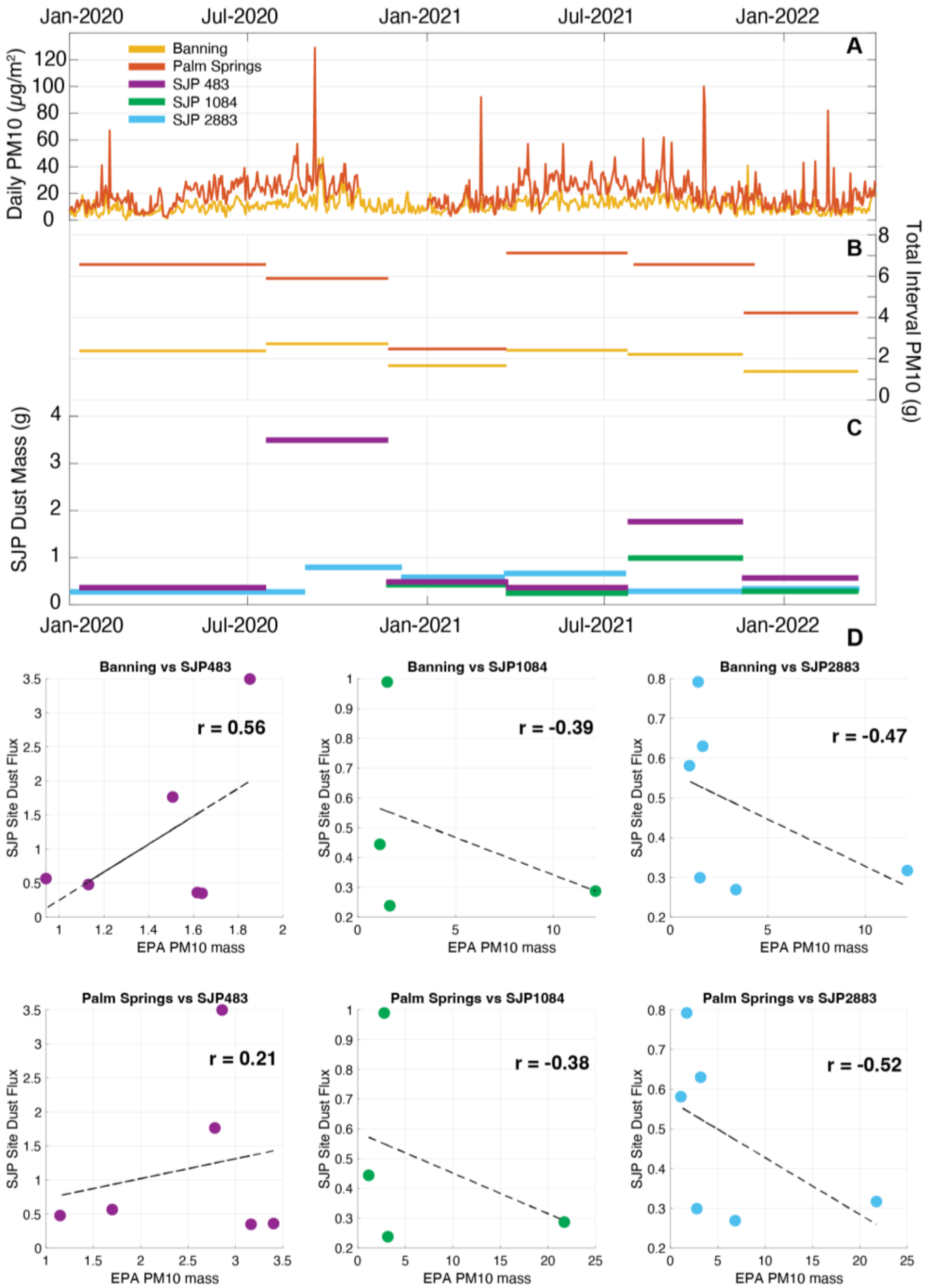
Supplemental Figure 1.3. Schematic and details of the field sampling method and data collection.



Supplemental Figure 1.4. Both panels contain results from PMF analysis with **a)** showing the average factor contribution to sites on the south transect (SJP977, SJP1552, and SJP2945) by the three sample intervals, while **b)** shows the average factor contribution to sites on the north transect (SJP483, SJP1084, SJP2883).



Supplemental Figure 1.5. Relationship of dust flux to climate and environmental factors. **A.** Dust flux versus soil moisture content in volumetric water content (VWC) at 10 cm depth. **B.** Dust flux versus average annual air temperature. **C.** Dust flux versus number of days with rainfall during the sample period. **D.** Dust flux versus the number of days with rain at each site during the sampling period. **E.** Dust flux versus the number of 10-minute intervals with a maximum wind gust speed greater than 6.5 m s^{-1}, normalized to the number of days during the collection period. For sites SJP483 and SJP1084 the weather station SE577 in Snow Creek was used, for SJP997 and SJP1084 the weather station KNWC1 near Idyllwild was used, and for SJP2883 and SJP2945 the weather station MSJC1 and the top of San Jacinto Peak were used for wind data (accessed via University of Utah, Mesowest (Horel et al., 2002).



Supplemental Figure 1.6. Correlation between PM10 monitors and dust deposition collectors. **A)** Average daily PM10 concentration (µg/m^3) from two EPA monitors in the Coachella Valley. **B)** Calculated total measured mass of PM10 (g) for intervals corresponding to our SJP dust samples. **C)** Collected dust mass (g) at the three SJP sites on the north transect in the Coachella Valley. **D)** Scatter plots of the Pearson's (r) correlation coefficient between the two EPA stations calculated dust mass versus the SJP site dust mass for the respective intervals.

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Chapter 2 Differential Health Impacts of Short-term Particulate Matter Exposure in Arid and Semi-Arid Regions

Abstract

Particulate matter (PM) exposure is a leading risk factor for cardiovascular and respiratory morbidity and mortality across the world. While arid and semi-arid regions account for more than 40% of terrestrial land, there is a paucity of research investigating the relationship between PM exposure and health outcomes or how risk varies across different populations specific to these climate regions. To address this, we performed a scoping review to assess the extent of literature in arid and semi-arid regions that consider such morbidity and mortality risks from short-term PM exposure. Our review identified 102 studies published after 2012 located within four arid climate classifications. The meta-analysis found a pooled overall increased relative risk (RR) of both cardiovascular and respiratory mortality and morbidity, with an 8% higher risk of morbidity in hot arid regions (1.11, 95% confidence interval (CI): 1.05-1.17) compared to cold semi-arid regions (1.02, 95% CI: 1.00-1.03) for adults (18-64). Further, we assessed effect modification (age, sex, population density and season) and found weak evidence of increased risk for both males and adults. The studies included in this review highlighted compounding risks, such as dust storms, wildfire, and extreme temperature, however there was a significant lack of studies situated in highly populated arid regions, with none identified in Africa. Given the public health burden from PM exposure and the impact of climate change and anthropogenic activities which destabilize land-surfaces and increase dust emissions, additional focus is owed to demographic effect modifiers and the role of climate specific risks.

1. Introduction

Air pollution, and specifically particulate matter (PM) pollution, constitute the largest risk factors contributing to the global burden of disease and deaths in 2021 (Brauer et al., 2024; Health Effects Institute, 2024). It is estimated that approximately 4.7 million deaths globally could be attributed to ambient, or outdoor, fine particulate matter ($PM_{2.5}$, or particles 2.5 micrometers or less in diameter) exposure (Health Effects Institute, 2024). PM is commonly studied in health research because it has well-documented, severe adverse health effects, differentially impacts health due to its source and chemical composition, and is a widespread pollutant generated by both human and natural sources (WHO, 2021a). A growing body of epidemiological evidence links both short- (over hours or days) and long-term (over months or years) exposure to PM with a wide range of acute and chronic health endpoints, particularly in respiratory and cardiovascular systems (e.g., Orellano et al., 2020; Rajagopalan et al., 2018; Atkinson et al., 2015). As PM can penetrate the respiratory system when inhaled, lung tissue becomes irritated and immune responses are triggered, leading to inflammation (Thompson, 2018). Fine particles can enter the bloodstream, resulting in systemic inflammation and oxidative stress that affects blood vessels, increases blood pressure, and promotes clot formation. These processes elevate the risk of respiratory infections, asthma exacerbation, and cardiovascular events like heart attacks and strokes, for example. Understanding the health impacts of short-term PM exposure is critical for policy changes as such knowledge informs emergency responses targeting vulnerable groups and can combat significant increases in emergency room visits and hospital admissions.

PM exposure is a serious public health concern at the global scale. It is estimated that 99% of the world's population has an annual mean exposure to $PM_{2.5}$ at concentrations higher than the 2021 World Health Organization (WHO) global air quality recommended exposure limit of 5 $\mu g/m^3$ (Yu et al., 2023). However, research has consistently found that there is no amount of PM exposure that could be considered safe at the population level (Rajagopalan et al., 2018). Although ambient $PM_{2.5}$ concentrations have decreased or stabilized in many (predominantly high-income) countries over recent decades, pollution levels remain high in Asian, African, and Middle Eastern countries (Health Effects Institute, 2024). As more people concentrate in urban areas globally, cities tend to experience higher pollution levels due to increased human activities and anthropogenic sources of PM emissions. Dense populations result in higher levels of deleterious pollution exposure, resulting in a greater public health burden. Human exposure to PM varies both temporally (e.g., day of week, season) and spatially (climate, land cover, PM source, proximity to emission source, urban vs rural). Climate change is expected to increase global particulate matter emission (Park et al., 2020) primarily through increasing droughts, with arid and semi-arid regions especially susceptible (Safriel et al., 2005). The combination of land-use (e.g., urban development, agriculture, mining, etc.) and anthropogenic emissions (fossil fuel combustion, biomass burning, etc.) alter PM concentrations and increase exposure risk for communities in arid and semi-arid regions.

Arid and semi-arid regions cover 30.2% and 15.2% of the terrestrial surface, respectively (Peel et al., 2007; Beck et al., 2018), supporting a combined population of 2.5 billion people (Scholes, 2020; Gaur et al., 2018). While anthropogenic activity drives significant short-term changes in PM emissions (Chen et al., 2018), increases in aerosol optical depth (AOD) between 2001 and 2010 are linked to long-term meteorological variability in arid and semi-arid regions such as the Middle East and Northern Africa (Pozzer et al., 2015), indicating the role of climate on PM emission increases. While semi-arid regions have already expanded in areal extent by 7% from 1948–1962 to 1990–2004 (Huang et al., 2016), climate models of the 21st century indicate these regions are expected to become more arid with decreased rainfall and increased evaporation (Scholes, 2020). In a scenario of 3°C global average temperature increases due to anthropogenic climate change, 5.2 million km^2, or 3.5% of total terrestrial land, is estimated to become arid (Spinoni et al., 2021). Between 1990 and 2020, 77.6% of the Earth's terrestrial land become drier (Vicente-Serrano et al., 2024). Further contributing to PM emissions, rapid population growth in arid and semi-arid regions has led to widespread land-surface disturbance during urbanization, industrial and commercial operations (Tsiouri et al., 2015), and agricultural activities (Guan et al., 2016). During the period 1982 to 2015, the combination of unsustainable land-use and climate change led to desertification of 6% of global dryland area (Mirzabaev et al., 2022). Deserts, unique to arid regions, currently emit more than half of the global PM mass (Kok et al., 2017), and exposure to desert dust specifically has been associated with adverse human health impacts (Zhang et al., 2016; Goudie, 2014). In the Southwest United States, climate change-induced increases in aridity are projected to increase the emissions of fine dust by 57% (Achakulwisut et al., 2019). Arid regions are especially vulnerable to soil disturbances, leading to new and emerging dust sources (Guan et al., 2016). In these regions, climate change is driving more frequent and intense extreme heat events, which often coincide with pollution episodes due to shared underlying meteorological drivers (Schnell & Prather, 2017). Additionally, heat can intensify the formation of PM through chemical reactions that occur more readily in hot, sunny conditions. Together, extreme heat and PM pollution have been shown to have a synergistic effect, worsening the health impacts beyond the sum of their individual contribution (Rahman et al., 2022; Schnell & Prather, 2017). Despite this, there is an absence in the literature assessing this compound exposure in arid environments.

Both exposure and vulnerability to PM and related health impacts are not equally distributed across space or populations. Previous work has shown that some population subgroups are systematically exposed to higher levels of PM than others (Jbaily et al., 2022; Ma et al., 2023), and that some are more susceptible to developing health outcomes from PM exposure compared to others due to risk factors (Bell et al., 2013). In complex settings such as urban environments, the interaction between variable PM composition exposure and the differential health outcomes is poorly understood at present, although necessary to reduce the overall and disproportionate health burdens experienced globally. Previous literature reviews of health impacts from short-term PM exposure show that some populations have an increased risk of PM-related health effects, often termed 'susceptibility' or 'vulnerability' (Sacks et al., 2010; Bell et al., 2013; Markri et al., 2008). Population effect modifiers such as age, sex, ethnicity (Jbaily et al., 2022), socioeconomic status (SES, Hajat et al., 2015; Jbaily et al., 2022), minority populations (Yang et al., 2022), and migration status (Fong et al., 2022; Liang et al., 2019) are found to be statistically significant indicators that increase the risk of mortality and morbidity. For example, in the United States non-Hispanic Black people are 3.47 more likely to die from $PM_{2.5}$-attributable cardiovascular disease than non-Hispanic white people (Ma et al., 2023).

Further, both environmental and sociodemographic characteristics of semi-arid and arid regions increase the potential for large and disparate health impacts related to PM. Roughly 72% of drylands and almost 100% of hyper-arid land are located in developing countries (Huang et al., 2017; Hulme, 1996; Safriel et al., 2005). Semi-arid regions are thought to be most vulnerable to destabilization and ecosystem loss as they are at the intersection between population growth and fragile land surfaces (Safriel et al., 2005). Broadly, rural communities (Singh & Chudasama, 2021; Sternberg & Edwards, 2017) and women (Yadav

& Lal, 2018) are identified as being most vulnerable to climate change-related impacts on environmental health in arid regions. This results from the intersection of multiple factors related to poverty, limited infrastructure, access to healthcare, gender norms, environmental exposures, and evidence of enhanced susceptibly to PM during pregnancy (Sapkota et al., 2012). Despite evidence that 80% of people who are exposed to unsafe average annual concentrations of $PM_{2.5}$ live in low- and middle-income countries (Rentschler & Leonova, 2023), the majority of research has focused on PM exposure in high income countries (Jaganathan et al., 2019; Yegros-Yegros et al., 2020). Separately, there is a lack of studies in many arid regions (e.g., Imane et al., 2022), leading to gaps in knowledge and potentially inaccurate effect estimates. For example, a study in Qatar found excess mortality from fugitive particulate matter to be 11 times greater than World Health Organization models (Hassan et al., 2022), highlighting the need for additional, location-specific studies to better understand this relationship.

Given the unique sources of PM and population demographics in different climates, this study aims to conduct a scoping review to summarize existing scientific literature on the association between short-term exposure to $PM_{2.5}$ and PM_{10} and cardiovascular and respiratory morbidity and mortality in global cities in arid and semi-arid regions. By doing so, we intend to address a gap in the literature regarding how PM exposure impacts population health in semi-arid and arid climates, and, particularly, how these impacts differ across sub-populations to identify those most vulnerable. To do this, we identify potential effect modifiers of this association that have been assessed in these regions and perform a meta-analysis to synthesize the existing literature around vulnerable populations in semi-arid and arid regions. Based on these analyses, we provide future directions for informed research to lessen the health burden in such regions.

2. Methods

2.1 Study Region

Arid regions are classified based on an aridity index, defined as the long-term average of the ratio of the mean annual precipitation to mean annual evaporation (arid = 0.05–0.20, and semi-arid = 0.20–0.50) and generally low precipitation (arid <25cm, and semi-arid <50 cm per year; Safriel et al., 2005).

This study uses the Köppen climate classification (Köppen, 1918) to identify regions that are arid or semi-arid, divided into four distinct subcategories. Hot arid climates (BWh) are typically low latitude deserts with average annual temperatures greater than 18˚C (64°F), such as Northern Mexico, parts of the southwestern United States, and Saudi Arabia. Cold arid climates (BWk) are typically found in mid-latitude deserts such as Central Asia or southern South America, with annual average temperatures below 18°C. Semi-arid regions are similarly characterized by hot (BSh) and cold (BSk), and extend from the boundaries of arid deserts into vast grasslands, notably the U.S. Great Plains and Eurasian Steppe (BSk), and in Southern Africa and Australia (BSh).

As city delineations can be arbitrary and many cities exist at a boundary between an arid and semi-arid climate or a semi-arid and continental climate, the most arid characterization was considered when identifying cities that fell within a boundary of two climate types. We sought studies which assessed the relationship of interest at the city scale to extract the estimated health impacts for a population that was likely exposed to more similar PM sources and concentrations as compared to wider geographic areas (e.g., counties, states, countries). To enhance comparability, cities were considered as meeting our criteria if they included semi-arid or arid climates, and larger spatial units were considered if either fully arid, semi-arid, or some combination of the two. For example, studies quantifying the relationship of interest for the country of Kuwait could be considered as the entire country is arid.

2.2 Search Strategy

The goal of this study was to assess the extent of global literature considering short-term PM-related cardiovascular and respiratory health outcomes in arid and semi-arid regions. We identified peer-reviewed studies published in English between 1 January 2012 and 24 April 2024 in two separate searches. The initial search was conducted in April 2024 and updated in September 2024 so as to include any articles published during the course of the review. We followed the Preferred Reporting Items for Systematic Reviews and Meta-Analyses Extension for Scoping Reviews (PRISMA-ScR) checklist to conduct this review (Tricco et al., 2018) with slight changes constructed through our own protocol.

Following the framework to establish effective questions in exposure and health research (PECO; Morgan et al., 2018): our review addresses human populations in semi-arid and arid regions (population) and the effect of short-term exposure to PM (exposure), comparing subpopulation effect modifiers (comparator) on cardiovascular (ICD-10:I00-I99), respiratory (ICD-10: J00-J99), and all-cause (ICD-10: A00-R99) mortality and morbidity (outcome).

The review further aimed to identify empirical epidemiology studies investigating effect modification in such associations between PM exposure and cardiovascular and respiratory mortality and morbidity. The review is composed of two searches, where the first component serves as an update of an existing systematic review published by Bell and colleagues in 2013. The second component is an expanded search based on the Bell et al. (2013) approach, designed to capture a broader range of studies by including more comprehensive keywords and study components explicitly grouped into 5 categories: health outcomes, study type, effect modification, PM exposure, and exposure type. All terms in their final searchable format are reported in Supplement S1. We used these terms to search the National Library of Medicine's MEDLINE database through PubMed (National Library of Medicine).

We followed the same approach for both the initial and updated search. Titles were first manually screened for duplicates, reviews, relevance, and locations outside of our climate classification of interest (Figure 1). Titles and abstracts were then further screened and included based on the following criteria: considered short-term exposure (i.e., same-day or within the last two weeks); PM was the primary exposure; considered cardiovascular or respiratory health outcomes using objectively measured data (i.e., hospital admissions, emergency department visits, mortality counts, etc.); provided an effect estimate on the relationship between PM and such health outcomes; provided this estimate for a specific semi-arid or arid city or area. As such, this meant that studies which provided estimates of the synergistic effect of PM with other variables, such as temperature, or only included pooled multi-city, nationwide, or multi-country estimates (and not city-specific estimates) were excluded. While the inclusion of an estimate of the relationship between PM exposure and cardio-respiratory health outcomes was necessary to be considered in this review, we regarded this criteria met for both explicitly stated estimates and those shown in exposure response curves so as to comprehensively capture research efforts that quantified this relationship, however, estimates could not be extracted (i.e., were only reported in figures). Given the nature of the studies and our inclusion criteria, a full-text review including supplementary material if relevant was required for the majority of studies. The following information was extracted from each study: citation, study location, coordinates, climate classification of the study area, study start and end dates, study population, data source, particulate matter type (i.e., $PM_{2.5}$, PM_{10}), temporal resolution or lag structure considered, health outcomes assessed, study method (i.e., time-series, case-crossover), statistical model employed, effect measure estimates, any confounders considered, effect modifiers analyzed, and estimates associated with the effect modifiers if considered. This data was obtained from tables, text descriptions, and supplemental material from the respective article. The studies which met our criteria were included in the review and synthesized in an overview table. Studies which did not report effect estimates in writing or tables (e.g., only in a figure), could not be included in the meta-analysis.

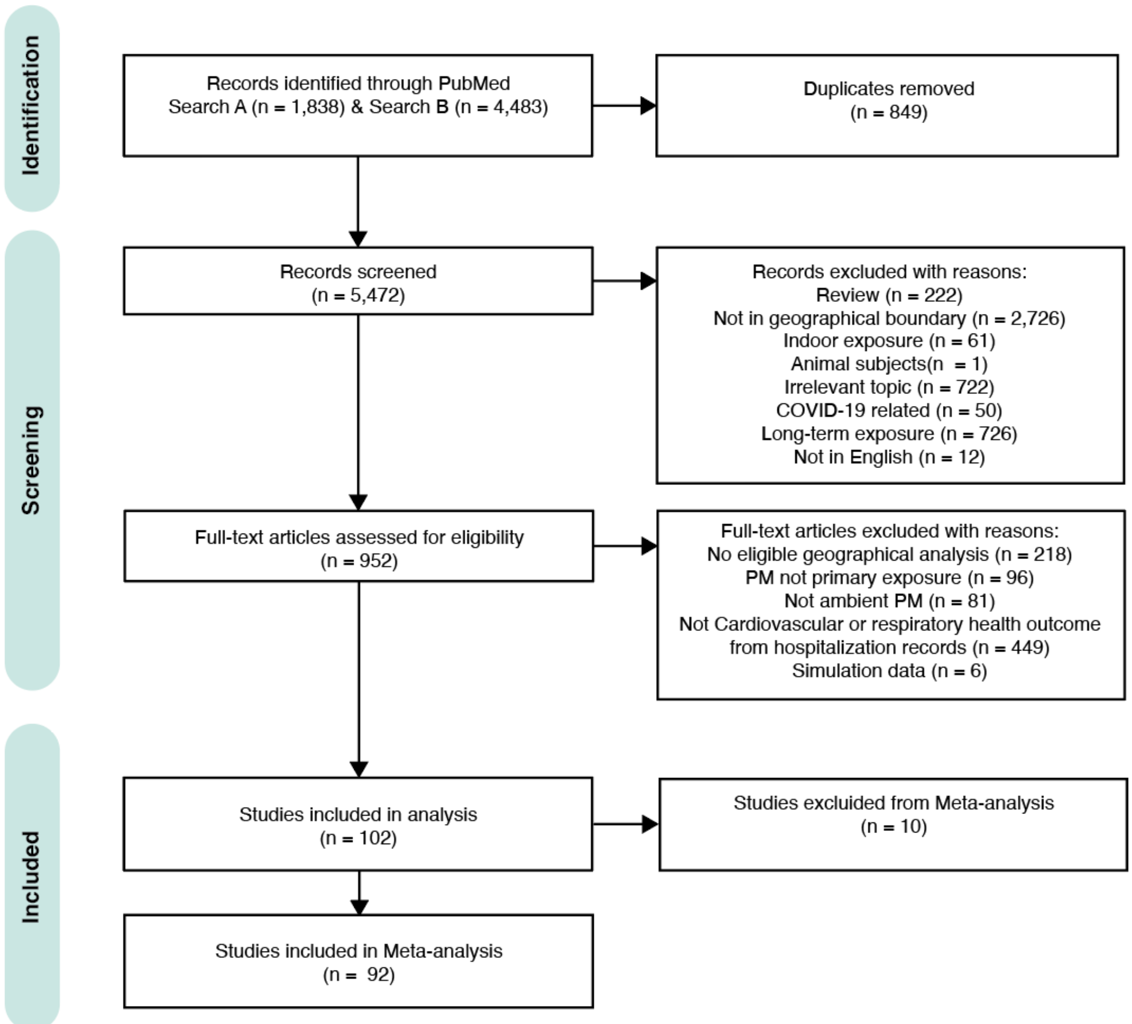

Figure 2.1. Flow diagram detailing the study selection process and reasons for study exclusion.

2.3 Meta-analysis

For the meta-analysis, a secondary table was created to include only studies with reported effect estimates. For each of these studies, we extracted the overall effect estimates, estimates associated with effect measure modifiers (along with all relevant details, such as subpopulation assessed, pollution unit considered, etc.), and the measure of uncertainty calculated for both types of estimates (e.g., confidence intervals). Both time-series and case-crossover studies were included and compared together in the meta-analysis, as estimates of air pollution related mortality and morbidity from both study types have been found to be comparable (Lu and Zeger, 2007; Fung et al. 2003). When risk estimates for multiple lags were available, the shortest (i.e., lag 0, or same day) was extracted to be used in the meta-analysis. While studies reported results using a variety of risk measures (Table S4), all estimates were converted to relative risk estimates (RR) and confidence intervals from 10 μg/m^3 increase in exposure for direct comparison. Estimates from studies that considered specific disease as well as general outcomes were grouped into either cardiovascular (e.g., ischemic stroke ICD-10: I63.*; and general, ICD-10: I00-I99), respiratory (e.g., asthma, ICD-10: J45; and general, ICD-10: J00-J99), or all-cause categories. Additionally, studies that used a range of age categories in the primary analysis or when assessing age as an effect modifier, were grouped into children (ages 0-18), adults (ages 0-65), and older adults (aged 65 and up) to allow for general comparison between studies. Studies reported effect estimates in two categories for sex (male/female) as a biological distinction, while some reported gender (male/female) which is a social distinction. For the purposes of this review, we have decided to use both estimates under the category of sex.

Following previously published methods (Benmarhnia et al., 2015), meta-analyses were considered for categories (climate, outcome, and PM exposure type) and effect modifiers (demographic or temporal) if estimates were available from more than 10 studies. The effect modifiers that were sufficiently numerous for meta-analysis were sex, age, season, climate, and population density. We performed separate meta-analyses for PM_{10} and $PM_{2.5}$, as well as mortality and morbidity for all modifiers if sufficient studies were available. Many studies reported effect estimates for multiple outcomes (e.g., both cardiovascular and respiratory diseases), which were assessed for heterogeneity separately, leading to a greater number of comparators than studies. In the case of stratification by age, we performed heterogeneity assessment for pairings (e.g., children to adults) if there were more than 5 effect estimates. First, a restricted-maximum likelihood (REML) random-effects model and I^2 statistic was used to create a pooled estimate for effect modifier subgroups. A heterogeneity assessment was then performed using the ratio of relative risks (RRR) for subgroups from the same study (Altman & Bland, 2003), calculated for all eligible estimates (e.g., RRMale/RRFemale). The RRR estimate and the I^2 statistic were used to assess the significance, with an I^2 threshold of >50% (Higgins & Thompson, 2002). This determines whether there was in fact statistically significant heterogeneity in the magnitude of effect between subgroups or if the variation is due to sampling error, or otherwise stated: testing if variation exceeds the amount expected under a null hypothesis of no heterogeneity (using 10% as an a priori threshold; Kaufman & MacLehose, 2013). Studies were excluded from heterogeneity assessment if two comparable subgroup estimates were not provided (e.g., both male and female). Further, a Cochran's Q test was performed to assess heterogeneity between population subgroup risk estimates across multiple studies. Equations for all meta-analyses are available in the supplement S1. Publication bias was assessed using both funnel plots and Egger's regression model for each subgroup RRR analysis. All analyses were performed in RStudio using the {meta} package (see Supplement S2).

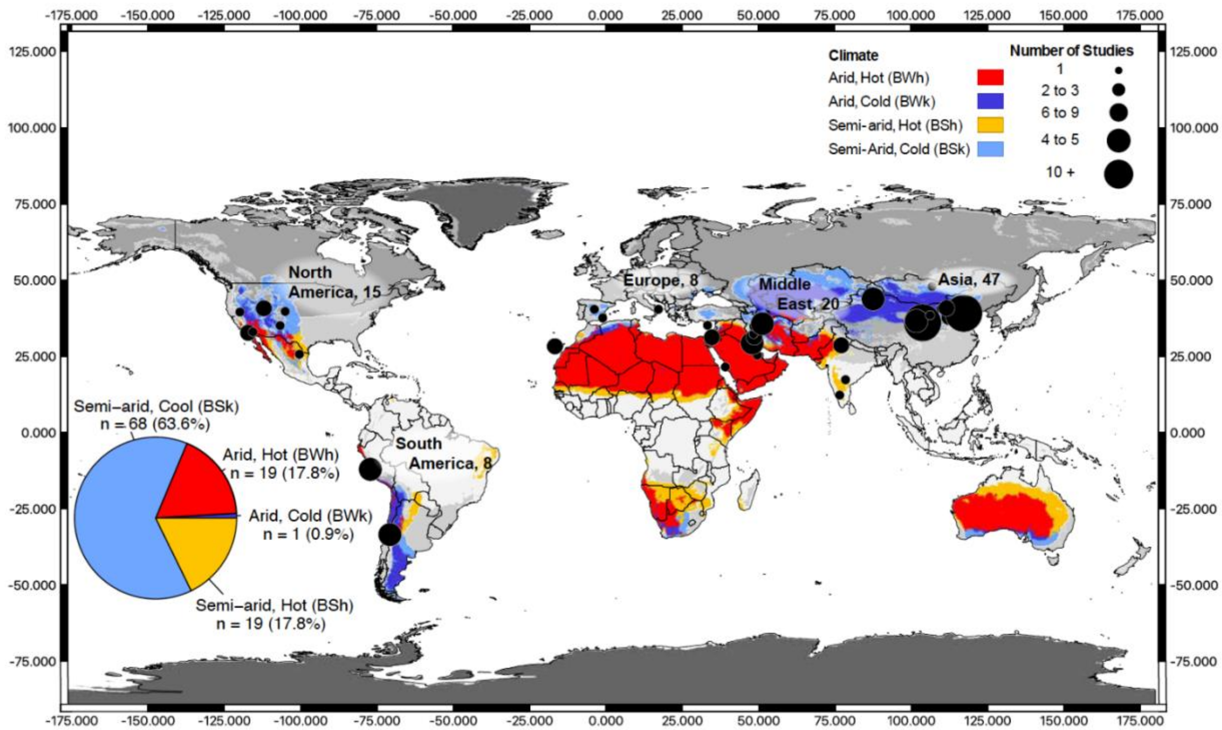

Figure 2.2. Global map, showing the location and number of studies, by city, region, and climate, and the arid and semi-arid climate zones as per the Köppen climate classification.

3. Results

3.1 Study locations and types

Our search identified 5,472 records, and after initial screening for inclusion criteria, 952 articles remained for consideration. After further screening for cardiovascular and respiratory studies and for sufficient geographic analysis, 102 studies were included in this analysis, including one study that was found outside of the search (Bayart et al., 2024). A detailed account of the number of studies along with the reason for exclusion is presented in Figure 1. Of the 102 studies included in this study, 70 reported morbidity and 37 mortality data, with the majority (68%) using time series analysis. Many studies assessed both PM_{10} and $PM_{2.5}$ exposure effects (n=43), while 22 considered only PM_{10}, 36 only $PM_{2.5}$ and one assessed PM of all sizes (total suspended particles; TSP). Age was the most considered modifier (n=67), followed by sex (n=55) and season (n =24) (Supplement S3). Ten studies did not consider any effect modification. Only two studies considered SES, and one poverty. Additionally, five studies explored the impact of dust storms on health, and five studies the effect of wildfire. Respiratory diseases were considered in 54 studies, while cardiovascular diseases were considered in 50 studies. Many studies (n= 28) also presented estimates for all-cause mortality and morbidity (ICD-10: A00-R99), which are also included for comparison. Supplement S5 contains all extracted data and the frequency of study location, study type, outcomes and additional statistics.

The location of studies is geographically inconsistent, with no studies located in Africa or Oceania (Figure 2) despite the presence of arid and semi-arid regions. In comparison, there were 43 studies located in China, 14 in the US southwest, and 11 in Iran. Overall, the most studies were conducted in Asia (n=49), followed by the Middle East (n=22), North America (n=16), South America (n=10), and Europe (n=8). The majority of studies were in the Semi-arid, Cool (BSk) zone (n=68), followed by the Arid, Hot (BWh) zone (n=19), then Semi-arid, Hot (BSh) zone (n=19), and only one study in the Arid, Cool (BWk) climate zone.

Table 2-1. Results of the scoping review, showing the number of studies in several categories.

Scoping Review Results Overview

Summary of Study Characteristics

Category	Count
Overall	
Total Studies	102
Climate	
Semi-arid, Cool (BSk)	82
Semi-arid, Hot (BSh)	21
Arid, Hot (BWh)	19
Arid, Cold (BWk)	1
Outcomes Studied	
Respiratory Disease Mortality	16
Respiratory Disease Morbidity	38
Cardiovascular Disease Mortality	20
Cardiovascular Disease Morbidity	30
All-cause Mortality	25
All-cause Morbidity	3
PM Exposure	
PM2.5	78
PM10	64
Effect Modifiers Considered	
Sex	55
Age	67
Season	24
Study Type	
Case-crossover	24
Time-series	69

3.2 Pooled effect estimates and heterogeneity assessments

This meta-analysis is intended to summarize the evidence of short-term PM exposure and cardiovascular, respiratory, and all-cause morbidity and mortality, and further, the potential effect modification of available variables from all applicable literature. After producing combined RR estimates for the entire population across all studies, we then systematically calculated pooled RR estimates by potential effect modifiers, including climate, sex, age, and population density using a restricted maximum likelihood random effects (REML) model. Using the same approach, heterogeneity between subgroups was explored and, finally, we provide a pooled ratio of relative risk (RRR) estimates. In line with our previously mentioned criteria, results are only reports for the specific combinations of PM exposure and disease categorization where there were sufficient (n=5) studies with usable estimates for assessing pooled relative risk as well as subgroup heterogeneity and its statistical significance regarding effect heterogeneity.

3.2.1 All population estimates

We considered the association of $PM_{2.5}$ and PM_{10} with all-cause, cardiovascular, and respiratory mortality and morbidity. As seen in Figure 3, for $PM_{2.5}$ exposure and morbidity, respiratory diseases had a higher pooled RR of 1.04 (95% CI = 1.04, 1.05) than cardiovascular disease (pooled RR= 1.03, (95% CI = 1.02, 1.05). However, for PM_{10} exposure, risk of cardiovascular morbidity (RR = 1.02, 95% CI = 1.01, 1.03) was greater than that for respiratory morbidity (RR = 1.01, 95% CI = 1.01, 1.01) (Supplement S6). The increase in cardiovascular, respiratory, and all-cause mortality risk related to $PM_{2.5}$ and PM_{10} exposure were extremely similar when considering the entire population (pooled relative risks of 1.01 with varying confidence intervals at or above the null, see Supplement S5 for more details). The only slight deviation occurred in the estimated pooled risk for all-cause mortality in relation to PM_{10} exposure (RR = 1.00, 95% CI = 1.00, 1.01).

3.2.2 Effect modification by sex

Of the 55 studies considering effect measure modification by sex, there are 13 which reported specific effect estimates for $PM_{2.5}$ exposure and 19 for PM_{10} (Supplement S4). For $PM_{2.5}$ exposure morbidity, females had a pooled relative risk of 1.04 (95% CI = 1.03, 1.06) for all respiratory diseases and 1.03 (95% CI = 1.00, 1.07) for all cardiovascular diseases, while males had a risk of 1.05 (95% CI = 1.03, 1.07) and 1.04 (95% CI = 1.00, 1.08) for the same, respectively (Figure 3). The pooled morbidity risk from PM_{10} exposure for women (respiratory RR = 1.01, 95% CI = 1.00, 1.02; cardiovascular RR = 1.01, 95% CI = 0.99, 1.03) and for men (respiratory RR= 1.01, 95% CI = 1.01, 1.02; cardiovascular = 1.02, 95% CI = 1.00, 1.05) is lower than that from PM2.5 exposure (Supplement S5). While the estimated associations between short-term exposures to PM2.5 and morbidity were generally slightly higher (and marginally statistically significant) for men compared to women, heterogeneity analyses found no statistically significant difference in respiratory (RRR= 1.00, I^2 = 33%) or cardiovascular morbidity risk (RRR= 1.00, I^2 = 0%) by sex (Table 2). PM_{10} exposure estimates were only sufficient (n=6) for all-cause mortality analysis, suggesting a 3% lower risk for females (RR = 0.97, 95% CI = 0.95-0.99) and a 4% lower risk for males (RR = 0.96, 95% CI = 0.94-0.97) (Supplement S6).

3.2.3 Effect modification by age

Effect estimates for three age categories (children (0-18), adults (18-65) and older adults (65+) were compared for all respiratory morbidity outcomes (Figure 3). From $PM_{2.5}$ exposure, respiratory morbidity risk was highest for adults, with a pooled RR 1.09 (95% CI = 1.05, 1.13), followed by children 1.06 (95% CI = 1.04, 1.08), while risk for older adults was null and insignificant (RR = 1.00, 95% CI = 0.96, 1.03). The difference in risk between adults and older adults is statistically significant with a RRR of 1.06 (I^2 = 89%). Regarding cardiovascular morbidity from $PM_{2.5}$ exposure, the adult pool RR was 1.05 (95% CI = 0.95, 1.06), whereas the older adult risk was 1.03 (95% CI = 0.99, 1.06). The ratio of relative risks is also statistically significant (RRR= 1.02) between adults and older adults for cardiovascular morbidity from $PM_{2.5}$ exposure.

3.2.4 Effect modification by climate zone

Effect modification was assessed for the three climate zones with greater than 10 studies: semi-arid, cool (BSk, n=67); semi-arid, hot (BSh, n=18); and arid, hot (BWh, n=13). Cardiovascular and respiratory morbidity risk from $PM_{2.5}$ exposure was highest in populations living in arid regions (RR =

1.05, 95% CI = 1.01, 1.09 and RR = 1.07, 95% CI = 1.05, 1.08, respectively). The risk of cardiovascular morbidity increased by 3% from $PM_{2.5}$ exposure in cold semi-arid regions (RR = 1.03, 95% CI =1.02, 1.04), however, in semi-arid, hot regions, the risk of cardiovascular morbidity increased by an insignificant 1% (RR = 1.01, 95% CI = 0.95, 1.08). Conversely, the risk of respiratory morbidity increased by 4% for populations exposed to $PM_{2.5}$ in semi-arid, hot regions (RR = 1.04, 95% CI =1.03, 1.05), while semi-arid, cool climates experienced a 2% insignificant increased risk of respiratory diseases (RR = 1.02, 95% CI =0.99, 1.06).

3.2.5 Effect modification by population density

We considered the effect of population density on mortality and morbidity risks by grouping studies into two categories: high (>4,500 people/mile) and low (<4,500 people/mile) density cities. For cardiovascular outcomes in relation to PM2.5 exposure, there was a 5% increased risk of morbidity (RR = 1.05, 95% CI = 1.02, 1.07) and a 2% increased risk of mortality (RR = 1.02, 95% CI = 1.01, 1.02) in high density cities compared to the observed 2% and 1% greater risk of disease (RR = 1.02, 95% CI = 1.01, 1.02) and mortality (RR = 1.01, 95% CI = 1.00, 1.02) in low density cities, respectively. Respiratory morbidity risk from PM2.5 exposure was also slightly greater in high density cities (RR = 1.05, 95% CI =1.03, 1.07) compared to low density cities (RR = 1.04, 95% CI = 1.03, 1.05). There was no observed increased or decreased risk of all-cause mortality from $PM_{2.5}$ exposure in high density cities (RR = 1.00, 95% CI = 1.00, 1.01), while a marginal increase in risk was observed for low density cities (RR = 1.01, 95% CI = 1.00, 1.01).

Considering PM_{10} exposure, both cardiovascular morbidity (RR = 1.02, 95% CI = 1.01, 1.04) and mortality (RR = 1.02, 95% CI = 1.00, 1.07) risk increased by 2% in low density cities compared to a 1% increased risk for morbidity (RR = 1.01, 95% CI = 0.99, 1.04) and mortality (RR = 1.01, 95% CI = 1.01, 1.02) in high density cities. However, it should be noted that the increase of 1% in cardiovascular morbidity risk from exposure to PM10 is non-significant in high-density cities. The small increase in respiratory morbidity risk was similar across low-density (RR = 1.01, 95% CI = 1.01, 1.01) and high-density cities (RR = 1.01, 95% CI = 1.00, 1.01). The estimate related to PM10 exposure and all-cause and respiratory mortality in high-density cities was 1.01 (95% CI = 1.00, 1.01) which was greater than the low-density risk of 1.00 (95% CI = 1.00, 1.00). If further stratified by climate, there are more significant differences. For example, in high-density cities located in arid, hot (BWh) regions there was a significantly higher risk estimates for respiratory morbidity of 1.09 (95% CI = 1.07, 1.11) compared to 1.02 (95% CI = 1.01, 1.02) observed in low-density, arid cities. In semi-arid, cool (BSk) climates the relative difference is inverted, with low-density cities having a pooled risk of 1.04 (95% CI = 1.03, 1.05) and high-density cities with a risk of 1.00 (95% CI = 1.00, 1.00).

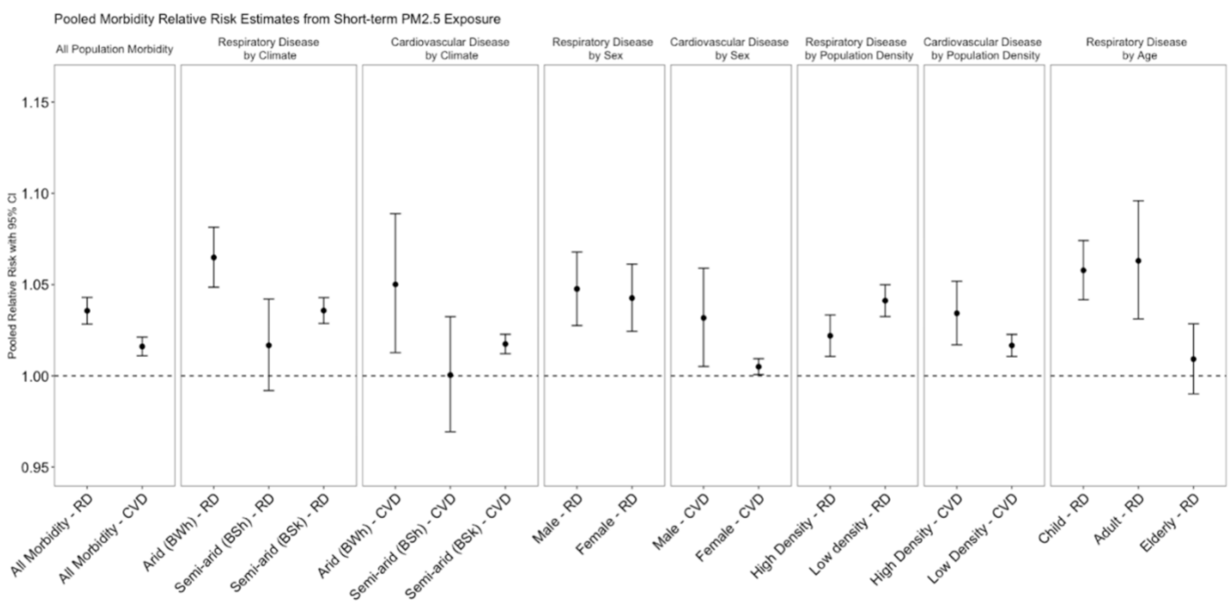

Figure 2.3**.** Pooled morbidity risk ratios from 10μg/m3 increase in PM2.5 exposure from included studies.

3.2.6 Effect modification by seasonality

Studies generally explored seasonal variation on effect modification by reporting risks from the “cold” and “warm” seasons, corresponding with local climates. From available relative risk estimates, the pooled respiratory morbidity risk from $PM_{2.5}$ exposure in the cold season was 1.05 (95% CI = 1.03, 1.08), in comparison to the warm season, which was 1.04 (95% CI = 1.01, 1.07). Heterogeneity analysis of these risks finds the cold season has a 2% increased risk (RRR = 1.02, I^2= 74%). Pooled respiratory morbidity risk from PM_{10} exposure was not significant, with a risk of 1.01 (95% CI = 1.00, 1.03) in the cold season, and 1.00 (95% CI = 1.00, 1.01) in the warm season. Similarly, PM10 exposure was found to have an insignificant 1.00 (95% CI = 1.00, 1.01) pooled relative risk for cardiovascular morbidity during both periods.

3.2.7 Event specific and effect modification by compounded risks

Multiple studies focused on the potential of PM concentration and composition modulating events to impact PM exposure-related health outcomes, specifically dust storms and wildfire related smoke. Dust storms were considered in eight studies, one in China (Ma et al., 2016); Spain (Díaz et al., 2012); the Canary Islands (Lopez-Villarrubia et al., 2021); and five in the Middle East (Al-Taiar et al., 2014; Aghababaeian et al., 2021; Shahsavani et al., 2020; Vodonos et al., 2015; Neophytou et al., 2013). In summary, the findings include: an increased number of ER visits for respiratory diseases during dust storms (Ma et al., 2016); increased daily mortality in one city, Ahvaz, but decreased in another, Tehran (Shahsavani et al., 2020); limited evidence for increased cardiovascular mortality during dust days (Neophytou et al., 2013); seasonally variable risk for cause specific mortality increase during dust days, though not statistically significant (Díaz et al., 2012); increased emergency department visits during dust storms, especially low altitude and long duration (>5 day) storms (Lopez-Villarrubia et al., 2021); statistically significant excess risk of mortality (Aghababaeian et al., 2021); increase in myocardial infarction hospitalization, but only at lag >2 (Vodonos et al., 2015); no significant association between dust storm events and same-day respiratory mortality (Al-Taiar et al., 2014).

Increased morbidity and mortality from wildfire were the focus of eight studies, six from the USA (Do et al., 2012; Kiser et al., 2020; Resnick et al., 2015; Leibel et al., 2020; Horne et al., 2024; Hutchinson et al., 2018), one from Europe (Faustini et al., 2015), and one global study, of which Kuwait was the only applicable location (Chen et al., 2021). Findings included; increased risk of mortality (Chen et al., 2021); elevated respiratory diagnoses during wildfire (Hutchinson et al., 2018); increased asthma hospitalization during the wildfire season (Horne et al., 2024); increased emergency visits for asthma when smoke was present (Kiser et al., 2020); increased risk of ED visits for respiratory and cardiovascular disease, with the older adults especially susceptible (Resnick et al., 2015); increased cardiovascular mortality on smoky days (Faustini et al., 2015), and an increase in pediatric respiratory visits (Leibel et al., 2020). Overall, studies consistently found an increase in the studied health outcomes during wildfire events in comparison to non-wildfire periods.

Table 2-2. Summary of meta-analysis of heterogeneity. Effect modification by age, sex and season are displayed for respiratory and cardiovascular morbidity from $PM_{2.5}$ exposure.

Meta-analysis Summary for Respiratory and Cardiovascular Morbidity from PM2.5 Exposure

Ratio of Relative Risk (RRR) estimates for Morbidity Outcomes

Meta-analysis	# of Studies	**RRR Estimates**			**Heterogeneity Statistics**					
		REML Estimate	Lower CI	Upper CI	I²	H Statistic	Tau	P-value	Cochran's Q	Z
Respiratory										
Season (RR_Cold / RR_Warm)	7	1.02	0.99	1.05	74.1%	1.96	0.02	0.00	23.15	1.88
Sex (RR_Female / RR_Male)	13	1.00	1.00	1.00	7.0%	1.04	0.00	0.38	12.90	−0.14
Age (RR_Adult/ RR_Older Adult)	8	1.06	0.97	1.15	88.7%	2.97	0.08	0.00	61.68	1.52
Age (RR_Child / RR_Adult)	11	0.98	0.95	1.00	85.8%	2.65	0.04	0.00	70.33	−1.92
Age (RR_Child / Older RR_Adult)	8	1.00	0.98	1.02	75.0%	2.00	0.01	0.00	27.95	0.15
Cardiovascular										
Sex (RR_Female / RR_Male)	8	1.00	0.99	1.01	4.8%	1.02	0.01	0.39	7.35	−0.98
Age (RR_Adult/ RR_Older Adult)	6	1.02	0.98	1.06	0.0%	1.00	0.00	0.78	2.48	1.07

Abbreviations: RR = Relative Risk Estimate; REML = Restricted Maximum Likelihood; CI = 95% Confidence Interval

3.3 Chemical constituent and source

Five studies (Joshi et al., 2022; Colonna et al., 2023; Basagaña et al., 2015; Kim et al., 2012; Lu et al., 2019) considered the effect of heterogeneous PM composition on health risk. Three of these studies considered chemical components (Joshi et al., 2022; Basagaña et al. 2015; Lu et al., 2019) while two studies considered source apportionment PM by positive matrix factorization (PMF, Colonna et al., 2023; Berger et al., 2018). Considering components, in India, Joshi et al. (2022) found nitrate, ammonium, chromium, and both elemental and organic carbon showed a higher mortality impact than the total PM2.5 mass, with the above 65 age group most vulnerable. In Denver, USA Kim et al. (2012) found elemental and organic carbon generally were associated with a higher immediate risk for cardiovascular and respiratory diseases, with other species such as sulfate and nitrate being associated with higher risk for longer lags. In Pakistan, Lu et al. (2019) found the relationship to exposure to nickel in PM2.5 (5–14% increase per interquartile range) and cardiovascular hospital admissions was most significant, which was thought to come from fossil-fuels combustion and industrial emissions, and to a lesser extent elements aluminum, iron, and titanium which were thought to come from crustal sources, indicating anthropogenic produced PM was more harmful. Considering PM sources, dust, as opposed to traffic and regional sources, had the higher respiratory disease hospitalization risk in Kuwait (Colonna et al., 2023). Ostro et al. (2016) found vehicular emissions uniquely associated with all cardiovascular emergency department visits, while vehicular emissions, biomass burning, and soil sources were associated with all respiratory emergency department visits, asthma especially, although the soil source (including road dust) had the highest risk estimate for asthma.

4. Discussion

4.1. Summary of main findings

This scoping review investigated literature on the cardiovascular and respiratory health outcomes related to PM exposure in semi-arid and arid regions across the world, using a meta-analysis to produce pooled risk ratios analysis of heterogeneity between effect modifiers. We found weak evidence of increased risk for males versus females, and for adults vs older adults. PM2.5 exposure in hot arid regions (BWh) was found to have higher risks for both cardiovascular and respiratory disease compared to semi-arid regions. Findings were mixed and inconsistent between high- and low-density cities across health outcomes and PM exposure types. Our review also partially serves as an update to the literature regarding short term PM exposure and differential susceptibility and vulnerability among populations, for which the most recent work was published in 2013.

The geographic focus of studies is strongly focused on China, the US, and Iran. Relatively high numbers of studies focused on PM exposure in China and the US have been identified in relevant previous studies (Bell et al., 2013; Yang et al., 2019; Yee et al., 2021), however never before in regards to arid regions. Of note is the complete lack of studies spanning the entire continent of Africa and the country and continent of Australia, despite the fact that these two continents combined contain more than half of global arid and semi-arid regions (Prăvălie, 2016). In the case of Australia, 87% of the population lives 50 km from the coast (Australian Bureau of Statistics, 2020), which is mostly not arid. However, this is not the case in Africa, with some of the continents' largest cities such as Cairo (Egypt), Luanda (Angola), Khartoum (Sudan), and Dhakar (Senegal) are characterized by an arid or semi-arid climate. Due to the criteria of this scoping review, the absence of studies in Africa may be partially due to data availability in many low and middle income countries, which is primarily in the form of surveyed health outcomes (such as the Demographic and Health Survey data) or at the national level as opposed to objectively measured hospitalization or mortality data (Zhao et al., 2021a: WHO, 2021b). This corroborates previous assessments finding a dearth of studies in low- and middle-income countries, and an over emphasis in high income countries where PM exposure is comparably lower (Lim et al, 2022). We conclude there is a gap in the representation of arid region low income and middle-income countries in short-term exposure related health outcome studies.

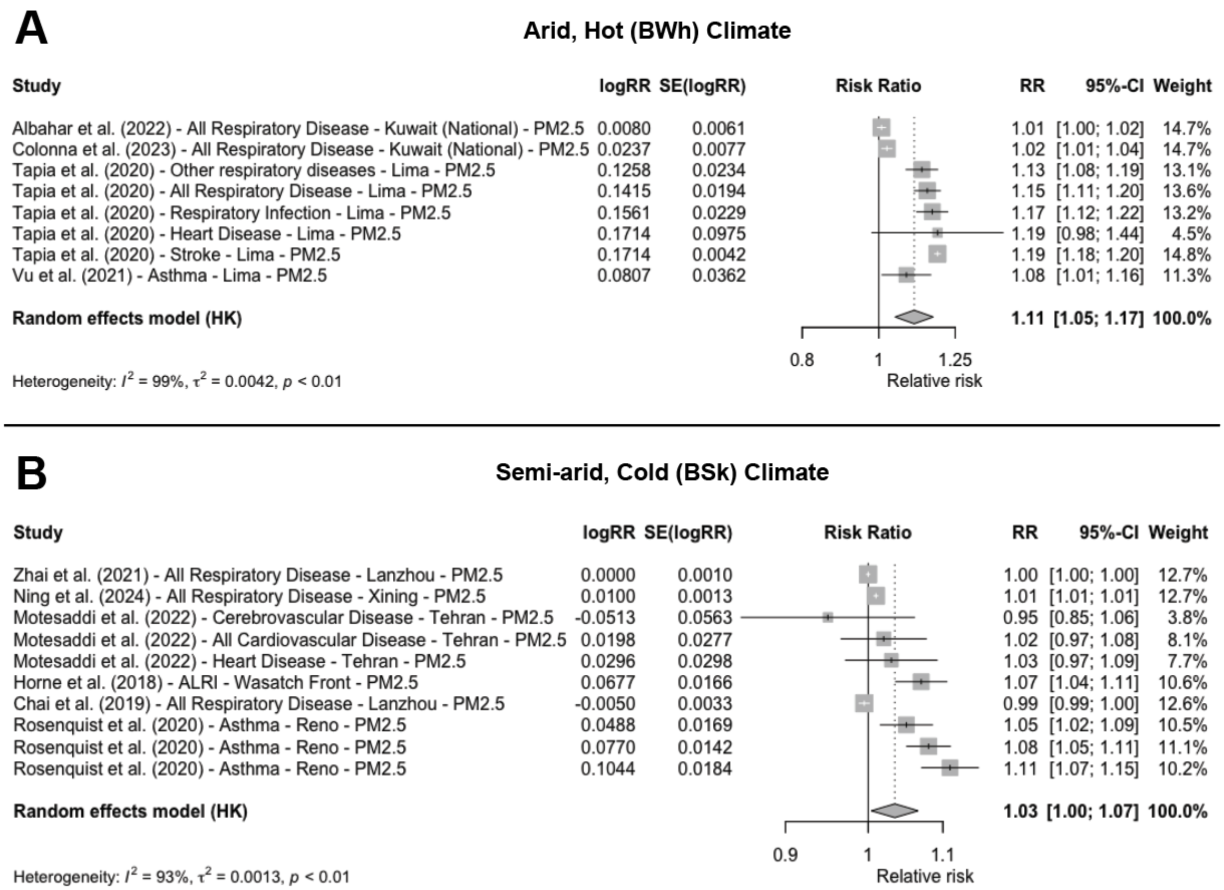

Figure 2.4. Comparison forest plots for cardiovascular and respiratory outcomes from PM2.5 exposure **A)** arid, hot (BWh) regions, and **B)** in semi-arid, cold regions. The pooled relative risk (RR) uses a restricted maximum likelihood random effects model.

4.2. Climate regions

Our meta-analysis finds the highest respiratory and cardiovascular morbidity risk from exposure to $PM_{2.5}$ in the most arid (BWk) regions relative to semi-arid regions (Figure 4). While there is an absence of literature specifically assessing differential health impacts between arid and semi-arid regions, many studies have investigated the variation in outcomes between multiple cities, in both arid and non-arid climate regions (Franchini & Mannucci, 2007; Yin et al., 2017; Liu et al., 2022c; Odo et al., 2022) with no systematic findings that semi-arid or arid cities have a higher total or disease specific mortality risk than non-arid cities. However, several studies we reviewed report higher risks related to PM exposure during specific events (i.e., wildfires and dust storms) or the synergistic effects with other environmental exposures (i.e., PM and heat) pointing towards the need for additional research climate-informed PM sources and risk. The results of this study show a slightly higher risk for cardiovascular and respiratory morbidity and mortality from $PM_{2.5}$ exposure in arid climates compared to both hot and cold semi-arid climates, which could be due to sociodemographic variables (e.g., access to air conditioning, workplace norms and location, acclimatization), compounding environmental exposures (e.g., humid heat and PM), or climate specific differences in PM source type and climate dynamics (e.g., dust storms). For example, populations in hotter, drier regions could experience enhanced dehydration or the presence of humid heat in semi-arid regions, which have more precipitation, can hamper the body's response to heat, both increasing strain on the cardiovascular system and the risk of adverse events in the presence of PM (Ebi et al., 2021). Further, the majority of studies focused on outcomes in cold, semi-arid regions, leading to an unequal distribution of data across climate types and temperature ranges and extremes.

Considering that half a billion people live in areas that are projected to become drylands in the 21st century in a 4˚C global warming levels (GWLs) scenario (Spinoni et al. 2021) the authors of this study feel there is significant need to better constrain the impact of climate on PM exposure and health outcomes. To our knowledge, prior to this study there has only been one previous review that considered climate specific short-term PM exposure effect estimates, which focused on cold climates (Wine et al., 2022). Wine et al. (2022) highlight that unique mechanisms of exposure due to temperature and other climate variables are poorly understood. Temperature alone has been found to exacerbate respiratory morbidity risk (Areal et al., 2022) though the synergistic relationship with PM exposure is poorly understood. This is especially of concern in arid and semi-arid regions, some of which are projected to surpass human tolerability in the 21st century (Raymond et al., 2020), putting vulnerable populations at an even higher risk in the context of climate change. In particular, the impact of climate and human development related destabilization of land surfaces in arid and semi-arid regions leading to more PM emission from various sources is an understudied topic (Safriel et al., 2005).

4.3 Overview of meta-analysis and effect modification

Our meta-analysis found a slight increase in the pooled relative risk of respiratory and cardiovascular morbidity and mortality across all populations included in our scoping review (Figure 3, Supplement S4-S5), with the greatest increase in risk for respiratory (4%; 4-5%) and cardiovascular disease (3%; 2-5%) related to $PM_{2.5}$ exposure. We also explored a disease specific sensitivity analysis. Considering specific respiratory diseases and $PM_{2.5}$ exposure, pneumonia (RR = 1.10, 95% CI = 1.06, 1.14), bronchiolitis and bronchitis (RR = 1.09, 95% CI = 1.06, 1.12), asthma (RR = 1.05, 95% CI = 1.03, 1.07), and acute lower respiratory tract infection (ALRI) (RR = 1.07, 95% CI = 1.05, 1.08) had considerably higher risk compared to estimates reporting all respiratory morbidity diseases combined (RR = 1.02, 95% CI = 1.01, 1.02). For cardiovascular disease, all stroke morbidity (RR = 1.05, 95% CI = 1.00, 1.09), acute coronary syndrome (ACS, RR = 1.05, 95% CI = 1.03, 1.07), and heart failure (RR = 1.02, 95% CI = 1.02, 1.03) risk was higher than all-cardiovascular disease (ICD-10: I00-I99, RR =1.01, 95% CI = 1.00-1.02), with only heart disease having a lower estimate (RR =1.00, 95% CI = 1.00, 1.00).

Four effect modifiers were included in multiple reviewed studies (age, sex, season, population density) and analyzed in our meta-analysis. Both season (cold versus warm) and population density (low versus high) had relatively inconsistent and weak associations across all studied health outcomes and PM exposures. While sociodemographic characteristics, such as those with lower socioeconomic status and in minority populations have been shown to increase the risk for adverse health outcomes in relation to PM exposure (Bell et al., 2013; Ma, et al., 2023), only one identified study considered SES and four considered race/ethnicity. This may be due to our study inclusion criteria, as SES and race/ethnicity are often studied at a national level (e.g., Ma, et al., 2023; Jbaily, et al., 2022), and thus, these publications were not included based on our criteria, despite the importance and need to assess differential health impacts across such sub-populations.

Regarding age–which was the most studied effect modifier–we observed a 9% increase in the risk of respiratory morbidity as a result of $PM_{2.5}$ exposure among the adult population (those aged 18 to 64). Children (0 to 18 years) experienced an increase in risk of 6% (95% CI: 4-8%), whereas older adults (ages 65 and up) experienced no increased or decreased risk in these regions. Though the finding is limited by a small number of studies (n=8), the heterogeneity in risk between adults and older adults (RRR = 1.06%) is significant. This is at odds with previous findings (Bell et al., 2013), which have found that older adults have a statistically higher cardiovascular risk from PM exposure. The inconsistency in our results could be potentially due to sampling bias, different population composition, competing risks or differences in activities across populations, such as an increase in workplace exposure for adults in arid and semi-arid regions.

As for effect modification by sex, pooled relative risk estimates indicate a general trend of higher risks for males compared to females, but we did not find a statistically significant difference when applying heterogeneity tests (e.g., $PM_{2.5}$ exposure respiratory RRR= 1.00, I^2= 7%). As more thoroughly addressed elsewhere (Clougherty, 2009), discussion of a binary sex-based effect modification is imprecise as it may be influenced by biological factors (e.g., lung size or hormonal influences on chemical transport) or societal differences in lifestyle (e.g., gendered occupations or behaviors such as smoking), leading to vastly different conclusions. Therefore, to investigate heterogeneity in risk based on differential exposure, or separately, differential susceptibility, by different genders or sexes, more specific geographically and socially qualified data is needed to reveal trends that may support activities to mitigate risks.

4.4. Unique sources of PM

Multiple studies note the lack of information concerning the chemical composition of PM a weakness in results (e.g., Vodonos et al., 2015, Al-Taiar et al., 2014). While the composition of PM was only the focus of five studies, all found that some sources had greater association with health outcomes than others (see section 3.3). Previous literature reviews concerning source-specific PM exposure on long-term (Wyzga & Rohr, 2015) and short-term (Xu et., 2022) health outcomes found this focus a growing trend in global studies, with significant potential to address uncertainties regarding the most harmful PM and associated variability in health outcomes. While the identification of PM sources using chemical composition is a well-established field of research (Colucci & Begeman, 1965; Karagulian et al., 2015), its application to qualifying PM exposure risk has recently been gaining more attention as applied to pediatric health outcomes (Huang et al., 2019) and regarding cause specific mortality (Berger et al., 2018). In some regions using chemical compositions to study PM source and associated health impacts is more accessible through the availability of national speciated PM monitoring networks in the US (Chemical Speciation Network; US EPA 2023) and China (National Aerosol Composition Monitoring Network (NACMON); Dao et al., 2019).

A related research trend concerns health risk from exposure to PM from unique events, such as dust storms and wildfires. Thus far, there is inconclusive evidence that dust storms are associated with increased cardiovascular and respiratory health risks (see section 3.2.6). The PM generated by dust storms may be less toxic than anthropogenic PM (Vodonos et al., 2015), although this conclusion is highly variable by location and possibly influenced by decreased healthcare utilization during dust storms. City size, frequency of storms, and composition of dust may influence PM health effects (Shahsavani et al., 2020), which we were unable to study in this review. Dust storms—which are associated with increased mortality (Zhang et al., 2016)—are an issue in arid regions, downwind environments, major urban centers (Goudie 2014), as well as in non-arid regions such as Korea (Lee et al., 2013) and Europe (Khaniabadi et al., 2017). All the studies reviewed that considered wildfires found increased relative risk during wildfire exposure, or increased risk in comparison to non-wildfire periods (see section 3.2.6). Wildfire smoke particles have been identified as being uniquely harmful for respiratory health compared to other PM (Aguilera et al. 2021), with synergistic adverse health effects with heat (Chen et al., 2023) which is a concern as compound exposures are increasingly occurring in many regions due to climate change. While wildfire and dust storm exposures are not unique to arid regions–though the production of dust storms is a unique phenomenon in drylands–the unique interaction between these events and regional climates is a subject requiring greater attention. Climate change is expected to intensify extreme temperatures, and increase the frequency of droughts, wildfires, dust storms, flooding, storm surges, and hurricanes (Khraishah et al., 2022). Given the potential for compounding and exacerbated climate-driven hazards, it is critical to understand current and future impacts of climate change on human health in different regions.

4.5 Future Directions

Air pollution does not abide by political borders (Schwarz et al., 2023). It is capable of traveling long distances from the primary emission source, and often disproportionately impacts those who are not responsible for the majority of emissions (Tessum, et al., 2019). In review of the current literature, we identified three main gaps and potential focus areas for future research to elucidate the differential relationships that exist between PM exposure and health: rural populations, comparisons between climate types, and source specific PM risk estimates. While the vast majority of PM exposure-related health studies are in dense urban environments, there is a need to understand this relationship in rural areas as the sources and composition of PM are different, as is the population exposed. For example, the impact of dust on human health may be exacerbated in cropland regions, areas which are understudied if not entirely left out of the literature (Guan et al. 2016). Beyond more research on the constituents and sources of PM, there is a dire need for epidemiological studies quantifying the specific relationship between health outcomes and populations at a local scale. Specifically, this is needed in understudied arid and semi-arid regions, but also for unique populations with sociodemographic characteristics that dictate their exposure and susceptibility to negative health impacts of PM. Further, such research is crucial to project future health impacts due to increased desertification and heat from climate change and design effective policies and interventions to address this.

4.6 Limitations

While this work contributes to a gap in the literature on health impacts from PM exposure in arid and semi-arid regions and how that relationship differs across sub-populations and climate, there are some limitations to this study that must be noted. Due to the exclusion criteria of our scoping review, there were studies assessing the relationship of interest that were not written in English or using subjectively measured health outcomes (e.g., surveys) that were not eligible. Further, many national or global studies exist with robust data that analyze arid region PM health risks, however as they only reported pooled estimates, we were unable to utilize their findings in our review. As a result, estimates of the relationship of interest exist for regions and populations not represented in our findings. These criteria were chosen to make the research feasible for the authors to complete and to enable direct comparisons between study findings but may have led to the exclusion of relevant studies in some regions, notably South America, Central Asia, the Middle East, and both sub-Saharan and Northern Africa, leading to incomplete or skewed conclusions. Additionally, the comparison of estimates from different regions, cities, and years within the 12-year period without equivalent studies in comparator regions may have led to inaccurate conclusions regarding the effect of geography on health risks. Another potential source of bias is the unequal number of studies in each subgroup comparison, which ideally would be balanced to ensure the statistical methods used yield robust and unbiased estimates. Lastly, we were unable to fully capture all adverse health outcomes related to PM exposure as our focus was solely on cardiovascular and respiratory related outcomes.

5. Conclusions

Through this scoping review, we identified 102 studies across four continents that quantified the relationship between exposure to PM and cardiovascular and respiratory morbidity and mortality in semi-arid and arid regions of the world. While the number of studies published per year generally increased over this interval, we identified significant gaps in both geographical representativity and quantitative estimation of the relationship between climate and health risks from PM exposure, especially concerning vulnerable subpopulations (e.g., migrants, lower socioeconomic status, and minority groups). The link between PM and morbidity and mortality is well established, although understanding concerning the direct relationship between specific sources of PM and health outcomes is underexplored. Through our meta-analysis of reported effect estimates, we found that across populations, exposure to $PM_{2.5}$ and PM_{10} resulted in increased risk of respiratory and cardiovascular morbidity and mortality. There was weak evidence of effect measure modification across subgroups, including sex, age, seasonality and aridity. Notably, pooled relative risk for cardiovascular and respiratory morbidity was greater in arid regions in comparison to semi-arid regions. Although limited, our findings highlight the existence of variation across populations and places, and the need for additional measurement of this relationship, especially in areas with limited air quality monitoring networks or a lack of existence in the literature. Given the co-occurrence of climate change induced environmental extremes and continued anthropogenic PM emissions, we urge future work to consider compounding health risks and climate synergies across diverse populations.

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Chapter 2 Appendix

Epidemiological Search after Bell et al., 2013 (PubMed)

Search A: ("effect modification" OR "effect modifier" OR "effect modifiers" OR "effect" AND "modifiers" OR "effect" AND "modifying") AND ("time series" OR "case crossover" OR "air pollution" OR "air pollutant" OR "air pollutants" OR "PM" OR "PM10" OR "PM2.5" OR "particles" OR "particulate matter")

Search B: ("modified" OR "modification" OR "modify" OR "modifying") AND ("effect" OR "effects") AND ("PM10" OR "PM2.5")

Search C: ("emergency department" OR "emergency visits" OR "emergency room" OR "hospital" OR "hospitals" OR "hospitalizations" OR "mortality") AND ("time series" OR "case crossover") AND "short term" AND ("PM" OR "PM10" OR "PM2.5" OR "particulate matter" OR "particles")

Epidemiological Search B (PubMed)

("Health outcome*" OR "public health" OR "health impact*" OR "health effect*" OR morbidit* OR mortalit* OR hospital* OR "hospital admission*" OR "emergency" OR "emergency department" OR "emergency visits" OR "emergency room" OR "emergency department visits")

AND (Cohort OR "case-crossover" OR "case crossover" OR "cross-sectional" OR "cross sectional" OR "time-series" OR "time series" OR “ecologic*”)

AND ("Differential exposure" OR susceptib* OR vulnerabil* OR inequality OR heterogene* OR subgroup OR sociodemographic OR "socio-demographic" OR demographic OR socioeconomic OR "socio-economic" OR age OR sex OR gender OR race* OR ethnicit* OR income OR education OR urbanicity OR pregnan* OR "access to care" OR "legal status" OR migra*)

AND (particle* OR particulate* OR "desert dust" OR "particulate matter" OR "particulate pollution" OR PM1 OR PM2.5 OR "PM 2.5" OR PM10 OR "total suspended particles" OR TSP OR "ultrafine particle*" OR UFP OR "fine particulate matter" OR "oxidative potential")

AND (((Temporal OR seasonal OR Annual) AND variation) OR ambient OR "short term" OR "short-term" OR (("particulate" OR "air pollution") AND (exposure OR "dose response*" OR "exposure response function" OR "exposure-response function")))

Supplemental Figure 2.1. Search Terms

Ratio of Relative Risk (RRR) calculation, after Altman & Bland, 2003

$$SE = \frac{UCI - LCI}{3.92}$$

$$\mathrm{d} = \ln\,(Estimate_{GroupA}) - \ln\,(Estimate_{GroupB})$$

$$\mathrm{SE} = \frac{\ln\,(UCI_{GroupA}) - \ln\,(LCI_{GroupB})}{2 \times 1.96}$$

$$\mathrm{SE(d)} = \sqrt{{SE_{groupA}}^2 + {SE_{GroupB}}^2}$$

$$\mathrm{UCI(d)} = d + 1.96 \times SE(d)$$

$$\mathrm{LCI(d)} = d - 1.96 \times SE(d)$$

$$\mathrm{RRR} = \exp(d)$$

$$\mathrm{UCI(RRR)} = \exp\big(UCI(d)\big),\ \mathrm{LCI(RRR)} = \exp\big(LCI(d)\big)$$

Cochran’s Q calculation, after Kaufman & MacLehose, 2013

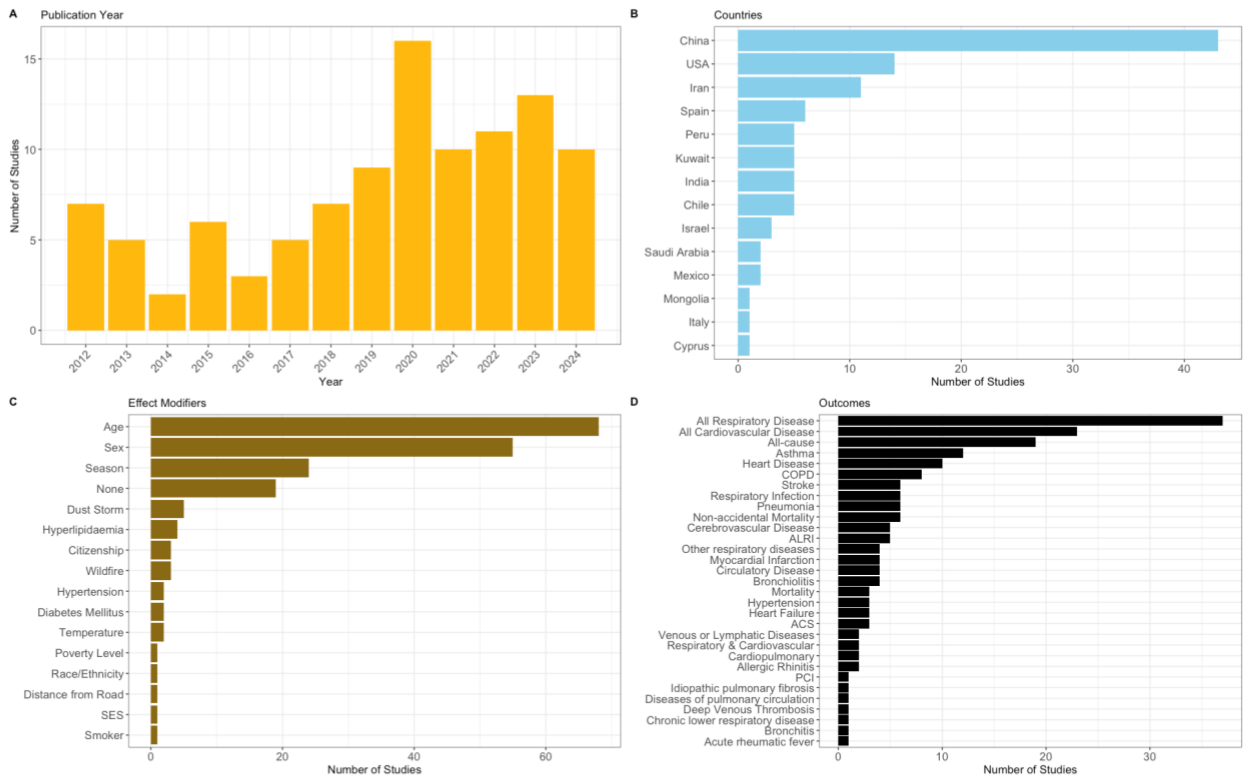
$$\text{ß} = \ln(Estimate)\,, \qquad SE = \frac{UCI - LCI}{3.92}, \qquad VAR = \mathrm{SE}^2$$

$$pooled\ \text{ß}\ or\ \text{ß}_p = \frac{\frac{\text{ß}_{GroupA}}{VAR_{GroupA}} + \frac{\text{ß}_{GroupB}}{VAR_{GroupB}}}{\frac{1}{VAR_{GroupA}} + \frac{1}{VAR_{GroupB}}}$$

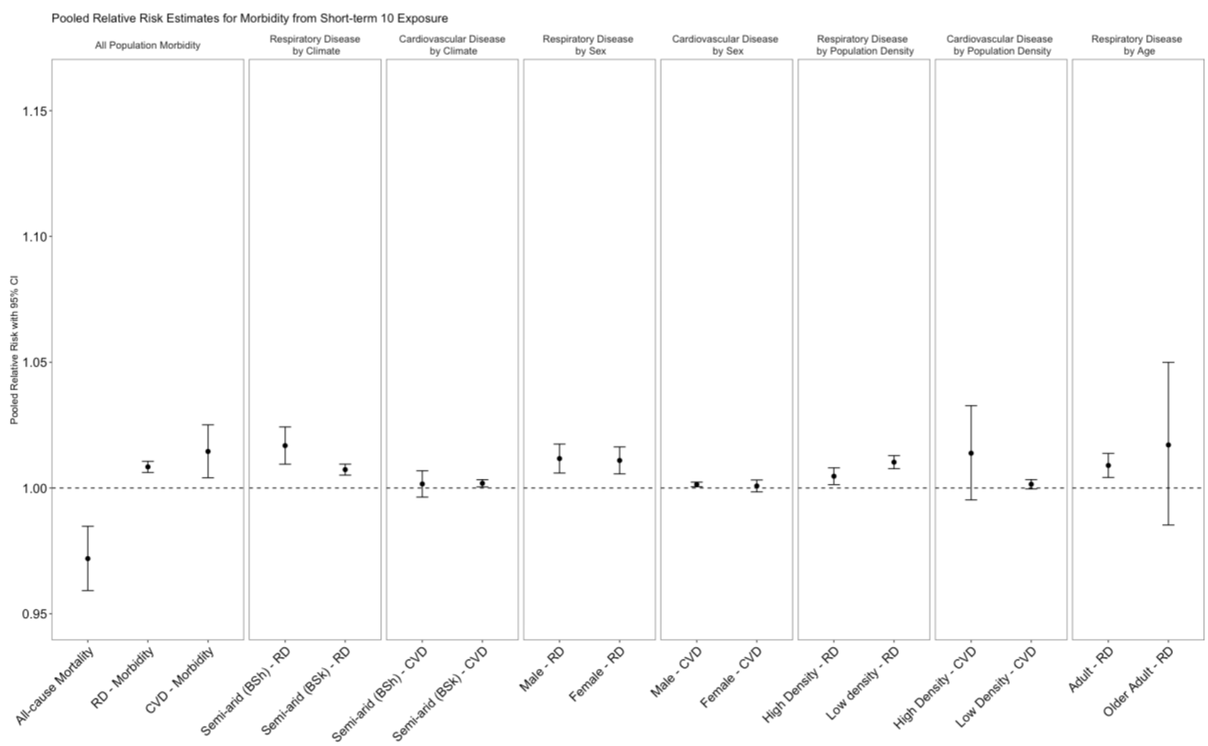
$$Cochran's\ Q = \frac{(\text{ß}_{GroupA} - \text{ß}_p)^2}{VAR_{GroupA}} + \frac{(\text{ß}_{GroupB} - \text{ß}_p)^2}{VAR_{GroupB}}$$

Meta-Analysis Code
RStudio Version 2023.03.0+386 Using package (meta)
Maximum-likelihood estimator for estimation the between-study variance calculated using metagen (method.tau="REML")

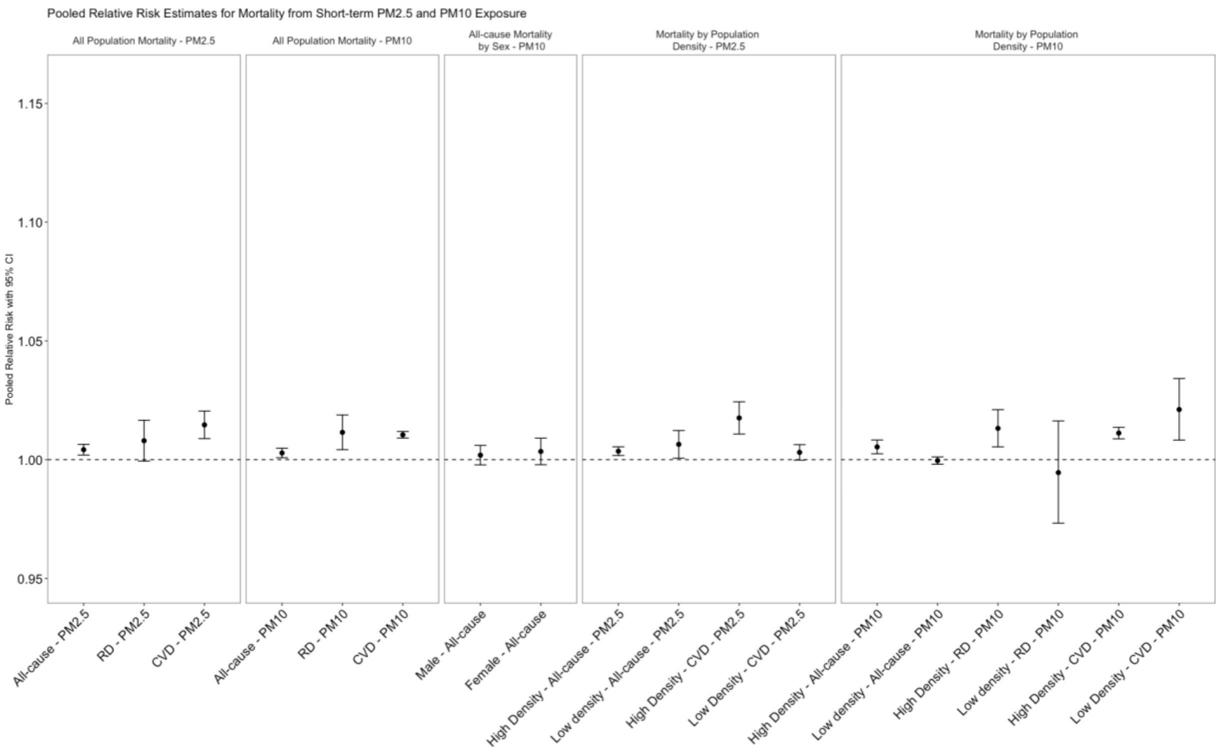
Supplemental Figure 2.2. Equations used in Meta-Analysis



Supplemental Figure 2.3. Study Statistics



Supplemental Figure 2.4. Pooled Effect Estimates for PM_{10} exposure



Supplemental Figure 2.5. Pooled Effect Estimates for Mortality

Chapter 2 References

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Chapter 3 Spatial and Temporal Variance in Source Apportioned $PM_{2.5}$ Health Risks Across California: Population and Climate Effect Modifiers

Introduction

Particulate matter (PM), emitted from multiple global sources, has a well-studied impact on the human body. Predominantly entering the body through inhalation, particles of many sizes and chemical compositions can cause harmful reactions such as inflammation, cell death and genotoxicity (Arias-Pérez et al., 2020; Riediker et al., 2019). Acute short-term exposure can lead to premature death from numerous conditions including stroke (Shah et al., 2015), and hospitalization from a range of diseases, of which cardiovascular and respiratory diseases are the most commonly studied (Kyung & Jeong, 2020; Martinelli et al., 2013). On long-time scales, particulate matter can lead to a wide range of health issues, such as asthma, chronic obstructive pulmonary disease (COPD), Alzheimer's disease, cancer, anxiety & depression, memory loss, and infertility (T. Li et al., 2022; Sharma et al., 2023; Thompson, 2018). Further, research has found significant differences in the overall health burden between socioeconomic and racial/ethnicity groups (Geldsetzer et al., 2024; Ma et al., 2023) and by sex and age (Bell et al., 2013), highlighting inequality in both exposure and susceptibility, as well as the complexity of exposure types. Overall, the relationship between exposure to specific particulate matter source composition in different communities is thought to play a role in health outcomes. To assess the health effects of short-term PM exposure, studies commonly study the variability of modeled or measured outdoor PM concentrations and the rates of hospitalization or mortality from specific diseases overtime in a chosen population, to see if there is a statistical correlation between increased exposure to ambient PM and increased health outcomes (e.g., Liu et al., 2019; Orellano et al., 2020).

Risk from exposure to PM has been predominantly studied based upon the concentration (often in $\mu g/m^3$) and size (e.g., $\leq PM_{2.5}$, particles of 2.5 micrometers or less) of particles (Pope et al., 2009), though increasingly the chemical composition and oxidative potential of particles, as a product of the emission source type, is found to significantly modulate its effect on the body (Kelly & Fussell, 2012). For example, recent studies have linked wildfire smoke to disproportionately high respiratory mortality (Aguilera et al., 2021). Additionally, using long-term PM monitoring in Atlanta, GA, researchers found biomass burning and vehicular emissions to be uniquely correlated to increased number of cardiorespiratory emergency department visits, relative to other PM sources such as road dust and secondary nitrate (Pennington et al., 2019). Similarly, in California across eight cities over an eight year period, biomass burning was associated with all-cause mortality, cardiovascular mortality, and ischemic heart disease, whereas vehicular emissions were associated with cardiovascular mortality (Berger et al., 2018). While some trends have emerged—such as findings of high risk from elemental and organic carbon found in wildfire, biomass burning, and vehicle exhaust—there is still a great deal of unknown concerning which sources of particulate matter are most harmful, and to which communities. The temporal and spatial variability of particulate matter emissions and human exposure may play a role in this as well, such as the longevity of heavy metals in urban road dust that will persist (Roy et al., 2022), versus volatile organic compounds that quickly degrade or evolve in sunlight (Atkinson & Arey, 2003). Notably, the complexity in heterogeneous particulate matter causes a difficulty in the ability of public health researchers to identify the most harmful PM sources.

The application of receptor modeling (Gordon, 1988) of chemically speciated particulate matter to assess differential impact on mortality and morbidity is a growing field of research used to address nuanced correlation between PM exposure and health outcomes (P. K. Hopke, 2016; Krall & Strickland, 2017). In the United States, there are two monitoring networks with stations across the nation that collect particulate matter and measure it for a wide range of chemical and physical parameters—every three days or every six days (Solomon et al., 2014). There are similar networks in China (Dao et al., 2019) and Europe (Pandolfi et al., 2020). These offer a unique resource, creating publicly available high temporal resolution and long-term PM data that can be directly studied alongside health data. Such data, however, requires substantial preparation and source apportionment modeling before being linked to health data. In addition, some of the issues with this approach are the lack of spatial resolution, as in most cases there is only one station in each city or region, and a limited ability to extrapolate findings over a wide region, due to the high variability in localized emission, transport behaviors, and longevity in the atmosphere. In most cases, researchers have chosen to only compare findings to hospitalization data from populations living within 10, 20, or 50 km of the monitoring station (Berger et al., 2018; Rowan et al., 2024).

In parallel, researchers have explored the effect modification by climate in particulate matter health risks. Notably, the relationship between high temperatures and PM exposure, creating a synergistic effect, has recently been demonstrated (Anenberg et al., 2020; Chen et al., 2024; Rahman et al., 2022) concluding there is an unexplained compounding health effect. As anthropogenic climate change has led to higher temperatures, heat waves (Marx et al., 2021), and exacerbated wildfires (Keeley & Syphard, 2016; Mansoor et al., 2022) are two of the numerous ways in which climate change may exacerbate particulate matter impacts and exposure. Additionally, climate change has–and is projected to increasingly–destabilize land surfaces through extreme and highly variable weather events, which is thought to cause a net increase in PM emissions from existing and emerging dust sources (Achakulwisut et al., 2019; Park et al., 2020, p. 202). Different climates have unique natural particulate matter sources, with a wide range of chemical composition, which create complex relationships between climate regions and health outcomes. The most highly investigated climate-specific source of particulate matter exposure is desert dust (Goudie, 2014; Zhang et al., 2016), while differences between soil types, different forms of agriculture, dwellings, industries, lifestyles, etc., are less explored.

In summary, there are numerous intersecting variables that can lead to heterogeneous particulate matter health impacts across communities. This research aims to explore the association between short-term exposure to $PM_{2.5}$ and cardiorespiratory hospitalizations across the geographically and socially diverse state of California, US. Using previously developed methods and existing datasets and methods, we explore the health risks specific to each $PM_{2.5}$ source and spatiotemporal effect modifiers.

2 Methods

This research employs three separate datasets and uses two sequential analytical modeling techniques. Following a framework to support best practices in association of environmental and other exposures with health outcomes (PECO; Morgan et al., 2018), we consider the population of California (population), and the effect of short-term exposure to PM (exposure), comparing sociodemographic and geographic effect modifiers (comparator) on cardiovascular and respiratory hospitalizations (outcomes).

2.1 Chemically speciated PM2.5 data

In this study, more than ten years of $PM_{2.5}$ data from 35 monitors across California were used (Figure 1), which are part of two nationwide chemically speciated monitoring networks (Solomon et al., 2014). The Interagency Monitoring of Protected Visual Environments (IMPROVE) network was initiated in 1985 to assist in the implementation of the 1977 Clean Air Act, and to monitor air quality in remote environments, specifically National Parks and Wilderness Areas. The EPA Chemical Speciation Network (CSN) was established in 2000 with monitors in urban and suburban sites to support research of health and exposure and environmental welfare. Both networks collect integrated 24-hr $PM_{2.5}$ samples on a 1-in-3-day or 1-in-6-day schedule. Owing to the fact that there are multiple types of samplers used (Koo et al., 2012) careful attention was spent to understand the error in the measure of each element and comparability of concentrations (Spada & Hyslop, 2018). Using the method detection limit (MDL), the fractional uncertainty, and sample element concentration, the total uncertainty was calculated for each sample and compiled in an uncertainty matrix, which is a required input for PMF.

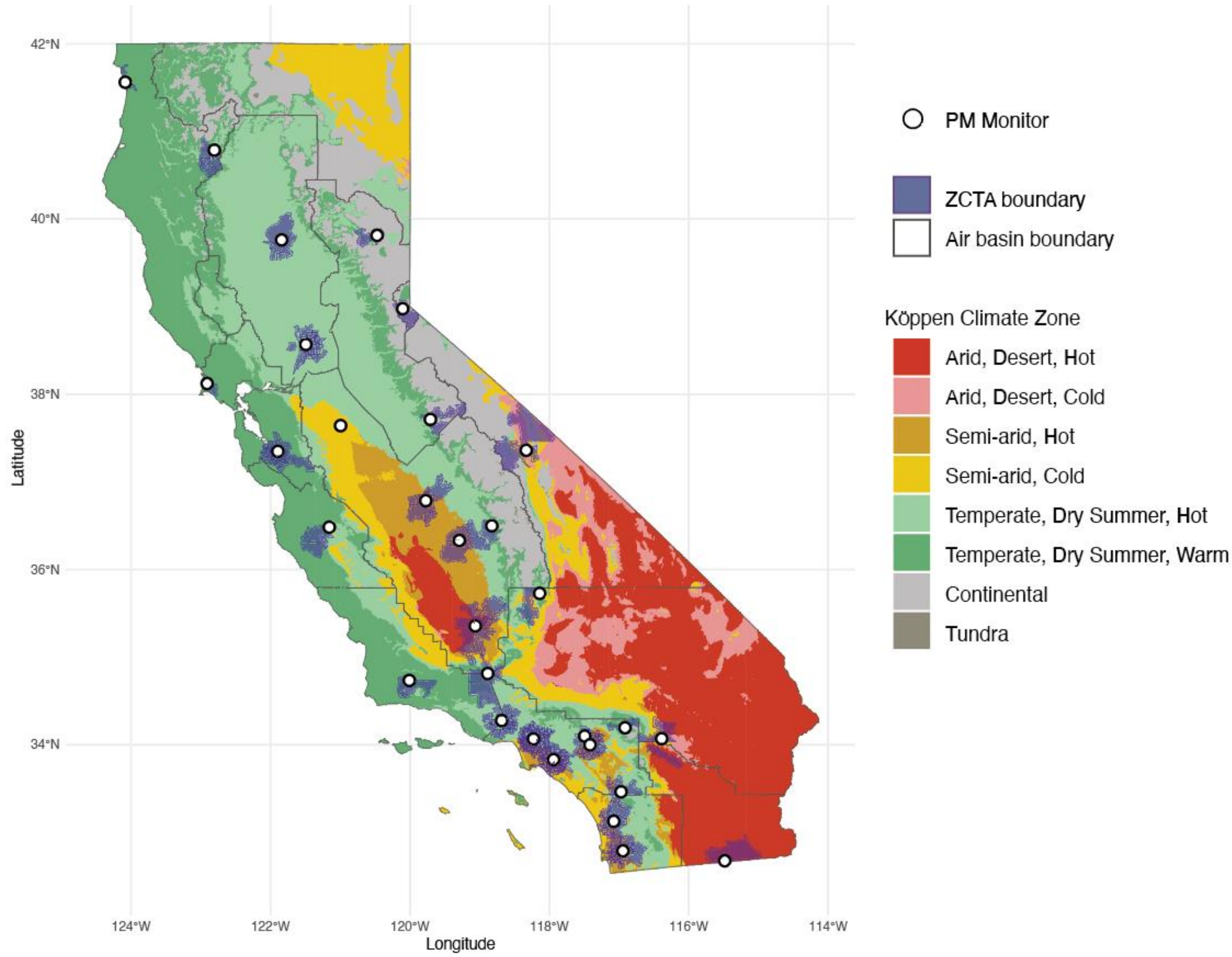

Figure 3.1. Study map showing the locations of PM2.5 monitors and corresponding ZCTA boundaries, along with outlines of air basins, overlaid on a map of the Köppen climate zones.

2.2 $PM_{2.5}$ Data Preparation

Data from both CSN and IMPROVE sites was accessed from the EPA DataMart and federal databases for the range of dates specified in Table S1. For CSN sites, the start dates correspond to the date after the method of carbon measurement was changed (Spada & Hyslop, 2018), while IMPROVE dates begin on January 1, 2006, and both end on December 31, 2020. The following data preparation follows previously established and tested methods for the preparation of CSN and IMPROVE data when applicable (Hasheminassab et al., 2014; E. Kim et al., 2005; Kotchenruther, 2013) and is shown in Figure 2. Samples on July 4 and 5, as well as December 31 and January 1 of all years were removed to limit the influence of uncharacteristic events, specifically fireworks, on the model. When not reported, average element specific method detection limits and uncertainties calculated by Kim et al. (2005) were used. Chemical species with more than 70% of the values below detection limit (BDL) were not used. To facilitate direct comparison between sites, if a chemical was higher than 70% BDL for the majority of sites, the element was not used for any site. Samples with all relevant species concentrations missing were deleted. Similarly, if a sample reported data for less than 50% of included species, it was discarded. Missing values were set as geometric mean of the species, and the sample uncertainties were set at four times the concentration (Hasheminassab et al., 2014; E. Kim et al., 2005). The organic carbon (OC) concentration was recalculated by subtracting the intercept of the OC to $PM_{2.5}$ mass regression to account for the OC artifact (E. Kim et al., 2005). Some elements are measured in multiple forms (e.g., K/K^+, Na/Na^+). To avoid counting the element concentration twice, one was removed, by assessing the data completeness, amount below detection limit, and S/N ratio during PMF analysis. Finally, K^+, Na^+, and sulfate (SO_4^{2-}), were selected, although some sites did not report Na^+, in which case Na was used. For sample concentrations that were below the BDL, the concentration was set to half the MDL and the uncertainty was set to ⅚ of the DL values (Polissar et al., 1998). Finally, 18 species were included in CSN and IMPROVE datasets to be used for PMF analysis (Figure 2). Additionally, all sites were characterized by the following categories: climate, using the Köppen climate classification scheme (Köppen, 1918); air basin, using the 15 distinguished air basins of California; and coastal (<50 km from Pacific Ocean) or inland (>50 km from Pacific Ocean).

2.3 Source Apportionment via Positive Matrix Factorization

For this study, source apportionment modeling was performed using the Environmental Protection Agency Positive Matrix Factorization model 5.0 (EPA PMF 5.0). A detailed description of the model, applications, and mathematical approach can be found in Paatero (1997), Paatero & Hopke (2003), and Norris (2008). The two required matrices (concentration and uncertainty) from each site were run separately for the entire period. Chemical species were discarded (set to "bad") if their signal-to-noise ratio (S/N) was less than 0.2. If the species S/N was greater than 0.2 but less than 2, it was set to “weak”. $PM_{2.5}$ mass was run as an individual species and set to “weak”. An extra 5% modeling uncertainty was added to account for additional modeling error (Hasheminassab et al., 2014). The model was run 20 times in robust mode to ensure the least-squares solution was a global, instead of local, minimum. An optional parameter to rotate factorized solutions (FPEAK) was left at the default value of 0. For each site, the model was run several times with between five and nine factors to find a solution that best represented the unique site data. The chosen number of factors was determined by a combination of considerations, including the Q/Qexp value, results from bootstrap (BS) error estimation, and the scaled residuals were generally distributed and between -3 and 3. In all cases the final number of factors chosen was between 6-8. To account for the inconsistency in sample frequency (e.g., every 3, 6, 9, etc. days), the factor concentrations from each sample in a given calendar week during the study period was averaged together to yield a weekly average for consistent pairing with hospitalization data.

2.4 Hospitalization Data

Information on daily unscheduled hospital visits and emergency department admissions (collectively referred to as “hospitalizations” in this study) in California between 2007 and 2019 was obtained from the Patient Discharge Data and Emergency Department Data maintained by the California Department of Health Care Access and Information (State of California, 2024). The dataset included each patient’s residential ZIP code, the date of hospitalization, and the primary International Classification of Diseases (ICD)-9 or ICD-10 diagnosis codes. This study focused on cardiorespiratory hospitalizations (both circulatory and respiratory hospitalizations), which were identified using the corresponding ICD-9 or ICD-10 codes (see Supplementary Methods 1). Each ZIP code was linked to its corresponding ZIP Code Tabulation Area (ZCTA), and the daily individual-level hospitalization data were aggregated into weekly counts of cardiorespiratory hospitalizations by ZCTA. We then calculated the weekly ZCTA-level cardiorespiratory hospitalization rates (per 100,000 people) as the outcome variable.

2.5 Air Temperature Data

Daily mean air temperature data, available at a 4 km × 4 km spatial resolution from 2007 to 2019, were obtained from the Parameter-elevation Regressions on Independent Slopes Model (PRISM) Climate Group (PRISM Climate Group, 2016). We extracted the daily ZCTA-level air temperature data using the population-weighted centroid of each ZCTA and then averaged the daily values into weekly means to align with the source-specific $PM_{2.5}$ data.

2.6 Epidemiological Analysis

To estimate the association between weekly exposure to source-specific $PM_{2.5}$ and weekly cardiorespiratory hospitalization rates, we applied a two-way fixed effects (TWFE) model which exploits spatiotemporal variation in both exposure and outcome. Only ZCTAs with a population of at least 1,000 in the 2010 U.S. Decennial census were included, to ensure statistical power. In our study, the main model can be expressed as

$$HospitalizationRate_{z,w} = \alpha_z + \theta_w + \beta SourceSpecificPM2.5_{z,w} + ns\left(Temperature_{z,w}, df = 3\right) + \varepsilon_{z,w}$$

where $HospitalizationRate_{z,w}$ represents the cardiorespiratory hospitalization rates in ZCTA z and week w. $SourceSpecificPM2.5_{z,w}$ is the weekly mean source-specific $PM_{2.5}$ concentration in in ZCTA z and week w. We ran separate models for each $PM_{2.5}$ source. We modeled weekly air temperature using a flexible natural cubic spline with three degrees of freedom (df). α_z captures ZCTA-specific, time-invariant effects and θ_w represents common time-varying effects across all ZCTAs. Including indicators for each ZCTA and each week ensures that the model can control for all spatial confounders that differ across ZCTAs (e.g., levels of urbanization) and all temporal confounders that differ over time (e.g., seasonal trends), either measured or unmeasured (Goodman-Bacon, 2021). $\varepsilon_{z,w}$ is the error term. To improve the precision of our estimates, we applied weights based on the population size of each ZCTA

(Zang & Kim, 2021). Clustered standard-errors (by ZCTA) were used to compute 95% confidence intervals.

We further examined whether these associations differed by time and location. To assess the potential effect modification by seasonality, we conducted stratified analyses by season (spring: March to May; summer: June to August; fall: September to November; winter: December to February). In addition, we explored regional variation in the association by performing stratified analyses by both climate zones and air basins. The 11 air basins in California were established in 1967, with data from California Air Resources Board (CARB) (California Air Resources Board, 2024). The Köppen climate classification scheme (Köppen, 1918) was used to classify the climate of each $PM_{2.5}$ monitor, using the 1991-2020 map (Beck et al., 2023). Only climate zones or air basins with at least 5 studied ZCTAs were included in these secondary analyses.

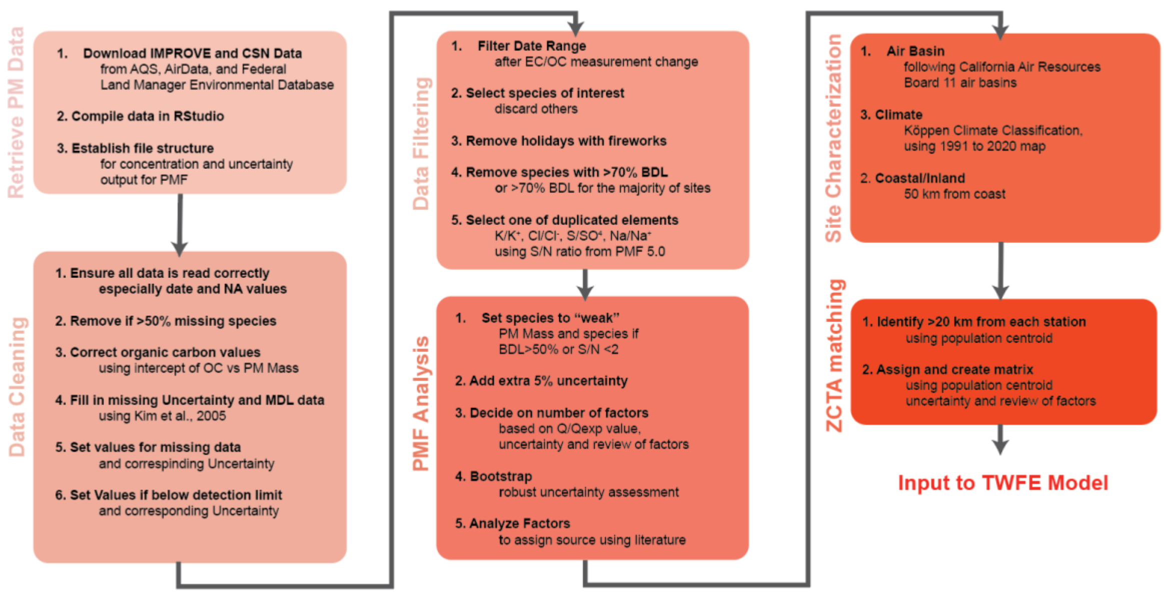

Figure 3.2. A conceptual flow diagram showing the steps to prepare speciated PM2.5 data for two-way fixed effects model, including data retrieval, data cleaning, data filtering, PMF analysis, and site characterization.

3 Results

3.1 Identified $PM_{2.5}$ sources and compositions

In consideration of the 35 sites spanning diverse geographic locations (Figure 1), we aimed at identifying sources from PMF modeled factors and determine the uniqueness and comparability of factors between sites. Overall, nine unique sources were identified; a detailed explanation of each source follows. Our factor identification builds off previous studies using similar PMF approaches and elements, and in some cases data from the same monitoring sites (e.g., Hasheminassab et al., 2014). As with all uses of PMF analysis, the assignment of sources to model-identified factors is subjective. For the purposes of this study, emphasis is placed on identifying comparable factors between sites, characterized by dominant species and contribution ratios, that may pose a unique impact on human health due to their composition or temporality. A full account of which factors were present at each site is shown in Table S1. The nine factors were identified as:

Dust. The most straightforward factor was characterized by high Al, Ca, Si, Ti, Mg contributions, which is commonly referred to as soil, road dust, or crustal material. This was found in all sites and has been identified in many previous studies (e.g., Kim et al., 2010).

Sea Salt. Especially in coastal environments, sea salt (predominantly Na and Cl) is one of the most abundant aerosols and undergoes complex reactions with nitrate (NO_3^-) and sulfate (SO_4^{2-}) in the urban atmosphere as it ages (Adachi & Buseck, 2015). The result is that Cl is generally stripped, leaving Na with a mixture of NO_3^- and SO_4^{2-}. This sea salt signature without Na is often called aged sea salt (w.g., Kim et al., 2010; Kotchenruther, 2013). Fresh sea salt was previously found to be highest in Southern California in the spring (Hasheminassab et al., 2014) because of southwesterly winds (Cheung et al., 2011). For the purposes of this study and modeling approach, the two sources of similar origin but different stages of evolution were combined into one category to enable differentiation between source and health outcomes.

Industrial. This source category combines two factor profiles with high concentrations of metals (Zn or Cu), and elemental carbon (EC), and organic carbon (OC), and NO_3^- or SO_4^{2-}. Similar profiles identified in PMF outputs in other studies identify these profiles as industrial, with high EC and OC and metal content (E. Kim et al., 2010). For example, this source is similar to what was identified as an incinerator in the South Coast Air basin (E. Kim & Hopke, 2007). With a large number of differences between industrial activities and emission types around the state, likely producing the range of similar factor profiles, we grouped the factors to distinguish all industrially sources PM from PM from other origins.

Biomass Burning. This factor, also covering a range of combustion processes, it is generally characterized by a high concentration of elemental and organic carbon in addition to K (Dadashazar et al., 2019; Hasheminassab et al., 2014; Schlosser et al., 2017). At different sites this factor had variable amounts of NO_3^- and SO_4^{2-} and other elements such as Fe, Cu, Cl, Si, and Ca, which may be due to the different source of the biomass burning—e.g., whether it was from wood-burning stoves, wildfire, or agricultural burning.

Secondary Nitrate. This factor is identified by a high concentration of nitrate—generally over 50% of the species—as well as ammonia with the inclusion of lower concentrations of several metals. Secondary nitrate is thought to be created through the evolution of precursors of nitrogen oxides (NOx) and ammonia (NH_3) that undergo transformation in the atmosphere with temperature, humidity, and reactive oxygen species. Agriculture (notably dairy farming) is a dominant source of NH3 in California (E. Kim & Hopke, 2007), while NOx is often attributed to vehicle emissions (McDonald et al., 2012), making this a complex source. Secondary Nitrate has a long history of study and attempts to reduce precursor concentrations in California (Christoforou et al., 2000).

Secondary Sulfate. High concentrations of sulfate with (>50% of species total) with lesser contributions of ammonium and metals such as bromine, magnesium, and sodium. Previously identified sources of secondary surface are shipping emissions and dimethlysulfide (DMS) emissions from the ocean (Dadashazar et al., 2019; Moldanová et al., 2009). A factor with a similar profile is sometimes termed Residual Fuel Oil (RFO, (Kotchenruther, 2013).

Secondary Mixture. This source was identified as a mixture of multiple species without a consistent characterizable profile, found in several cities. Species contributions include a large percentage of metals (e.g., Cu, Zn), elemental carbon, and both nitrate and sulfate with highly variable concentrations. This is interpreted as an unidentifiable mixture of PM emissions from several source types.

Combustion (OC) and Combustion (EC). Two forms of carbonaceous particles included, elemental carbon (EC) and organic carbon (OC), are distinguished during measurement by optical properties and are thought to reflect different types of carbon combustion (Spada & Hyslop, 2018). EC particles originate from incomplete combustion of fossil fuels and are often a marker of diesel emissions (Schauer, 2003). Organic carbon includes a wider range of particles that come from both primary combustion and formation in the atmosphere through secondary reactions (Seinfeld & Pandis, 2016). Previous source apportionment studies with similar factor profiles characterized by high OC concentrations were identified as gasoline vehicle emissions, while factors with high EC concentrations were identified as diesel emissions (E. Kim et al., 2010). For the purposes of this study the profiles are named as combustion (OC), with higher OC than EC concentrations, and combustion (EC) for profiles with relatively higher EC contributions.

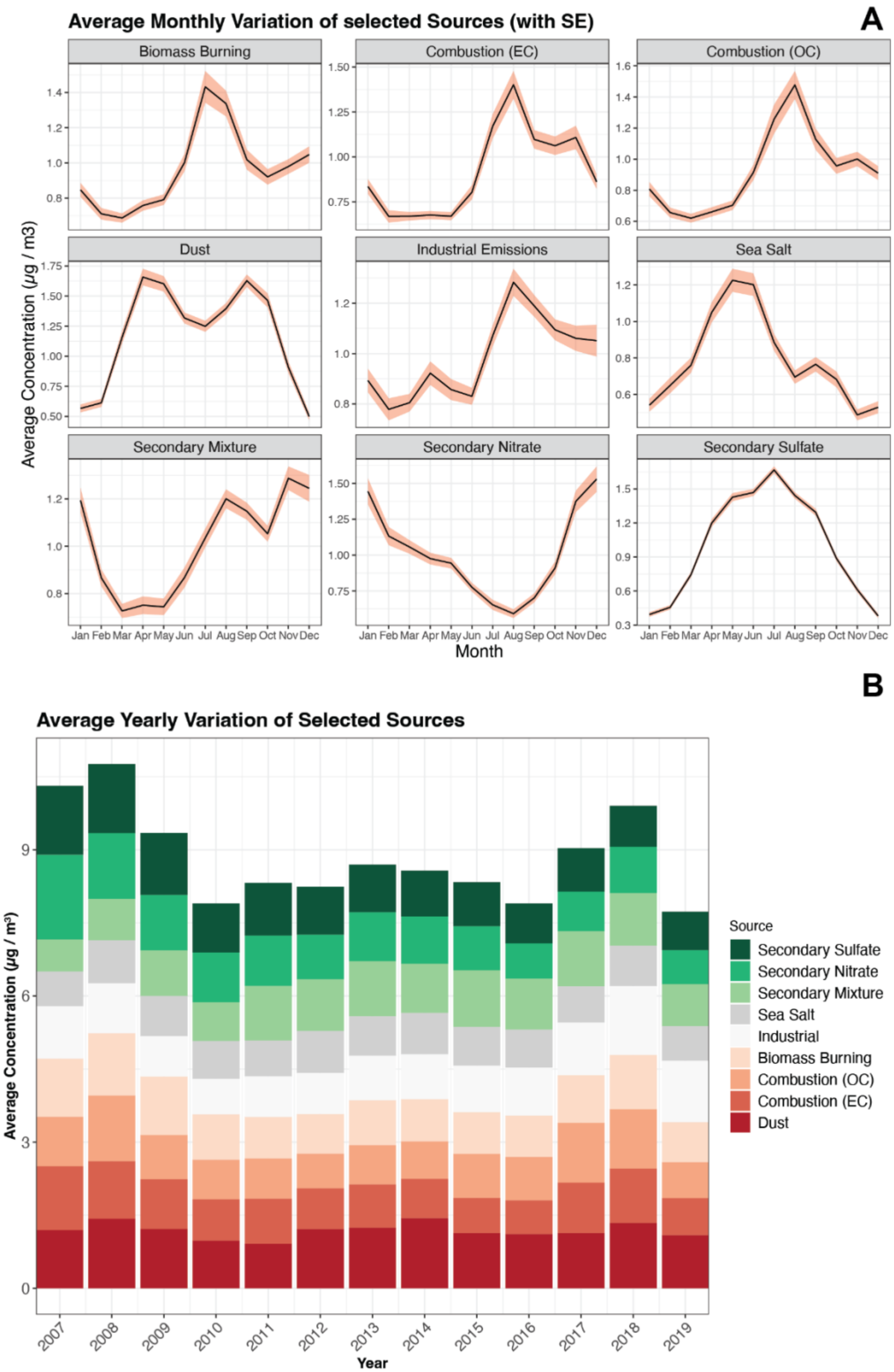

Figure 3.3. A) The average monthly source concentrations (μg/m3), and B) annual average concentrations show variation over the study period with standard error (red shading).

Table 3-1. Descriptive statistics for weekly source-specific PM2.5 and cardiorespiratory hospitalization among the 486 ZCTAs within 20 km of speciated PM2.5 monitors in California, 2007-2019

	Mean (SD)	Min	Median (IQR)	Max
Vehicular (OC) (μg/m^3)	1.94 (2.28)	0.00	1.26 (2.11)	24.36
Vehicular (EC) (μg/m^3)	1.49 (1.42)	0.00	1.05 (1.48)	21.20
Secondary nitrate (μg/m^3)	1.98 (2.73)	0.00	1.24 (1.92)	50.89
Secondary sulfate (μg/m^3)	1.86 (1.64)	0.00	1.50 (2.06)	15.67
Sea salt (μg/m^3)	1.33 (1.61)	0.00	0.86 (1.99)	13.59
Soil (μg/m^3)	1.41 (1.58)	0.00	0.96 (1.86)	15.54
Biomass (μg/m^3)	1.58 (1.49)	0.00	1.22 (1.42)	44.50
C-metals (μg/m^3)	1.70 (1.55)	0.00	1.31 (1.76)	23.07
Secondary mixture (μg/m^3)	1.63 (1.61)	0.00	1.21 (1.62)	13.96
Total $PM_{2.5}$ mass (μg/m^3)	11.95 (6.66)	0.15	11.45 (5.80)	75.41
Cardiorespiratory hospitalization rate (per 100,000 people)	95.78 (57.75)	0.00	84.23 (58.25)	1086.96

3.2 Descriptive results

A total of 10,898,971 cardiorespiratory hospitalizations across 486 ZCTAs in California from 2007 to 2019 were included in this study. The average weekly ZCTA-level cardiorespiratory hospitalization rate was 95.78 (SD: 57.75) per 100,000 people (Table 1). Among the 9 identified sources of $PM_{2.5}$, the weekly mean concentration of secondary nitrate was the highest (1.98 μg/m^3; SD: 2.73 μg/m^3), followed by combustion (OC) (1.94 μg/m^3; SD: 2.28 μg/m^3) and secondary sulfate (1.86 μg/m^3; SD: 1.64 μg/m^3) (Table 1). Different sources of $PM_{2.5}$ showed weak to moderate correlations (Table S2). Source concentrations show variability over time and between sites. Averaged across the 35 sites, individual sources showed unique seasonal concentration patterns, with most sources (e.g., combustion (OC and EC), biomass burning, industrial) reaching maximum concentrations in the fall, while some had maximums in the summer (Sea Salt and Secondary Sulfate) and others in the winter (Secondary Mixture and Secondary Nitrate) (Figure 3A). Secondary nitrate and secondary sulfate decreased from 2007 to 2019, while others had no consistent trends. Several sources—including combustion (OC), combustion (EC), biomass burning, industrial, and dust—show an anomalous increase in concentrations in 2018, with the spike starting in 2017 for combustion (OC) and combustion (EC).

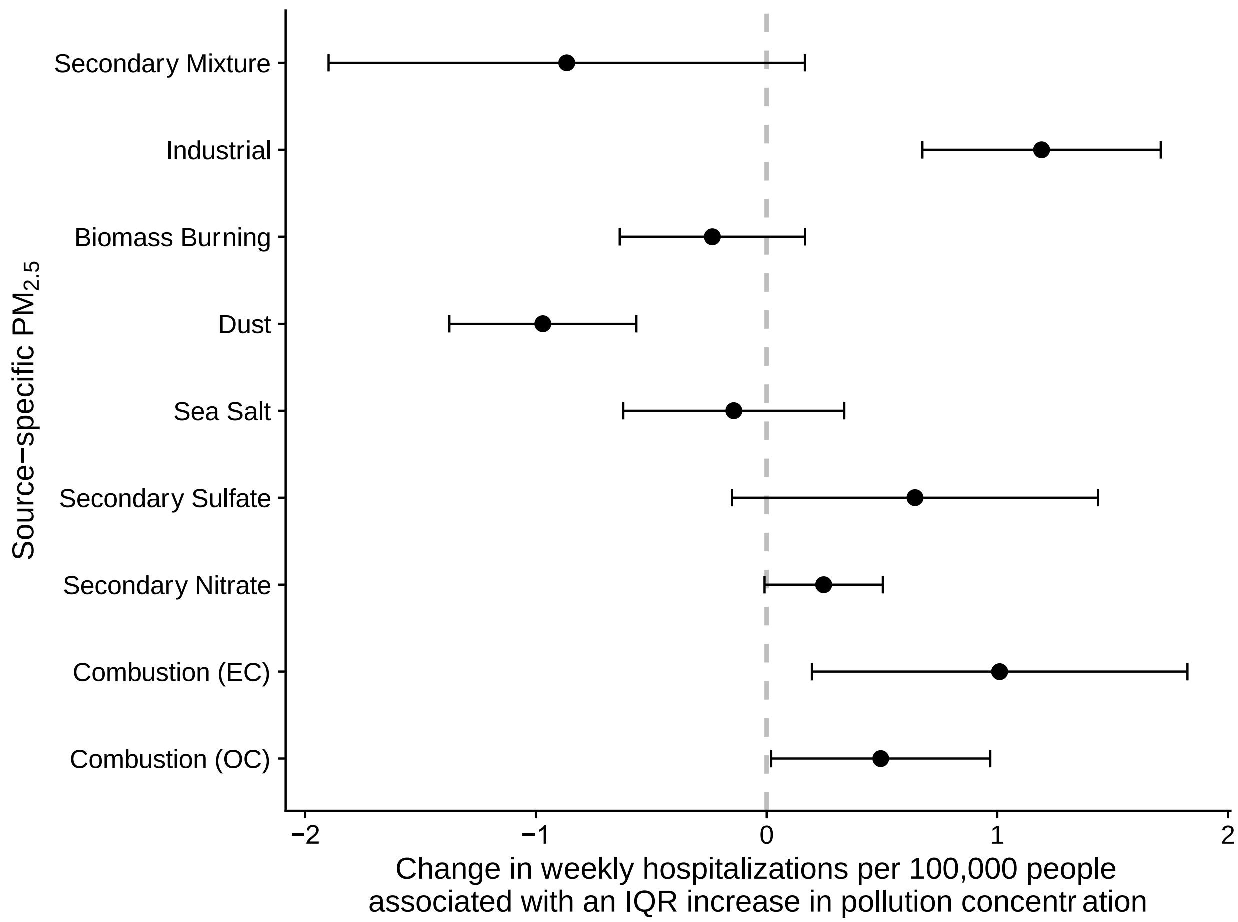

Figure 3.4. Association between weekly exposure to source-specific PM2.5 and weekly cardiorespiratory hospitalizations in California3.4 Association between source-specific $PM_{2.5}$ and hospitalizations

3.3 Overall Effects

Figure 4 presents the estimated associations between exposure to $PM_{2.5}$ from each source and weekly cardiorespiratory hospitalization rates. We found that each interquartile range (IQR) increase in $PM_{2.5}$ concentrations from industrial, secondary sulfate, secondary nitrate, combustion (EC), and combustion (OC) were associated with increases in cardiorespiratory hospitalizations of 1.19 (95% CI: 0.68, 1.71), 0.64 (95% CI: -0.15, 1.44), 0.25 (95% CI: -0.01, 0.50), 1.01 (95% CI: 0.20, 1.82), and 0.49 (95% CI: 0.02, 0.97) per 100,000 people, respectively. In contrast, increases in $PM_{2.5}$ originating from biomass burning, dust, and sea salt showed negative associations.

3.4 Effect modification by season

Our stratified analyses suggested that season modifies the association between source-specific $PM_{2.5}$ and cardiorespiratory hospitalizations, and that these seasonal patterns vary by $PM_{2.5}$ source (Figure 5). For example, the secondary mixture showed a positive association in fall, but a negative one in spring. While industrial emissions were consistently associated with increased hospitalization rates across all seasons, the magnitude was greater in fall and winter than in spring and summer. For biomass burning, soil, and sea salt, some seasons displayed negative estimates, while others were nearly null. For $PM_{2.5}$ from secondary sulfate, secondary nitrate, and combustion (EC), we observed positive associations during spring, summer, and fall, but negative associations in winter. Meanwhile, combustion (OC) was positively associated with cardiorespiratory hospitalizations in every season, though the effect magnitude was highest in spring.

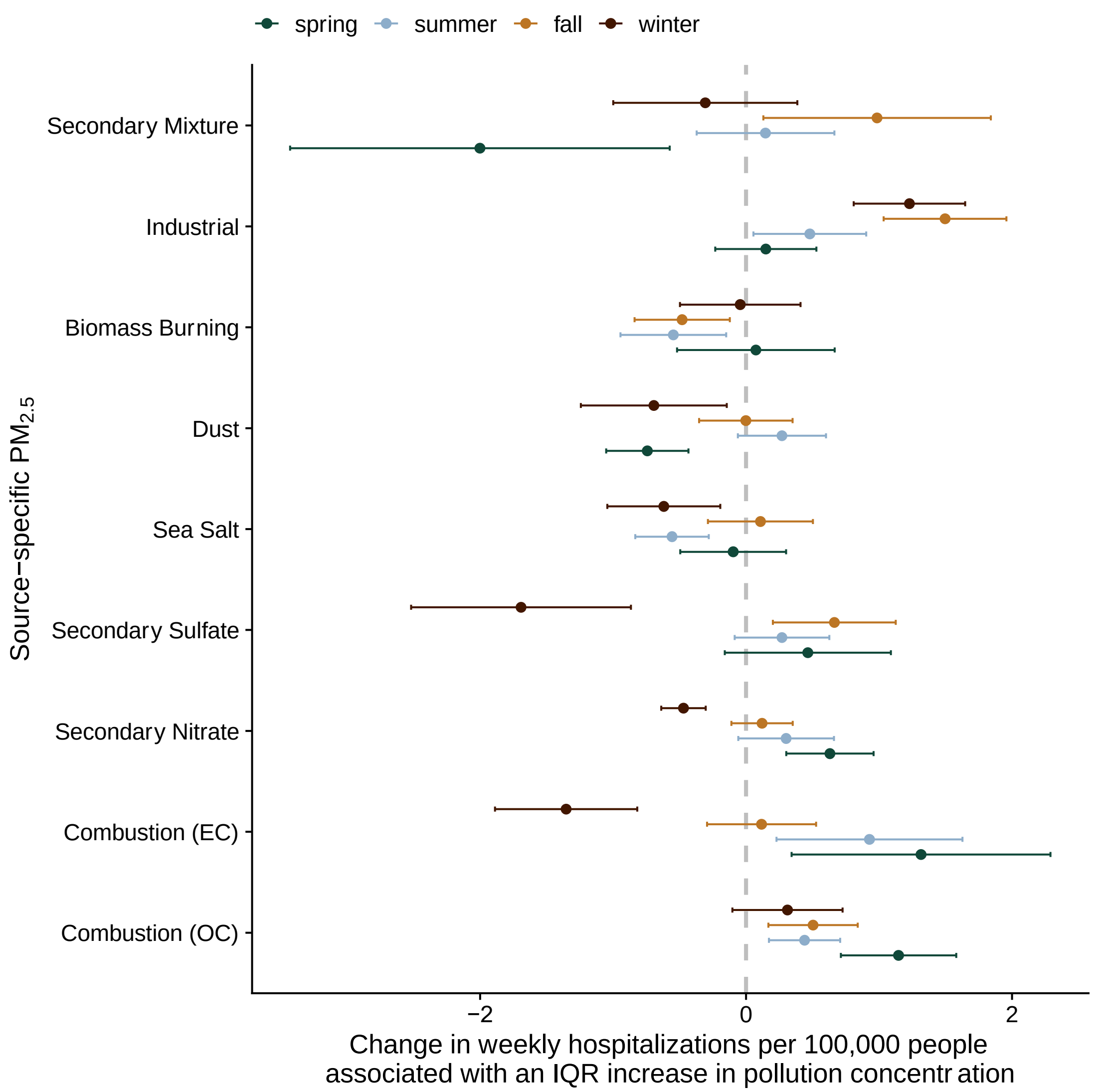

Figure 3.5. Association between weekly exposure to source-specific PM2.5 and weekly cardiorespiratory hospitalizations by season in California

3.5 Effect modification by region

We observed that the associations between source-specific $PM_{2.5}$ and cardiorespiratory hospitalizations varied across different climate zones (Figure 6). In arid, desert, hot regions, industrial emissions were linked to increased hospitalization rates, whereas $PM_{2.5}$ from biomass burning, soil, secondary sulfate and nitrate, and vehicular (OC) emissions showed negative associations. In semi-arid, cold regions, secondary nitrate and vehicular (OC) emissions were positively associated with hospitalizations, but $PM_{2.5}$ from soil had a negative association. In semi-arid, hot regions, secondary mixture, industrial, and vehicular (EC) emissions were positively associated, while other $PM_{2.5}$ sources had negative or negligible effects. In temperate regions with dry summers, $PM_{2.5}$ from sea salt, secondary nitrate, and vehicular (OC) emissions were positively associated, whereas other sources showed negative or no associations. Finally, in temperate regions with warm summers, secondary sulfate and vehicular emissions (both OC and EC) were positively associated, while other sources again exhibited negative or negligible effects. Similarly, heterogeneity was observed in stratified analysis by air basin (**Figure 7**).

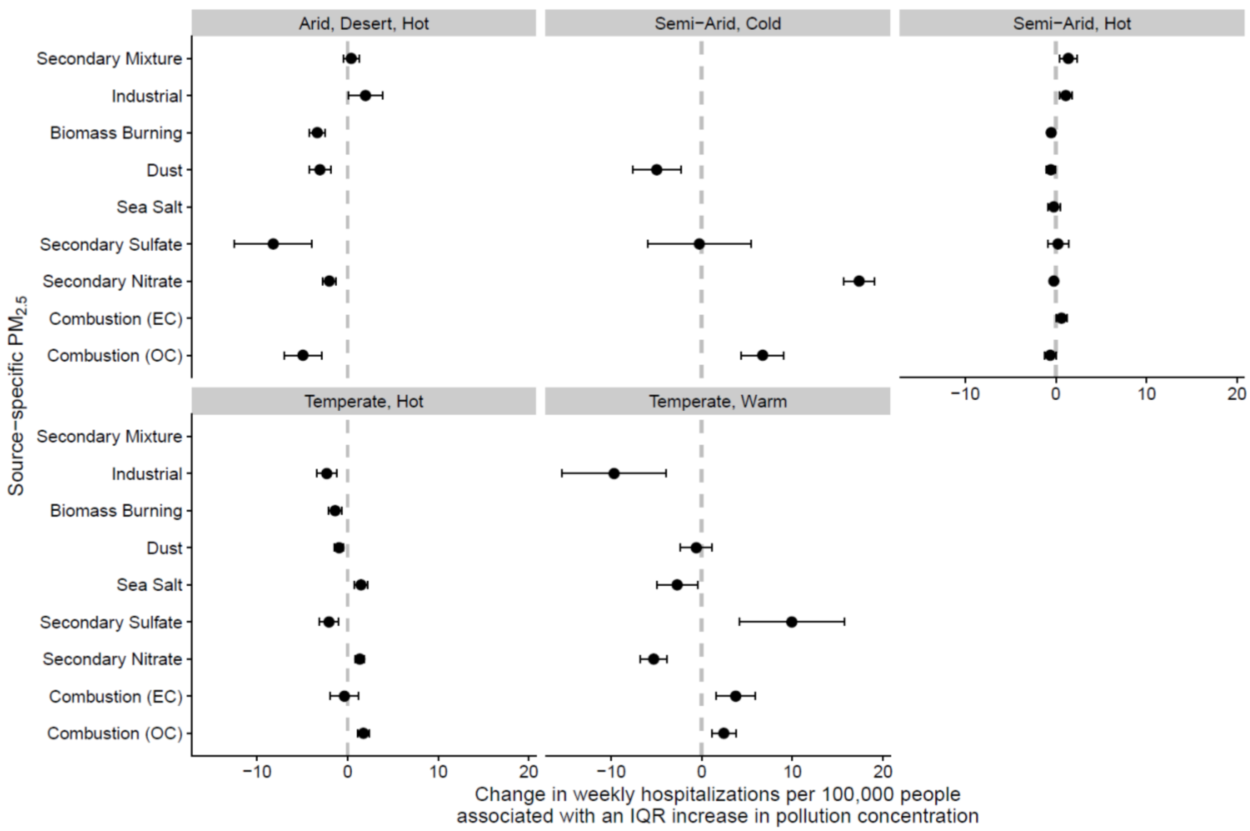

Figure 3.6. Association between weekly exposure to source-specific PM2.5 and weekly cardiorespiratory hospitalizations by climate zone in California

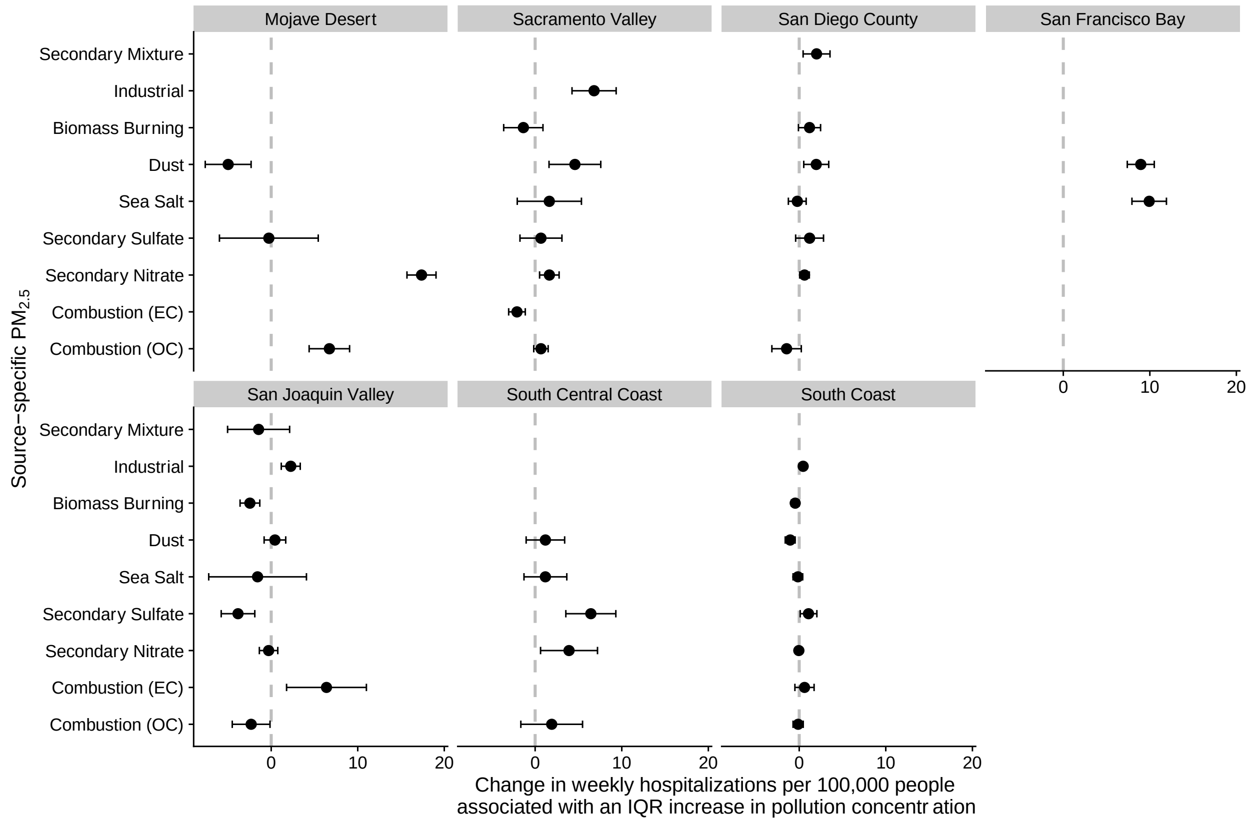

Figure 3.7. Association between weekly exposure to source-specific PM2.5 and weekly cardiorespiratory hospitalizations by air basin in California

4 Discussion

While the adverse relationship between PM and human health has been well characterized, less is known about source specific PM exposure risks and variability over space and climate. This study used more than 10 years of chemically speciated $PM_{2.5}$ data at 35 sites across California to assess the association with cardiorespiratory hospitalizations. PMF analysis identified reoccurring $PM_{2.5}$ sources across sites and revealed spatial and temporal variations in source concentrations (Table S1), while TWFE modeling found unique relationships between sources and hospitalization rates, which was variable across the study sites.

The average annual concentration in total $PM_{2.5}$ was found to slightly decrease over this more than 10-year study period, with increases in 2017 and 2018 (Figure 3B). Secondary nitrate and sulfate and combustion (EC) show decreases trends over this period, although others such as industrial PM increase, and others are stable (Figure 3A). The 2018 fire season, including the Camp Fire, may be responsible for increased PM concentrations and metals (Boaggio et al., 2022), explaining the large increases in biomass burning, combustion (OC), and combustion (EC) PM concentrations during that year.

Annually, total $PM_{2.5}$ concentrations increase in the summer and late fall. This is consistent with summer and fall increases in sources including: combustion (EC) and combustion (OC); secondary sulfate; and biomass burning, which is opposite to the wintertime maximum of secondary nitrate, which had the highest average weekly mean concentration (Table 1). The low correlation between source concentrations (all <0.5, Table S2) potentially indicates that the identified factors from PMF are indeed representing the range of unique real-world sources of $PM_{2.5}$, each created by unique dynamics.

We find that across the 35 sites in California an IQR increase of industrial, combustion (EC) and secondary sulfate $PM_{2.5}$ had the largest positive association with cardiorespiratory hospitalization rates. Previous studies that also identified sources similar to combustion (EC) (e.g., vehicular emissions) found this source associated with: increased emergency department visits for cardiovascular and respiratory diseases across eight cities in California (Ostro et al., 2016); cardiovascular mortality in the same eight cities (Berger et al., 2018); non-accidental mortality in Chile (Cakmak et al., 2009); and all-cause deaths, hospitalizations and emergency department visits in New York City (Kheirbek et al., 2016). Also, Secondary sulfate was previously found to have the greatest association (compared to soil, nitrate, traffic, etc.) with total mortality in Washington D.C. (Ito et al., 2006) and also in Phoenix, AZ (Mar et al., 2006). Interestingly, biomass burning—which in several previous studies was found to have the greatest association with mortality or morbidity (Berger et al., 2018; Heo et al., 2014; Pennington et al., 2019)—was negative in our study. Similarity in findings of source-specific associations and health outcomes with other studies (e.g., Combustion (EC) and secondary sulfate) supports the growing literature that demonstrates the differential exposure risk from some sources, while the differences (e.g., biomass burning) highlights the need for further advancement of methods and focus on how different populations are affected by specific PM sources. Particularly, metals (e.g., Zn, Cu) and organic compound constituents of PM are thought to generate oxidants and oxidative stress in the body, respectively (Ghio et al., 2012), leading to inflammation and cell death, which in turn causes disease or mortality. Continued focus on advancing methods to attribute PM composition to sources, will enable efforts to limit the emission or exposure to these particles, and thus decrease the burden of disease.

Effect modification by season shows that each source has a unique effect overtime on the population studied. For example, in the fall, when industrial and secondary mixture sources have generally a higher concentration, a significantly higher association with hospitalization is observed (Figure 5). However, for combustion (EC)—which similarly has peak concentrations in fall—the effect was considerably lower than spring or summer seasons. Similarly, secondary nitrate—with the highest average concentrations in winter—has a negative association in the winter season, indicating there is no categorial relationship between increased total concentrations and risk, per increase in IQR. Seasonal risk and $PM_{2.5}$ exposure may also vary by population behaviors, such as indoor heating in the wintertime and increased time indoors, which complicate source specific estimates of risk, overtime.

Of the seven main climates in California, there was sufficient data to assess the differential health risk in five of them (Figure 6). Interestingly, for the secondary sulfate source, in the driest climate (Arid, Hot) the change in hospitalization with an IQR increase was the most negative of all sources, whereas it was the highest in the wettest climate region (Temperate, warm summer). This difference between climate extreme is also seen for industrial PM, where the hospitalization rate per IQR is highest in the Arid, hot regions and lowest in the Temperate, warm summer. There was however not a linear relationship between aridity and all source-specific outcomes. For secondary nitrate, the largest hospitalization increase per IQR was in the Semi-arid zone, while there was a decrease in the previously mentioned wettest and driest regions. Some regions, such as semi-arid, hot and temperate, hot summer, had little variation in hospitalization rate variation between sources, while it was high for others. As with seasonal effect modification results, the differences in population behaviors between climate zones may cause the difference in hospitalizations, as opposed to inherent differences in source exposure risk. To assess this, other factors—such as climate variability, controls for exposure duration, and human behavior—would need to be considered.

Finally, effect modification by air basins shows variation source-specific $PM_{2.5}$ hospitalization rates, unique from climate zones. The most extreme variation in source-specific hospitalization rates was in the Mojave Desert air basin, with an IQR increase in secondary nitrate associated with a more than 10x increase in weekly hospitalizations—when compared to the average across all sites. Additionally, although dust concentrations were negatively associated with hospitalization overall, in five air basins (San Diego County, Sacramento Valley, San Joaquin Valley, San Francisco Bay, and South Central Coast) an increase in a dust IQR lead to an increase in hospitalization. Only in San Diego County was biomass burning concentrations positively associated with hospitalization. As air basins are regions generally separated by both climate zones and unique industries and urban densities (e.g., the Mojave Desert is arid with no large cities, the San Joaquin Valley is heavily dominated by agriculture, and South Coast has large population centers), this effect modification analysis may show how both climate and exposure type—mediated by occupation and lifestyle—affect health impacts of PM exposure.

Overall, we found specific associations between sources of $PM_{2.5}$ in different seasons, climate regions, and air basins across California that had not previously been observed. Prior studies in California have considered $PM_{2.5}$ data at multiple sites and emergency department visits (Ostro et al., 2016) or mortality rates (Berger et al., 2018), observing unique differences but not assessing the results as a function of geospatial variability. In light of the evidence of the compounding health impacts of high temperatures and $PM_{2.5}$ exposure (Rahman et al., 2022), we see a gap in the literature concerning how the health burden varies across climates, communities in unique parts of the state, and further, from exposure to unique types of $PM_{2.5}$.

This study demonstrates the heterogenous relationships between weekly $PM_{2.5}$ exposure and cardiorespiratory hospitalization across PM2.5 sources, seasons, climate zones, and air basins. Further work is needed to probe the relative significance of these factors in consideration of localized source emission variables and population dynamics. In particular, the inclusion of population effect modifiers would reveal who within the studied communities are most affected, and potential reveal information about the exposure pathways.

Previous studies have shown that some people are more susceptible to developing negative health outcomes from similar exposure than others (Bell et al., 2013; Sacks et al., 2011). Separately, associated research has found that some communities are exposed to disproportionately high concentrations of particulate matter, notably, minority, racial/ethnic groups, migrants, and lower economic groups (Geldsetzer et al., 2024; Liévanos, 2019; Ma et al., 2023). These two—often compounding issues—may be discussed as an environmental injustice, as disproportionate exposure to environmental pollution is often an outcome of systemic racism and economic marginalization, while additionally many of the communities who are disproportionately exposed to PM, are not the ones who created it (Tessum et al., 2019). This may be seen in the chosen locations of polluting industries, hazardous and toxic waste disposal and landfills (Tessum et al., 2021), and the use of toxic chemicals in agricultural settings (Donley et al., 2022). In the case of agriculture, PM exposure disproportionately impacts migrant workers, who are in an especially vulnerable population due to lack of legal, political, and social agency (Moyce & Schenker, 2018). Overall, there is significant evidence of PM exposure effect modification based upon sociodemographic factors, leading to a greater impact of exposure for some communities. Future research of speciated PM exposure risk in different climate regions should consider and investigate the impact on different PM sources on different communities.

Additionally, analysis of the role of temperature in modifying health outcomes from $PM_{2.5}$ exposure would reveal information about the current burden of disease and potential risks in future climate scenarios. Previous research in California has shown the effect of compounding high PM concentrations and high temperatures, finding that co-exposure leads to adverse outcomes that are more than the sum of individual effects (Rahman et al., 2022). Furthering this research by exploring if certain sources uniquely exacerbate hospitalizations, would provide new insights into the risk facing communities in different climates and conditions. Overall, we see the benefit of building off this study framework to pursue additional nuanced relationships and links between climate and population variables.

Some limitations should be noted. We assigned source-specific $PM_{2.5}$ to ZCTAs based on proxy to the monitoring sites (< 20 km), but did not consider variations in concentration within this distance.

Relying on monitoring sites limited our spatial coverage and ability to include all ZCTAs in California. In addition, our statistical model did not account for the complicated interactions among $PM_{2.5}$ from different sources. Future studies applying mixture analysis are warranted to better reflect the real-world exposure complexity. To make consistent comparisons between source-specific $PM_{2.5}$ concentrations and hospitalization records, we created weekly averages of daily results. This may have led to the loss of important and unique daily interaction and made us unable to capture more acute health effects occurred in a few days. Also, in including data from a diverse population, a potential source of error is the inconsistent access to or use of hospital care (Fiscella et al., 2000), which may lead to underestimation or relative overestimation of PM exposure impacts.

5 Conclusion

Analysis of the impact from exposure to different types of $PM_{2.5}$ reveals meaningful information about the global burden of disease and risk faced by communities in diverse regions. We used PMF to identify unique sources of $PM_{2.5}$ at 35 sites across California from 2007 to 2019 and assessed the association with weekly cardiovascular hospitalizations. Increases in $PM_{2.5}$ from industrial, secondary sulfate, secondary nitrate, combustion (OC), and combustion (EC) sources were associated with an increase in cardiorespiratory hospitalizations, whereas suggestive negative associations were found with biomass burning, secondary mixture, sea salt, and dust. These associations were modified seasonally and by geographic regions and climate. We identify a need for additional source-specific PM research to consider the interaction between climate and population differential susceptibility.

Acknowledgments

Chapter 3, in part, is currently being prepared for submission for publication of the material. Norris, Emmet; Ma, Yiqun; Benmarhnia, Tarik. The dissertation author was the primary researcher and author of this material.

Chapter 3 Appendix

Supplemental Table 3-1. Monitor site data, with overall and seasonal source-specific PM2.5 concentrations and study start date. (continued)

Site	Season	Start Date	Soil	Sea Salt	Industrial	Combustion OC	Combustion EC	Biomass	Secondary Mixture	Sulfate	Nitrate
Units	Season	yyyy-mm-dd	μg/m³	μg/m³	μg/m³	μg/m³	μg/m³	μg/m³	μg/m³	μg/m³	μg/m³
Agua	Overall	2006-01-08	1.1 (1.1)	1.2 (1.6)		1.1 (0.9)	1.1 (0.8)			1.1 (0.9)	1.2 (1.1)
	Spring	2006-01-08	1.3 (1.4)	1.8 (1.9)		1 (0.7)	0.9 (0.8)			1.1 (0.8)	1.4 (1.1)
	Summer	2006-01-08	1.1 (0.7)	1 (1.3)		1.6 (1.1)	1.1 (0.7)			1.8 (0.8)	1.1 (0.8)
	Fall	2006-01-08	1.1 (1.1)	0.9 (1.4)		1.1 (0.9)	1.3 (0.9)			0.9 (0.7)	1 (1)
	Winter	2006-01-08	0.6 (0.7)	0.9 (1.4)		0.6 (0.5)	1 (0.7)			0.3 (0.3)	1 (1.3)
Anaheim	Overall	2010-02-13	1.1 (1.2)	1.1 (0.9)	1.1 (1.4)	1.1 (1)		1.1 (0.9)		1.1 (0.9)	1.1 (1.7)
	Spring	2010-02-13	1 (1.1)	1.3 (0.9)	1.2 (1.3)	0.7 (0.7)		1 (1)		1 (0.8)	0.8 (1.1)
	Summer	2010-02-13	0.5 (0.5)	1.3 (1.1)	0.7 (0.9)	0.7 (0.5)		0.9 (0.6)		1.8 (0.8)	0.5 (0.5)
	Fall	2010-02-13	1.4 (1.4)	1 (0.7)	1.2 (1.5)	1.3 (1.1)		1.1 (0.8)		1 (0.9)	1.2 (1.7)
	Winter	2010-02-13	1.5 (1.2)	0.8 (0.6)	1.4 (1.7)	1.7 (1.2)		1.3 (1)		0.3 (0.3)	1.9 (2.3)
Bakersfield	Overall	2007-03-07	1.1 (1.1)		1 (1.1)	1.1 (0.9)		1 (1.1)	1 (1.2)	1 (0.6)	1.2 (1.9)
	Spring	2007-03-07	0.7 (0.7)		0.8 (1.1)	0.6 (0.5)		0.6 (0.7)	0.6 (0.9)	1 (0.6)	0.6 (0.7)
	Summer	2007-03-07	1.4 (0.9)		0.9 (1.2)	1 (0.9)		0.9 (1.1)	1.4 (1.3)	1.4 (0.5)	0.2 (0.2)
	Fall	2007-03-07	1.6 (1.5)		1.2 (1.1)	1.4 (1)		1.4 (1.3)	1.4 (1.3)	1 (0.6)	1.3 (1.9)
	Winter	2007-03-07	0.4 (0.6)		1.2 (0.9)	1.3 (0.9)		1.3 (1.1)	0.7 (0.7)	0.6 (0.6)	2.7 (2.7)
Bishop	Overall	2013-07-12	2 (1.8)				1.1 (2.5)	1 (1.2)		1 (0.7)	
	Spring	2013-07-12	2 (1.8)				0.3 (0.2)	0.7 (0.6)		1.1 (0.7)	
	Summer	2013-07-12	2.4 (1.5)				2.1 (3.1)	1.3 (1.2)		1.6 (0.7)	
	Fall	2013-07-12	2.4 (1.9)				1.4 (3.4)	1.3 (1.6)		0.9 (0.6)	
	Winter	2013-07-12	1.2 (1.5)				0.3 (0.3)	0.7 (0.8)		0.4 (0.4)	
Bliss	Overall	2006-01-05	1.1 (1.4)	1.1 (1.2)	1 (0.8)			1.1 (3.2)		1 (1)	1 (1.3)
	Spring	2006-01-05	2 (2.1)	1.1 (1.1)	0.8 (0.5)			0.4 (0.4)		1.5 (1.4)	1.3 (1.2)
	Summer	2006-01-05	1 (0.9)	1.8 (1.6)	1.5 (0.9)			1.9 (5.1)		1.3 (0.8)	1 (1.4)
	Fall	2006-01-05	0.8 (1)	0.8 (1)	1.2 (0.9)			1.4 (3.3)		0.9 (0.7)	1 (1.2)
	Winter	2006-01-05	0.4 (0.6)	0.5 (0.5)	0.5 (0.4)			0.6 (1.2)		0.4 (0.4)	0.8 (1.2)
Calexico	Overall	2009-03-02	1 (0.9)		1.3 (2.1)	1.1 (0.9)	1 (0.8)	1.1 (1.5)	1.4 (2.2)	1 (0.7)	1 (1.3)
	Spring	2009-03-02	0.9 (0.7)		0.5 (0.5)	1 (0.8)	0.7 (0.5)	0.7 (1.2)	1.3 (2.4)	1.1 (0.7)	0.7 (0.6)
	Summer	2009-03-02	0.9 (0.8)		1.2 (1.1)	0.7 (0.4)	0.7 (0.4)	2 (2.2)	2.1 (2.6)	1.5 (0.7)	0.5 (0.3)
	Fall	2009-03-02	1.4 (1.1)		0.7 (1.5)	1.1 (0.8)	1.1 (0.7)	0.7 (0.7)	1.2 (1.6)	1 (0.7)	0.7 (0.7)
	Winter	2009-03-02	0.8 (0.8)		2.7 (3.5)	1.5 (1)	1.6 (1.1)	0.9 (1.2)	0.9 (1.6)	0.6 (0.5)	2.2 (2.1)
Chico	Overall	2009-03-26	1.1 (1)		1.2 (1.1)	1.1 (1.3)	1.1 (1.3)	1.1 (0.6)	1.1 (1.5)	1 (0.7)	1.1 (1.8)
	Spring	2009-03-26	0.8 (0.8)		1 (1.1)	0.5 (0.4)	0.7 (1)	1 (0.4)	0.9 (1.1)	1.2 (0.8)	0.6 (0.7)
	Summer	2009-03-26	1.3 (0.7)		1.2 (1.1)	0.8 (0.9)	0.8 (0.9)	1 (0.5)	1.1 (0.8)	1.5 (0.7)	0.5 (0.5)
	Fall	2009-03-26	1.6 (1.4)		1.4 (1.1)	1.1 (1.2)	1.2 (1.1)	1.1 (0.6)	0.8 (1)	0.9 (0.6)	0.9 (1.2)
	Winter	2009-03-26	0.3 (0.3)		1.3 (1)	2 (1.7)	1.7 (1.8)	1.2 (0.9)	1.7 (2.6)	0.5 (0.3)	2.4 (3)
DomeLands	Overall	2006-01-02	1.1 (1)	1.1 (1.2)	1.1 (1.6)	1.1 (1.7)				1 (0.8)	1.1 (1.5)
	Spring	2006-01-02	1.1 (1)	1.3 (1.3)	0.3 (0.2)	0.7 (0.6)				1.1 (0.7)	1.6 (1.7)
	Summer	2006-01-02	1.5 (0.9)	1.6 (1.2)	2.3 (2.1)	1.7 (2.5)				1.7 (0.7)	0.6 (0.8)
	Fall	2006-01-02	1.2 (1.1)	1.1 (1.1)	1.4 (1.6)	1.3 (2.1)				1 (0.6)	1 (1.6)
	Winter	2006-01-02	0.3 (0.4)	0.5 (0.6)	0.2 (0.2)	0.7 (0.6)				0.4 (0.3)	1 (1.7)
ElCajon	Overall	2007-03-10	1.1 (1)	1.2 (1.2)	1.1 (0.9)	1.2 (1.7)		1.1 (1.1)	1.1 (0.6)	1.1 (1.1)	1.1 (1.4)
	Spring	2007-03-10	1 (0.9)	1.5 (1.3)	0.7 (0.5)	1.1 (1.3)		0.7 (0.5)	0.9 (0.6)	1 (1)	1.1 (1.3)
	Summer	2007-03-10	0.6 (0.5)	1.5 (1.2)	0.6 (0.4)	1 (1.5)		0.6 (0.6)	1.4 (0.7)	1.7 (1.1)	0.9 (1)
	Fall	2007-03-10	1.5 (1.2)	1 (1)	1.2 (0.9)	1.2 (1.6)		1.2 (1.1)	1.1 (0.6)	1.1 (1)	1.2 (1.7)
	Winter	2007-03-10	1.2 (1)	0.6 (0.7)	1.8 (1)	1.5 (2.3)		1.8 (1.4)	0.8 (0.5)	0.4 (0.5)	1.3 (1.7)
Escondido	Overall	2007-03-07	1 (0.8)	1.2 (2.1)		1.1 (1)		1 (0.5)	1.1 (0.6)	1.1 (0.9)	1.1 (1.3)
	Spring	2007-03-07	1 (0.6)	2.2 (3)		0.6 (0.5)		0.9 (0.2)	1.1 (0.7)	1.1 (0.8)	1 (1.2)
	Summer	2007-03-07	0.9 (0.4)	0.7 (1.1)		0.4 (0.3)		1 (0.2)	1.4 (0.6)	1.7 (0.9)	0.8 (0.7)
	Fall	2007-03-07	1.3 (1)	0.7 (1.6)		1.3 (1)		0.9 (0.3)	1.1 (0.6)	1 (0.9)	1.1 (1.6)
	Winter	2007-03-07	0.8 (0.9)	1.2 (1.9)		2 (1.1)		1.3 (0.9)	0.6 (0.5)	0.3 (0.4)	1.3 (1.5)
Fresno	Overall	2009-03-23	1.1 (1.2)	1.2 (1.2)	1.1 (1.8)	1.1 (1)		1.1 (1.2)	1.1 (1)	1 (0.7)	1.2 (2)
	Spring	2009-03-23	0.7 (0.6)	1.2 (1.3)	0.7 (1.4)	0.5 (0.4)		0.6 (0.7)	0.6 (0.5)	1.1 (0.7)	0.4 (0.5)
	Summer	2009-03-23	1.3 (0.7)	0.9 (1.3)	0.4 (1.1)	0.8 (0.9)		0.8 (0.9)	0.5 (0.3)	1.4 (0.6)	0.2 (0.2)
	Fall	2009-03-23	1.9 (1.6)	1 (1)	1.3 (1.9)	1.4 (1.2)		1.3 (1.4)	1.4 (1)	1 (0.6)	1.2 (1.7)
	Winter	2009-03-23	0.3 (0.5)	1.6 (1)	1.8 (2.4)	1.6 (1.1)		1.6 (1.3)	1.7 (1.1)	0.6 (0.6)	2.8 (2.9)

Supplemental Table 3-2. Monitor site data, with overall and seasonal source-specific PM2.5 concentrations and study start date. (continued)

Gorgonio	Overall	2006-01-02	1.1 (1.3)		1.1 (0.8)	1.1 (1.3)	1 (1)			1 (0.8)	
	Spring	2006-01-02	1.8 (1.9)		1.1 (0.7)	1.8 (1.5)	1.1 (1)			1.1 (0.7)	
	Summer	2006-01-02	1 (1.1)		1.7 (0.6)	0.9 (0.9)	1.5 (1.1)			1.7 (0.8)	
	Fall	2006-01-02	0.8 (0.9)		1 (0.6)	0.9 (1.2)	1 (0.9)			0.9 (0.7)	
	Winter	2006-01-02	0.6 (0.8)		0.3 (0.4)	0.6 (1)	0.5 (0.5)			0.3 (0.3)	
Hoover	Overall	2006-01-02	1 (1)		1 (1.6)	1 (2.3)		1 (1.4)		1 (0.8)	1 (1.2)
	Spring	2006-01-02	1.5 (1.4)		1.9 (2.3)	0.4 (0.3)		0.9 (0.8)		1.2 (0.9)	1.2 (1.1)
	Summer	2006-01-02	1.2 (0.7)		0.9 (1.1)	2.2 (3.2)		1.7 (2)		1.4 (0.7)	1.1 (1.3)
	Fall	2006-01-02	0.9 (0.9)		0.7 (0.9)	1.2 (2.5)		1 (1.3)		0.9 (0.7)	0.9 (1.1)
	Winter	2006-01-02	0.4 (0.5)		0.5 (1.1)	0.2 (0.2)		0.3 (0.3)		0.4 (0.3)	0.8 (1.2)
Joshua	Overall	2006-01-02	1.1 (1.2)	1.1 (0.9)		1.4 (39.3)	1 (0.9)	1 (1.5)		1 (0.8)	1.1 (1.3)
	Spring	2006-01-02	1.3 (1.3)	1.3 (1)		0.2 (0.2)	1 (0.7)	0.7 (0.8)		1.1 (0.7)	1.5 (1.4)
	Summer	2006-01-02	1.2 (1.4)	1.4 (1)		3.7 (66.8)	1.4 (1)	2 (2.2)		1.7 (0.8)	0.8 (0.6)
	Fall	2006-01-02	1 (1.2)	0.9 (0.9)		0.3 (0.4)	1 (0.9)	0.9 (1.4)		0.9 (0.7)	0.8 (1.1)
	Winter	2006-01-02	0.5 (0.6)	0.8 (0.7)		0.1 (0.1)	0.7 (0.6)	0.5 (0.6)		0.3 (0.3)	1.3 (1.8)
Lebec	Overall	2006-01-02	1.1 (1)			1 (1.4)	1 (1.1)	1 (1.1)		1 (0.8)	1.1 (1.7)
	Spring	2006-01-02	1 (0.9)			0.7 (0.5)	0.7 (0.5)	0.9 (0.7)		1.3 (0.8)	1.4 (1.6)
	Summer	2006-01-02	1.6 (0.9)			1.7 (1.7)	1.5 (1.3)	1.7 (1.2)		1.6 (0.6)	0.6 (0.6)
	Fall	2006-01-02	1.2 (1.1)			1.1 (1.9)	1.2 (1.5)	1.1 (1.2)		0.8 (0.5)	1.1 (2)
	Winter	2006-01-02	0.3 (0.4)			0.5 (0.3)	0.6 (0.4)	0.4 (0.5)		0.3 (0.3)	1.1 (2.1)
LosAngeles	Overall	2007-03-07	1.1 (1)	1.1 (0.9)	1 (1)	1.1 (1.1)				1.1 (0.8)	1 (0.8)
	Spring	2007-03-07	1.3 (1)	0.7 (0.6)	1.1 (0.9)	1.1 (1.1)				0.9 (0.7)	0.7 (0.5)
	Summer	2007-03-07	1.5 (1.1)	0.8 (0.7)	0.7 (0.6)	1.7 (1.1)				1 (0.7)	0.6 (0.4)
	Fall	2007-03-07	1 (0.9)	1.4 (1)	1.3 (1.2)	1 (1)				1.1 (0.9)	1.3 (0.9)
	Winter	2007-03-07	0.5 (0.5)	1.5 (0.9)	1 (0.9)	0.3 (0.6)				1.2 (0.8)	1.5 (0.9)
Pinnacles	Overall	2006-01-08	1.1 (1.2)	2.2 (2.7)	1.1 (0.8)	1.1 (1.4)				1 (1)	1.1 (1.1)
	Spring	2006-01-08	1.3 (1.4)	2.9 (3.1)	0.9 (0.6)	0.9 (0.8)				1.2 (1)	1.1 (0.9)
	Summer	2006-01-08	0.9 (0.6)	2.6 (2.9)	1.1 (0.8)	0.9 (1.8)				1.5 (0.9)	0.7 (0.9)
	Fall	2006-01-08	1.5 (1.5)	1.7 (2.2)	1.5 (1)	1.3 (1.6)				1 (1)	1.2 (1.2)
	Winter	2006-01-08	0.5 (0.6)	1.4 (2)	0.8 (0.6)	1.2 (0.9)				0.4 (0.4)	1.3 (1.2)
PointReyes	Overall	2006-01-02	1.1 (1.6)	1.1 (0.9)		1.1 (1.4)	1.1 (2.1)			1 (0.8)	1.1 (2.3)
	Spring	2006-01-02	1.5 (1.7)	1.2 (0.9)		0.6 (0.5)	0.5 (0.5)			1.1 (0.7)	0.5 (0.6)
	Summer	2006-01-02	0.4 (0.6)	1.2 (1.1)		0.9 (1.3)	0.5 (1)			1.3 (0.7)	0.4 (0.5)
	Fall	2006-01-02	1.4 (2)	0.9 (0.8)		1.6 (2.1)	1.6 (3.3)			1 (0.7)	1 (1.6)
	Winter	2006-01-02	1.1 (1.4)	0.8 (0.7)		1.2 (1.1)	1.8 (2.2)			0.6 (0.6)	2.6 (4.1)
Portola	Overall	2009-04-01	1.1 (1.1)		1.2 (1.2)	1.1 (1.1)		1 (0.6)	1 (0.7)	1 (0.9)	1 (0.8)
	Spring	2009-04-01	1.2 (1.4)		1.3 (1.4)	0.8 (0.7)		0.9 (0.5)	1.2 (0.8)	1.1 (1)	1 (0.7)
	Summer	2009-04-01	1.4 (1)		1.1 (1.3)	0.2 (0.3)		0.8 (0.4)	1.3 (0.6)	1.3 (0.9)	0.6 (0.5)
	Fall	2009-04-01	1 (0.9)		1.1 (1)	1.2 (1.1)		1 (0.5)	1 (0.6)	0.9 (0.7)	0.9 (0.6)
	Winter	2009-04-01	0.6 (0.9)		1.3 (1.2)	2.1 (1)		1.4 (0.7)	0.6 (0.4)	0.8 (0.9)	1.6 (1.1)
Redwood	Overall	2006-01-02	2 (1.9)	1 (1.1)		1.1 (2)	1 (1.9)			1 (0.9)	1 (0.9)
	Spring	2006-01-02	2.8 (2.3)	1.2 (1.1)		0.5 (0.4)	0.6 (0.6)			1.2 (1)	1.1 (0.9)
	Summer	2006-01-02	1.9 (2.1)	1.2 (1.4)		1.1 (2.1)	0.8 (1.3)			1.5 (1)	1.1 (1)
	Fall	2006-01-02	1.8 (1.5)	0.9 (0.9)		1.9 (3)	1.8 (3.3)			0.9 (0.7)	1.1 (1)
	Winter	2006-01-02	1.5 (1.3)	0.8 (0.8)		0.7 (0.8)	0.8 (1)			0.4 (0.4)	0.7 (0.7)
Rubidoux	Overall	2007-03-07	1 (1)	1.2 (1.2)		1.2 (0.9)	1 (0.9)	1.1 (1.1)		1.1 (1)	1.2 (1.4)
	Spring	2007-03-07	1 (0.9)	1.5 (1.2)		0.9 (0.7)	0.7 (0.6)	1.1 (1.4)		1.1 (0.9)	1.2 (1.3)
	Summer	2007-03-07	0.9 (0.6)	1.6 (1.3)		1.4 (0.9)	0.6 (0.4)	1 (1)		1.8 (0.9)	1 (1)
	Fall	2007-03-07	1.5 (1.4)	0.9 (1)		1.3 (1)	1.4 (1)	1.1 (1)		1 (0.8)	1.3 (1.7)
	Winter	2007-03-07	0.7 (0.7)	0.7 (0.6)		1.2 (1)	1.5 (1)	1.2 (1)		0.3 (0.3)	1.1 (1.3)
SacramentoT	Overall	2009-03-26	1.1 (1.1)	1.1 (0.8)	1.1 (1)	1.1 (1.1)	1 (0.8)	1 (0.6)		1.1 (0.8)	1.2 (2.2)
	Spring	2009-03-26	1.1 (1)	1.1 (0.7)	1 (0.9)	0.5 (0.6)	0.8 (0.5)	1 (0.4)		1 (0.8)	0.3 (0.4)
	Summer	2009-03-26	0.9 (0.7)	1.5 (1)	0.9 (0.9)	0.6 (0.6)	0.6 (0.4)	1.1 (0.4)		1.3 (0.8)	0.2 (0.2)
	Fall	2009-03-26	1.7 (1.5)	1 (0.7)	1.2 (1.1)	1.5 (1.2)	1.3 (0.9)	1 (0.8)		1.1 (0.7)	0.9 (1.5)
	Winter	2009-03-26	0.5 (0.6)	0.7 (0.4)	1.3 (1.1)	1.7 (1.3)	1.4 (0.9)	1 (0.6)		1 (0.8)	2.9 (3.2)

Supplemental Table 3-3. Monitor site data, with overall and seasonal source-specific PM2.5 concentrations and study start date. (continued)

SanJose	Overall	2009-03-26	1 (1.3)	1.1 (0.8)		1.1 (1.1)	1.1 (0.9)		1.1 (1.2)		1 (0.9)
	Spring	2009-03-26	0.7 (0.4)	0.7 (0.5)		1.4 (1.2)	0.9 (0.8)		0.5 (0.4)		1 (0.7)
	Summer	2009-03-26	0.9 (1.3)	0.7 (0.4)		1.6 (1.4)	1.5 (0.8)		0.4 (0.4)		0.6 (0.5)
	Fall	2009-03-26	1.4 (1.9)	1.2 (0.8)		0.8 (0.8)	1.1 (0.9)		1.4 (1.3)		1.4 (1.2)
	Winter	2009-03-26	1.2 (1)	1.6 (1.2)		0.5 (0.5)	0.6 (0.6)		1.9 (1.5)		1.1 (0.9)
SanRafael	Overall	2006-01-02	2 (1.6)	1 (1.5)		1 (1.5)	1 (2)			1 (0.8)	1 (1)
	Spring	2006-01-02	2.6 (1.8)	1.7 (1.9)		0.7 (0.5)	0.7 (0.5)			1.1 (0.8)	1.3 (1)
	Summer	2006-01-02	2.6 (1.3)	1.2 (1.6)		1.8 (2.3)	1.6 (3)			1.6 (0.8)	1.1 (0.9)
	Fall	2006-01-02	2 (1.5)	0.6 (1)		1.1 (1.5)	1.2 (2.3)			0.9 (0.7)	0.9 (0.9)
	Winter	2006-01-02	0.9 (0.8)	0.6 (1)		0.4 (0.7)	0.5 (0.6)			0.4 (0.3)	0.8 (1.1)
Sequoia	Overall	2006-01-02	1.1 (1.1)	1.1 (0.8)	1.1 (1.1)	1 (1.3)	1 (1.2)			1 (0.8)	1.2 (2.2)
	Spring	2006-01-02	0.8 (0.8)	1.4 (0.9)	0.9 (1)	0.7 (0.5)	0.7 (0.4)			1 (0.7)	0.7 (0.9)
	Summer	2006-01-02	1.6 (0.9)	1.3 (0.9)	1.1 (1.1)	1.5 (1.5)	1.4 (1.3)			1.7 (0.7)	0.3 (0.3)
	Fall	2006-01-02	1.6 (1.4)	0.8 (0.7)	1.3 (1.2)	1.4 (1.9)	1.4 (1.8)			0.9 (0.6)	1.5 (2.7)
	Winter	2006-01-02	0.2 (0.3)	0.6 (0.5)	0.9 (1)	0.5 (0.3)	0.5 (0.4)			0.4 (0.4)	2.1 (3)
SimiValley	Overall	2009-03-05	1.1 (1.1)	1.1 (1.1)		1.1 (0.7)		1.2 (1.1)		1.1 (1.1)	1 (1.2)
	Spring	2009-03-05	1.4 (1.3)	1.6 (1.4)		0.8 (0.5)		1 (1)		1.2 (1.3)	1.1 (1.2)
	Summer	2009-03-05	0.7 (0.6)	1.4 (1.1)		1.2 (0.4)		0.9 (1)		1.7 (0.9)	1.1 (1)
	Fall	2009-03-05	1 (0.8)	0.8 (0.8)		1.2 (0.8)		1.1 (0.9)		1 (0.9)	0.9 (1.2)
	Winter	2009-03-05	1 (1.4)	0.4 (0.4)		1 (0.9)		1.6 (1.3)		0.3 (0.4)	1 (1.4)
Trinity	Overall	2006-01-02	2 (2.5)	1 (1.4)		1.1 (4)	1 (3.6)	1.1 (2.3)		1 (0.9)	
	Spring	2006-01-02	2.9 (3.1)	1.3 (1.6)		0.2 (0.2)	0.3 (0.3)	0.6 (0.6)		1.2 (1)	
	Summer	2006-01-02	2.5 (1.9)	0.9 (1.1)		2.3 (6.5)	1.8 (5.4)	1.7 (3.4)		1.4 (0.8)	
	Fall	2006-01-02	2.1 (2.6)	0.9 (1.4)		1.5 (3.9)	1.8 (4.7)	1.5 (2.9)		1 (0.7)	
	Winter	2006-01-02	0.5 (0.7)	1 (1.5)		0.2 (0.3)	0.4 (0.4)	0.4 (0.6)		0.4 (0.3)	
Visalia	Overall	2007-03-13	1.2 (1.1)		1.1 (1.4)	1.1 (0.9)	1 (0.6)	1 (0.9)		1 (0.6)	1.1 (1.5)
	Spring	2007-03-13	0.8 (0.7)		0.7 (0.7)	0.6 (0.5)	0.7 (0.4)	0.8 (0.4)		1 (0.6)	0.6 (0.5)
	Summer	2007-03-13	1.5 (1)		1.2 (1.5)	1.1 (1)	0.9 (0.4)	1 (1)		1.5 (0.5)	0.2 (0.2)
	Fall	2007-03-13	1.8 (1.5)		1.3 (1.3)	1.2 (0.9)	1.3 (0.7)	0.9 (0.9)		0.9 (0.5)	1.3 (1.6)
	Winter	2007-03-13	0.4 (0.4)		1.4 (1.9)	1.4 (1)	1.3 (0.8)	1.4 (1.1)		0.6 (0.5)	2.2 (2.1)
Yosemite	Overall	2006-01-05	2 (2.1)	1 (0.8)		1 (3)	1 (1.9)	1 (2)			1 (1.7)
	Spring	2006-01-05	2.7 (2.8)	1.3 (0.9)		0.3 (0.2)	0.5 (0.4)	0.6 (0.5)			1.1 (1.1)
	Summer	2006-01-05	2.5 (1.3)	1.6 (0.6)		1.9 (4.3)	1.6 (2.7)	1.9 (3.3)			1 (1.4)
	Fall	2006-01-05	2.1 (2.2)	0.9 (0.6)		1.6 (3.7)	1.5 (2.4)	1.1 (1.8)			1.1 (1.9)
	Winter	2006-01-05	0.6 (0.9)	0.3 (0.3)		0.2 (0.2)	0.3 (0.4)	0.3 (0.3)			0.8 (2.1)

Supplemental Table 3-4. Correlation table for source-specific PM2.5 variables

$PM_{2.5}$ sources	Combustion (OC)	Combustion (EC)	Secondary nitrate	Secondary Sulfate	Sea salt	Dust	Biomass	Industrial	Secondary mixture
Vehicular (OC)	1.00	0.46	0.28	0.20	-0.02	0.16	0.36	0.14	0.05
Vehicular (EC)	0.46	1.00	0.41	0.12	-0.09	0.18	0.32	0.26	-0.05
Secondary nitrate	0.28	0.41	1.00	0.29	0.15	-0.07	0.32	0.24	0.25
Secondary sulfate	0.20	0.12	0.29	1.00	0.27	-0.03	0.10	-0.01	0.06
Sea salt	-0.02	-0.09	0.15	0.27	1.00	0.09	0.01	0.08	0.41
Soil	0.16	0.18	-0.07	-0.03	0.09	1.00	0.01	0.10	0.22
Biomass	0.36	0.32	0.32	0.10	0.01	0.01	1.00	0.23	0.22
C-metals	0.14	0.26	0.24	-0.01	0.08	0.10	0.23	1.00	0.30
Secondary mixture	0.05	-0.05	0.25	0.06	0.41	0.22	0.22	0.30	1.00

Note: Pearson correlation coefficients were used to measure the correlation among variables.

Chapter 3 References

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Chapter 4 Dust accountability: discussion on multi-scale frameworks to research harms

1. Introduction and Overview

The burden of particulate matter in the atmosphere has been substantially altered by human land use trends, mostly associated with European colonization and capitalist extraction (e.g., Holleman, 2017; McNeill & Vrtis, 2017; Schneider et al., 2022), and these changes have modified key Earth system functions, especially during the past 300 years (Chen et al., 2018; Lambin & Geist, 2008). Widespread land use changes represent new dust sources, such as associated with deforestation for soybean farms in the Amazon Rainforest (Trivino et al., 2024), mountain-top removal coal mining in Appalachia (Aneja et al., 2017), explosive rubblization of urban areas in Homs, Syria or Gaza City, Palestine (Abuawad et al., 2024; Miškinytė, 2024), desiccation of the Aral Sea via water withdrawal for irrigation (Gaybullaev et al., 2012), and urbanization in Southern California (Chen, et al., 2018; Heald & Spracklen, 2015). Although different land uses and technologies interact with specific ecosystems and create distinct types of dust, many are connected through their origins in colonial projects to acquire land for extraction and short-term economic and cultural benefit. Can these disparate land use change processes be represented in a common framework? If so, what would this framework include, and what questions could be asked using it? In the first three chapters of this dissertation, I used trans-disciplinary methods to investigate the characteristics of dust and public health risks from exposure. This led to new findings about unique harms associated with specific sources of dust, and advancement of methods to quantify these relationships. With these methods and results, I consider the contributions and limitations of this approach. In this closing chapter, I consider how including frameworks that focus on the characterization of long-term and systemic dynamics contributes vital information to comprehensively understand the burden of global colonial land use.

The largest single risk factor for human death is environmental pollution, to which airborne particulate matter is the largest contributor (Fuller et al., 2022). Mitigating the effects of environmental pollution and particulate matter (PM) arising from land use changes generally is accomplished using regulations and improvement of technology, aimed at limiting the generation of and exposure to PM.

Regulations establish limits of acceptable exposure that (1) often are based on universalizing and peripherally relevant scientific studies; (2) are economically and technologically feasible; (3) enable the continuation of exploitation and extraction associated with land use change; and (4) stabilize unjust systems (Liboiron, 2021). In most cases, regulations are created in response to resistance from communities experiencing a public health crisis; these communities analyze and expose injustices and propose locally relevant solutions (Bullard & Johnson, 2000), which generally address the specific injustice, and not systemic issues (Sarine, 2020). Differential impacts and susceptibility of people and ecosystems to dust exposure has inhibited widespread action to reduce the critical health burden of PM exposure (Adeola, 2012; Hunnicutt & Henderson, 2022), as communities with less political and economic agency often are more negatively affected by PM than privileged communities that disproportionately contribute to or benefit from dust emissions (Tessum et al., 2019).

Many academic studies focus predominantly on short time scales to attribute harm to individual actors, and do not directly address the relationships between continuous short-term harms and the systems that create them. The diversity of academic fields and communities grappling with PM and dust emissions reflects the heterogeneous, multi-faceted character of the impact and interaction of dust with society and culture. Methods generally focus on one time scale, such as a short-time-scale study of the sources of PM impacting a community (e.g., asthma diagnosis in Imperial Valley youth, Farzan et al., 2019), in contrast with long-time-scale analyses (e.g., how the colonization of Southern California has led to extensive land-surface and water manipulation, causing ecological instability and generation of toxic dust with intergenerational health impacts, Voyles, 2021). Incorporating multiple scales into a study reveals how behaviors that lead to short-term stability—such as food production and economic benefit—may cause instability on long time scales, such as the displacement of Indigenous communities and investment in extractive land uses. Incorporating multiple scales allows for a more comprehensive understanding of complex causes and effects.

As shown in previous chapters, epidemiological investigations use multiple tools to study the health of populations as a function of varying levels of PM exposure, producing a detailed analysis of the burden of disease attributable to PM on short time scales (Davidson et al., 2005; Kim et al., 2015). Separately, study of dust emission, transport, and deposition in Earth science fields has highlighted the role of modern human activity in manipulating "natural" dust cycles, and introducing "anthropogenic" particles with a range of consequences (Chen et al., 2018; Conway et al., 2019; Duan et al., 2023), as demonstrated in Chapter 1. Tracing the pathways of these particles (termed dust, or particulate matter when affecting human health) from source to sink is accomplished through multiple Earth science techniques, and leads to information about the origins of particles, potential harms associated with specific sources, and ways to mitigate health impacts. This research often employs environmental records, and studies how long-time-scale systems are changed by short-time-scale behaviors (e.g., the impact of mining as seen from a sediment core, Rúa et al., 2014). At a slightly longer time scale, the study of environmental injustices, specifically through the Critical Environmental Justice (CEJ) framework, analyzes the systems of interconnected social injustices (e.g., evidenced through epidemiological research) that produce environmental pollution (e.g., toxic PM from petrochemical refineries) and complex environmental injustices (Pellow, 2017). With a wider systemic view focusing on longer time scales, colonial and anti-colonial studies by Indigenous and non-Indigenous scholars address the ongoing harm of colonization and settler-colonial production of PM and dust, including the academic stabilization of colonial land relationships that actively displace Indigenous land uses and relationships (Carlson, 2019; Liboiron, 2021; Smith, 2021; Whyte, 2016). Each of these frameworks, which analyze the systems and impacts of PM and dust, focus on different aspects and scales with distinct perspectives and goals. To explicitly explore how modern land uses perpetuate systems rooted in genocide and short-term economic and cultural gain at the expensive of Earth (including humans), I argue for the inclusion of multiple frameworks in the study of PM and dust emissions. Such a research approach would foreground multi-scale behaviors and causes of harms that are commonly not considered or left as suggestions for other researchers to explore.

In this chapter, I review findings from the first three chapters of my dissertation, and evaluate the efficacy of methods to study the burden of air pollution on humans and Earth, emphasizing the importance of the underlying relationships between people and land that both impact and are impacted by dust and PM. Inspired by accounting systems developed for virtual water trading and water grabbing (Dalin et al., 2012; Rulli et al., 2013), I discuss methods of associating virtual dust fluxes with the beneficiaries of land-use processes. In the first section, I explore how settler colonialism, characterized by colonial land uses and land relations—including genocide and land dispossession—is consistent with dust and PM production through land-surface destabilization and extraction. In the second section, I review different disciplines through discussing their focus on various time scales and tools. In the third section, using a complex systems science approach, I propose the use of a multi-scale study framework that integrates systemic perspectives from anti-colonial methodologies and Critical Environmental Justice, with quantitative data analysis from geochemistry and epidemiology. In the final section, I discuss insights from each of the frameworks in Imperial County, CA, addressing the history of colonization, land use and public health crises related to toxic dust transport. Specifically, I attempt to connect modern PM emissions—and associated harms—to long-time scale dynamics that are responsible for harmful land use activities, with the purpose of creating a web of accountability and in support of further inquiry. Development of the proposed method described here, and its application, are the subject of ongoing research and refinement.

2. Heterogenous dust production and land surface destabilization

2.1 Dust connects physical and biological Earth systems at global scales

The global dust cycle (Shao et al., 2011) exists in a sensitive equilibrium as part of greater Earth-system functions, contributing life sustaining nutrients to ecosystems (e.g., the Amazon Rainforest) and playing climate-sustaining roles (e.g., scattering or absorbing solar radiation and fertilizing phytoplankton). Recently, intensive manipulation of land surfaces by human technology (Vitousek et al., 1997) has significantly altered the distribution of dust sources and the total amount of dust in the atmosphere (Huang et al., 2014; Tegen & Fung, 1995). These short-time-scale changes to dust sources from human activity, including indirect sources (e.g., cropland) and direct sources (e.g., industrial emissions or construction) has caused a 19% net increase in the total amount of dust in the atmosphere

(Chen, et al., 2018), though some estimates indicate the increase could be as high as 60% (Mahowald & Luo, 2003). While different in scale, seasonality and location (e.g., croplands contribute 75% of indirect anthropogenic dust; Chen et al., 2018), these recent land use changes combine to significantly alter what global dust is made of, where it comes from, and how it affects life on Earth. Dust particles from industrial and urban sources are often quite different in chemical composition than dust emitted from crustal sources, and are characterized by high concentrations of the following components: (1) heavy metals (e.g., Hg, Pb, Zn, Cr, Cu from fossil fuel combustion, mining and industrial activities like metal refinement); (2) carcinogenic polycyclic aromatic hydrocarbons (PAHs) and complex carbonaceous compounds (largely from fossil fuel combustion and biomass burning); (3) microplastics (from industrial and urban environments, and the disposal of plastic products); (4) ammonium, sulfates and pesticides from agricultural fields; and (5) an array of ecologically harmful and highly engineered compounds like "forever chemicals" (e.g., PFASs) from various industries (Arslan & Aybek, 2012; Brahney et al., 2021; Chen et al., 2023; Faust, 2023; Patra et al., 2016). Additionally, the accumulation of toxic chemicals in terminal basins (e.g., lakes) from a mixture of long-term industrial and agricultural land uses creates uniquely harmful types of dust (Doede & DeGuzman, 2020; Putman et al., 2022).

Several types of land use are overwhelmingly responsible for disturbing soils and emitting particles, namely: industrial agriculture (Tegen et al., 2004; Guatam et al., 2024); urbanization (Xia et al., 2021); mining (Kasongo et al., 2024; Csavina et al., 2012); other types of industrial activities (Huang et al., 2014); and water use and the manipulation of rivers for the benefit of the aforementioned land uses (Alla and Liu, 2021). The creation and dissemination of land use technologies (Ax et al., 2014; Gann & Duignan, 1969; Vainer, 2014) and cultural land relationships have played pivotal roles in creating health and ecological impacts and damage. Altogether, emission of particles from these dust sources have led to a massive burden of disease and ecological harm (Fuller et al., 2022; Liu et al., 2019). This burden of disease from exposure to particles is unequally distributed, notably along racial and socioeconomic divides (Alvarez, 2023; Bell & Ebisu, 2012; Fairburn et al., 2019; Liévanos, 2019; Ma et al., 2023), and continues a history of environmental racism, systemic marginalization and social inequality, and the treatment of some people as dispensable (Pellow, 2017). Particularly acute examples are in Canada's so-called "Chemical Valley" (Bagelman & Wiebe, 2017; Wiebe, 2016) and the U.S.'s "Cancer Alley" (Martin, 2023), where a high density of petrochemical industries have created uniquely toxic environments that disproportionately impact Indigenous, Black and Brown, and other marginalized communities. Indigenous communities globally face large health burdens from PM and environmental pollution from mining, agriculture, urbanization, waste disposal and energy production (Fernández-Llamazares et al., 2020; Zanetta-Colombo et al., 2022), with global trends of decreased PM in the last decades not occurring to the same degree in Indigenous communities as compared to non-Indigenous communities (Kirby-McGregor et al., 2023; Li et al., 2022). Additionally, though of a slightly different nature, during colonization it is estimated that 95% of the more than 100 million Indigenous people of the Americas prior to European colonization were killed by airborne infectious disease and violence (Li et al., 2021). Overall, air pollution and associated inequality perpetuate the history of genocide, dispossession and displacement of Indigenous communities (Weisiger, 2023).

While global societies began to manipulate Earth's land-surface in a large-scale and ecologically meaningful way roughly 10,000 years ago in the Neolithic Revolution (Steffen et al., 2007), these manipulations were substantially different in scale and extent from the land use and land-surface transformation that began in the Industrial Revolution (Lambin et al., 2006). The Industrial Revolution is characterized by reliance on fossil fuel combustion and fast-paced and intensive land-surface transformation through globalized trade, which stemmed from European colonization of large parts of Earth (Takács-Sánta, 2004). Presently, it is estimated that up to 95% of all ice-free land on Earth has been transformed by land use (Ellis, 2021), with a 550% increase in cropland and 660% increase in pastureland in the past 300 years (Goldewijk, 2001). By 1914, it is estimated that Europeans had colonized 84% of terrestrial Earth (Hoffman, 2015). As seen throughout the past 400 years, the expansion of colonial powers to occupy and manipulate diverse ecosystems for economic and social benefit entailed the implementation of new land use technologies and the exploitation of global land. Examples include the transformation of diverse geographic areas in the Americas, Pacific Islands and South East Asia into cash-crop plantations in the 1800s and 1900s, such as: the United Fruit Company in Central America and the Caribbean (Marton and Lehoczki, 2024); US imperialism and Dole pineapple plantations in Hawai'i (Nasser, 2020); Spanish and US colonial sugar, tobacco, coffee and indigo production in Cuba, The Philippines and Puerto Rico (Burtchardt and Leinus, 2022); and the 1930s Dust Bowl, stemming from the convergence of large-scale colonizing agricultural projects and drought across the Great Plains (Holleman, 2018). In each of these cases, a combination of colonialism (e.g., colonial nation-led political campaigns and violence, and private corporate and settler occupation of land) and agricultural technologies (e.g., monoculture, pesticide use, replacement of local flora with non-native crops) resulted in land-surface damage and ecological harm.

The fast-paced expansion of European land use technologies and relationships to all inhabited continents in the past 400 years (Vertovec & Posey, 2004), can be contrasted with land use and land relationships of Indigenous communities living in regions prior to colonization through long-term environmental records. One scientific methodology to show the difference in land-use-related particle emissions is lacustrine (lake), glacial ice, and marine sediment cores that preserve changes in deposited particles and sediment in basins over time, which span the period of colonization in the region. Examples include: Tasmania, where lake cores spanning several hundred years show peak heavy metal concentrations following the onset of British mining activities (Schneider et al., 2022); a 1,000 year coastal record in the Darién Gulf showing increases in mercury and total sedimentation corresponding to Spanish gold and silver mining (Rúa et al., 2014); sediment cores from the Tapajós River in the Amazon, where fine particulates and soil erosion sharply increase in the 1950s-1970s, corresponding to massive deforestation and agriculture during Brazilian colonization (Roulet et al., 2000). A ~300 year ice core record in Wyoming, US, documenting far travelled heavy metal and black carbon pollution from the Industrial Revolution (Aarons et al., 2016; Chellman et al., 2017); and a 5,000 year record from alpine lakes in the western US that shows a 500% increase in dust transport in the 19th century following "westward expansion", including agricultural and grazing practices (Neff et al., 2008). A review of 25 global sediment cores finds the total amount of dust emitted in response to human activity more than doubled after 1750 CE, the beginning of the Industrial Revolution (Hooper & Marx, 2018).

These records are proxies for PM emission and their impact over time and are not comprehensive—nor can they be extrapolated to all forms and occurrences of colonization. However, they indicate the importance of considering the impact of colonial land use when researching specific geographic regions and current land-use-related PM emission concerns. While the focus of my work is on land use relationships, it is important to note that genocide, slavery, racism, colonial education campaigns, and methodological efforts to destroy Indigenous relationships to land, occurred as part of these and other colonial projects (Wolfe, 2001). Despite the centrality of these processes for strengthening colonial power and repressing resistance, here I will focus on colonial land use changes, because of their strong impacts on dust emissions.

2.3 Colonizing Dynamics

The direct relationship between colonization and land uses, which has resulted in multi-scale social and environmental harms from dust production and land-surface destabilization, indicates that focus on the systemic dynamics of colonialism could meaningfully support attempts to address the issues arising from them. Scholar Max Liboiron has characterized the colonial view of land as a resource for exploitation, as *colonial land relations*, meaning the "*assumed access by settler and colonial projects to Indigenous lands for settler and colonial goals*" (Liboiron, 2021). This framing stresses how colonial land uses often remove and replace existing land relationships, created over long time periods by Indigenous communities. Colonialism employs processes of conquest, genocide, acquisition and ongoing access to land and resources. Multiple methods can be used to describe these mechanisms, which can be separated into material and discursive processes (Dunbar-Ortiz, 2014). Settler colonialism, and extractive and exploitative colonialism, are considered external (often material) forms of colonization, as external powers seek to extract components of Earth (e.g., minerals, human lives and labor) to benefit colonizers (Tuck & Yang, 2012). Settler colonialism is the process in which settlers come to a land and claim it as their own, often employing genocide, erasure or subjugation of Indigenous people, and environmental exploitation (Rowe & Tuck, 2017; Veracini, 2011; Whyte, 2018; Wolfe, 2006), which are all explicit efforts to replace and dominate existing relationships between humans and land (Wolfe, 2006). As expressed by la paperson, "*Land is the prime concern of settler colonialism, contexts in which the colonizer comes to a "new" place not only to seize and exploit but to stay, making the "new" place his permanent home*" (paperson, 2017). As many colonial projects and land uses have endured over time and morphed into modern nations, it is clear that settler colonialism is a structure rather than an event. Colonial structures produce colonial land relationships, which are strengthened as genocidal projects, erasure of Indigenous people and land exploitation continues to evolve and stabilize (Wolfe, 2006).

A related concept to settler colonialism is the creation of private property and land ownership, which is implicated in land relationships and the lasting impact of settler colonialism (Atleo & Boron, 2022). Defining and ensuring settler access to territory (Wolfe, 2006) necessitates the implementation of ownership to enable the legal, just and defensible exploitation of land, which is labeled a resource. This recasting of land as property not only serves to sever Indigenous ties to land, but produces a set of alienable rights (right to own, right to law, right to govern, etc.) by which exploitation and white supremacy are legitimized (paperson, 2017). The United States Government's history of making and breaking treaties shows how using the idea of land as property—which can be bought, sold, and traded—allowed them both to interfere with Indigenous relations and to gain access to land, using violence and abrogation of treaties to continue taking and owning land (Harjo, 2014). Notably, the colonial concept of wilderness (Robinson & Tout, 2020) and the doctrine of discovery (Miller, 2010), relying on the legal term "Terra Nullius" (literally "nobody's Land": Parbury, 2011), provided justification for colonizers to claim that Indigenous land was not owned by Indigenous people, was not being used for its most utilitarian purpose, and should be owned, used, protected and conserved for settler futures (Choudry & Kapoor, 2013; Domínguez & Luoma, 2020; Whyte, 2020). This discursive justification of settler colonialism is reflected in the language that settlers use to describe the process: "discovering", "taming", “domesticating”, and “saving” places (Andres, 2003). These discourses contrast with the material reality that settled places had already experienced intensive and stable human presence for time immemorial, characterized by strong reciprocal relations between land and people (Cajete, 2016; L. J. Johnston, 2022; Kawagley et al., 1998; Whyte, 2018a).

External forms of colonialism differ from internal colonialism, which uses methods such as segregation, marginalization, forced education, and criminalization to manage and control social identities and protect colonial supremacy (Liboiron, 2021; Tuck & Yang, 2012). Accompanying settler colonialism is a history of concerted efforts to break ties between people and land. Education programs were, and are, one of several means to achieve this, both asserting the superiority of the colonizer (e.g., white supremacy) and teaching an ideology to colonized people that stabilizes and rationalizes colonial land relations, including land ownership (Kharem, 2006). Pertinent examples are boarding schools created by the United States, such as the Carlisle Indian boarding schools, founded in 1879 to “kill the Indian” to “save the Man” (Alvarez, 2023), and military run schools in The Philippines starting in 1901 to pacify Indigenous people (Constantino, 1966). Though these schools often fueled increasingly powerful resistance ideology and schooled leaders of anti-colonial resistance (Lomawaima & McCarty, 2006; paperson, 2017; Thiong’o, 2012) and were unsuccessful in converting people into acquiescent colonized citizens or erasing culture, they physically removed people (especially children) from land, and interrupted Indigenous land relations. Notably, the teaching of private property land ownership models discursively weakened existing cultural beliefs of Indigenous peoples and worked to replace them with a colonial view of individual rights to own land, while materially dispossessing them of land (Smith et al., 2022). The dissemination of colonial ideologies of how to interact with land is a central strategy in colonizing campaigns.

The terminology and common practice of distinguishing “anthropogenic” versus “natural” dust has inconsistent boundaries at multiple scales and artificially imposes a discourse of human separation from the environment. While reviewing the past 400 years of colonization, it is a short step to assert that human societies create instability and dust and are therefore harmful to the environment. This statement makes two assumptions: (1) all human societies do similar things on Earth, and (2) human societies are not part of the environment. These assumptions are not in keeping with many cultural beliefs, nor scientific ecological practice, as human societies are part of Earth ecosystems in both viewpoints. Diverse voices from a wide range of cultural and spiritual beliefs, religions, scientific disciplines, and traditions including Sufi teacher and scholar Pir-Zia Inayat-Khan, faithkeeper of the Wolf Clan of both the Onondaga Nation and the Seneca Nation of the Six Nations of the Grand River Chief Orens Lyons, Grand Chief (Mugema) and spiritual arbiter Tamale Bwoya of the Buganda Kingdom, Indian environmental activist and scholar Vandana Shiva, Franciscan friar Richard Rohr, Joyce Clague Yaegl elder and Aboriginal Australian political activist, Buddhist monk Thich Nhat Hanh, Diné and Tsétsêhéstâhese scholar and community organizer Lyla June Johnston, articulate the essential role of human societies as part of the environment and Earth (Lyons et al., 2016). Deviation from these views, where human societies are separate from and superior to nature, and are compelled to master it, is a specifically colonial perspective, and is seen in countless acts of colonial projects that pervade much of the industrialized global economy (Lalude et al., 2022).

The colonizing dynamics highlighted above are foundational in many academic disciplines. Specifically, earth sciences research and education reflect biases that are related to aspects of white supremacy, termed “white geology”, in both what is researched and how Earth systems are characterized (Yusoff, 2018). For example, the Anthropocene, when human and more-than-human dynamics are purported to have become strongly coupled, is typically placed at the beginning of the Industrial Revolution or the explosion of first atomic bomb, reflecting white experiences of impacts of capitalism and colonialism (Baldwin & Erickson, 2020). In contrast, some scholars place the beginning of the Anthropocene in 1492, at the beginning of European colonialism of the Americas, or earlier, in 1443, at the beginning of the kidnapping and enslavement of African people as part of the Portuguese slave trade (Yusoff, 2018).

2.3 Reflection on Chapters 1-3

The arc of my dissertation follows efforts to increasingly understand the effects of societal perturbations of the dust cycle. Starting in Chapter 1, we found evidence that emissions from several industrial activities and vehicles contributed up to 75% of the dust reaching a Southern California mountain. While attribution to specific dust sources was beyond the scope of the research, we discuss how the nearby sprawling Los Angeles urban environment, and the numerous land uses in valleys surrounding the mountain, cumulatively outpaced dust production from arid valleys and nearby deserts, which are thought to be the largest global dust sources (Chen et al., 2023). In keeping with estimates of dust deposition in nearby valleys (e.g., Frie et al., 2019), this result shows the significant modification to nutrient deposition and the potential introduction of harmful chemicals to montane environments, especially considering the differences between rock and vehicle tailpipe chemical compositions. Though geochemical analysis was used to reveal these trends, it could not be used to answer questions that arise from it, such as: *what are the implications of this significant industrial and urban dust transport for human and regional environmental health*? Introducing additional analytical methods is necessary to take the next step.

Combining geochemical questions with epidemiological approaches enabled me to address the question of how climate, dust from human land uses, and public health interact in Chapter 2. Specifically, following my geochemical study in Southern California, we were interested if similar climate regions (arid and semi-arid) around the globe faced similar issues, and what was known about them. In this research, we found that while previous work had documented adverse relationships relating temperature and health outcomes (Rahman et al., 2022), effects of climate change on health (Haines et al., 2006), expansion and susceptibility to dust emission by disturbance of arid and semi-arid regions (Huang et al., 2016; Safriel et al., 2005) and differential harm from differing sources of dust (Berger et al., 2018; Pennington et al., 2019), there was not significant research probing the connection between or the compounding effects from these factors. Our study found notable gaps in knowledge, which were addressed in Chapter 3. In focusing on commonalities in health risk from exposure to PM, we recognized that many types of questions were not asked concerning the health impacts outside of urban centers and the long-term connections between dust sources and human exposure. We found an overemphasis on research in higher income regions, especially near universities, while large populations and parts of the world did not receive attention, though they likely faced similar issues. We concluded that both addressing climate-related factors in studying PM exposure risk, and closing wide geographical gaps in studies, are needed.

In a concerted effort to probe interactions between specific sources of dust and how they impact people in different climate regions, we used advanced geochemical and epidemiological tools to explore how $PM_{2.5}$ exposure health impacts are modified by a range of geospatial factors. Our findings highlight how different sources have distinct impacts on public health across different climate regions, and that increased concentrations of several industrial and vehicle-related $PM_{2.5}$ sources are associated with increased hospitalizations, while desert dust is not. These findings build upon previous results that some sources of PM are more associated with hospitalization and mortality (Berger et al., 2018), but adds that geospatial variables—including climate, air basin, and seasonal variation—modify this impact.

In conclusion, important questions concerning the climate and dust sources remain unanswered and likely require additional methods to address them. While revealing previously unstudied dynamics, the scope of Chapters 1-3 remains on short-time-scale interactions and does not consider important questions such as: *what is the context for generation of these dust sources*; and *how are some people more impacted by these sources than others*? Answering these questions requires treatment of longer-term systemic variables in a methodology that includes additional approaches. Particularly, with a goal of locating the most harmful sources of dust, I recognize the need to consider how the geochemical properties of dust and current public health operate within the context of the long-term dynamics of human land use, and industrial technologies and societal inequalities, which shape questions addressed within geochemical and epidemiological research.

3. Frameworks Spanning Multiple Scales

As I have argued, considering only individual aspects and scales of dust emission and transport and health impacts using siloed analysis methods cannot fully capture ways that dust carries and distributes harm among human and more-than-human populations. To address this issue requires developing an overarching research framework in which the interconnections between individual frameworks can be mapped out. Particularly, I include research frameworks that (1) explore the nuances and apportionment of PM and dust to specific land uses and land relationships, and (2) can disentangle the distinct components of these systems at different time scales. In the following sections, I describe specific tools from frameworks that are well suited to contribute to a multi-scale analysis of dust systems and additionally highlight the limitations of each framework. Rather than construct a multi-scalar method for dust accountability by pasting together different methods that merely complement each other, I seek to join these methods in a way that reveals their interrelations and the ways that they contrast with or complement each other. The science of complex systems offers an overarching framework for doing so.

3.1 Complex Systems Science

Complexity is a property of a subset of systems in which system behaviors, when viewed from varying perspectives, can be described as, on the one hand, complicated, heterogeneous and intricate, and, on the other hand, simple, homogenous and straightforward. Complexity is illustrated by considering the example of a system where toxic dust particles are emitted from the surface of a dry lake bed, carried by wind and inhaled by agricultural workers. The intricate processes by which toxins are concentrated in sediments of the lake bed, entrained within the wind, carried through the atmosphere, and concentrated where agricultural workers gather, and then go on to impact, often in varied ways, the health of individual workers, could be described as complicated. In a societal sense, the overall result could be described in a relatively simple manner as a public health crisis, which undergoes a straightforward process of discovery, study and mitigation. In this sense, the dust-human health coupled system could be classified as a complex system.

A set of concepts (or tools) has been developed for studying complex systems that can be used qualitatively, quantitatively or both. Those concepts that are directly relevant to dust systems are described here (and more generally delineated in McNamara & Werner, 2008). First, complex systems analyses describe system behaviors or *dynamics*, which specify how a system evolves from one *configuration* to the next over short time intervals. The dynamics of multi-scalar complex systems are analyzed in an organized fashion with a foundation built from three key properties. First, *nonlinearity* is a type of dynamics characterized by strong, coupled, two-way interactions between the constituents of a system. This contrasts with *linearity*, in which these interactions are weak and one way. A conversation is nonlinear, but a regulation acts upon proposed actions in a linear fashion. Second, *dissipation* is a category of dynamics that is characterized by reduction in differences in properties of the elements of a system, as in mixing or diffusion. Processes that reduce wealth differences are dissipative. *Friction* is a type of dissipation, and when a block slides on an inclined plane, friction acts to reduce the velocity difference between block and plane. *Stability* is promoted when dissipation overshadows nonlinearity, and instability can result from the reverse situation. A property of dynamics related to dissipation is *time scale*, which describes the characteristic time over which dynamics responds to a perturbation. (In the simplest cases, *time scale* is the characteristic time for an exponential decay, with the decay scale being the time scale.) A single dust particle responds to a wind gust almost instantaneously, and the time scale of its trajectory is well below a second, whereas a dust plume in a community responds much more slowly to a change in wind conditions, lingering well after the winds that brought the cloud to the community have diminished; the time scale of the dust plume can be hours to days.

The dynamics of a complex system is organized in a succession of *levels of description*, ways of describing the system that are differentiated by time scale and by differing dynamics (a dust particle trajectory behaves very differently than a dust plume). *System boundaries* at each level of description are determined by the extent of a chain of nonlinear interactions—so short-time-scale levels of description are less extensive than long-time-scale levels of description. System boundaries encompassing a dust particle are less extensive than boundaries enclosing a dust plume.

Levels of description of a complex system occur at distinct and separated time scales. The gap between time scales of successive levels of description is determined by the rate of dissipation at the faster time scale level. For successive levels to be dynamically distinct, dynamics at the faster level must dissipate over the time scale of the slower level. Two successive levels are tied together with *feedback relations*. Dynamics of the slower scale level *dynamically slaves* and provides context for dynamics at the faster scale. Dynamics of a dust plume provides a context for trajectories of individual dust particles. Elements of the faster scale level interact and *self-organize* to give rise to and constrain, but not determine, slower scale level dynamics. Individual dust particle trajectories contribute to and constrain, but do not determine, dust cloud dynamics.

Systems with high rates of dissipation are characterized by small time scale gaps between levels, with rich, branching upward successions of levels of description—an organic, highly interactive system. Systems with very small rates of dissipation have large gaps between levels, which, because the longer-time-scale level has such extensive system boundaries, retain the context of dynamical slaving but fail to retain the action of self-organization, and therefore lose two-way, nonlinear, feedback relations between levels. This contrast can be identified in the differences between capitalist societies in which firms, agencies and individuals are focused on efficiency, or reducing frictional, dissipative processes that detract from goals and profits in transactions (Griffin, 1993), and Indigenous, relational societies that prioritize stability, balance, responsibility and good relations over the smoothness of interactions (LaDuke, 1999; Whyte, 2016a). Traditional Indigenous societies are characterized by a rich network of relations across numerous scales, where human and more-than-human, gender, sexuality, time and temporality are blurred distinctions (e.g., Rifkin, 2010, 2017; Whyte, 2018a). In capitalist societies, a primary and asymmetrical interaction exists between individuals and the state across a time scale gulf, and a proliferation of categories, often determined statistically, provide paramount context for individual dynamics (e.g., Foucault, 1990). Because efficiency in capitalist societies pushes overarching dynamics to very long time scales, these dynamics can appear static and resistant or impervious to change, reflected in the slogan (referring to liberal capitalism and free markets), ‘There is no alternative’ (e.g., Thatcher, 1980), as well as arguments for ‘the end of history’ (e.g., Fukuyama, 1992).

This apparently static characteristic of capitalism and of associated systems of power focuses much of the identification of injustices by communities and the goals of their resistance on fairly limited, shorter-time-scale, proximate, analyses in traditional environmental injustice organizing. This is because individual communities are dwarfed by the enormity of the systems of power providing the dynamical context for these injustices. In contrast, resistance movements that focus on longer/larger-scale systems of oppression like capitalism and colonialism tend to be multi-decade or multi-century struggles, as in the Native-American struggle against European settler colonialism of Turtle Island (Dunbar-Ortiz, 2014), the struggle of workers against capitalist exploitation (Thompson & Bekken, 2006), or the Palestinian struggle against Zionist settler colonialism (Qumsiyeh, 2011). The goals of these movements can be vague or undefined, and progress is slow, episodic or difficult to discern. Analysis methods for connecting struggles across scales are largely lacking, and here complex systems methods can make a contribution.

The disciplines that form the components of my proposed methods focus on analyzing aspects of systems that differ in time scale and scope. All methods address dynamics that vary in time scale from fast-scale actions to long-time-scale context, but the ways that they define the extent and scope of the system—and the system boundaries—center dynamics on a narrow range of time scales, because of the connection between system boundaries and time scale via nonlinearity. Geochemical methods focus on the dynamics of dust grains and associated patterns of chemical processes, and so this scope puts the spotlight on shorter time scales associated with the immediate sources of dust particles. Epidemiology expands the scope to include human health processes and patterns, and so zooms out to consider somewhat longer time scales associated with patterns of dust generation and harms. Critical Environmental Justice analyses include health, environmental harms and political processes, and so expands the scope and time scale at which dynamics are focused to include the political actors and institutions and their economic collaborators that underlie dust generating systems and their selective modes of mitigation. Anti-colonial methods, which expand to encompass longer-term, larger-scale processes of colonial displacement, exploitation and extraction, track the historical processes that generate persistent dust-producing and dust-impacted population classes, as well as dust generating sacrifice zones and privileged 'natural' and built environments, where harms are limited or remediated.

3.2 Geochemistry

The geochemical study of dust as part of the Earth-System framework (Ridgwell, 2002; Shao et al., 2011) provides information about the composition of dust, its behavior in various settings, and, importantly, how human societies modify its role in larger biogeochemical cycles (Chen et al., 2023; Webb & Pierre, 2018). In the proposed framework, I focus on tools to identify the multiple sources of dust within the atmosphere by characterizing its chemical composition—known as source apportionment—to enable analysis of how land use affects dust production, dust toxicity and its potential impact on humans and the more-than-human world (Berger et al., 2018; Hopke, 2000; Reche et al., 2012). Previously in this dissertation, I investigated multiple source apportionment methods to identify both the presence of urban and industrial dust in montane environments (Chapter 1) and the differential risk of exposure to different sources of dust (Chapter 3). These tools quantify land-surface processes over limited time horizons and potentially can contribute to analyses over the longer scales of epidemiological, CEJ, and anti-colonial research.

3.3 Epidemiology

As discussed in this dissertation's introduction and Chapters 2 and 3, epidemiology is a core discipline of public health. Epidemiology encompasses the analytical or descriptive study of the distributions and determinants of health outcomes in populations (Song & Chung, 2010; Szklo & Nieto, 2014). In this framework, epidemiology provides the tools to measure the statistical significance of associations between population health metrics and air pollution exposure. Over short time horizons, epidemiology provides a means to assess the health effects of dust production from different land uses, and how that impact, or risk, is distributed across a diverse population. Epidemiological results contribute to CEJ analysis of structural social and environmental impacts of dust over a longer time horizon.

Epidemiological research does not address: how harms faced by humans are similarly experienced by diverse organisms (such as studied in ecological research); how environmental system functions are adversely altered by land uses; and how interconnected and compounding social and environmental harms are wrought over hundreds of years by processes associated with colonialism.

3.4 Critical Environmental Justice (CEJ)

The Critical Environmental Justice (CEJ) framework can determine how epidemiological research of the burden of disease from exposure to particles (researched using geochemical tools) exists within socio-political structures of power and systemic marginalization across multiple intersecting scales (Pellow, 2017). In *What is Critical Environmental Justice?,* David Pellow makes connections between polluted, unsafe, and toxic environments and co-occurring social violence and injustice. Specifically, the framework of CEJ articulates how the oppression of one group or place is never isolated; rather systems of oppression—whether, for example, devaluing the lives of people based on race, or exploiting land—are integrated in such a way "*that augments and compounds the mistreatment of others*" (Nilbert & Fox, 2002). In the context of dust, extractive land uses, and creation of toxic dust compounds other forms of injustice experienced by ecosystems and people from the same land use systems. Pellow describes four central pillars of CEJ—intersectionality, multiscalar research, abolition and indispensability—and how together these analysis tools can enable communities, researchers and activists to better understand environmental injustices and to meaningfully address them (Pellow, 2017).

The first pillar, intersectionality, examines how experiences of injustice are interwoven through multiple forms of social, environmental, and economic violence and exploitation. Analysis of how interwoven means of oppression operate, provides deeper insight into how to resist systemic injustices (Pellow, 2017). Research that integrates multiple spatial and temporal scales can focus on different causes, consequences and resolutions to struggles (Pellow, 2016). Multi-scale research, introduced in Section 3.1, allows for a better grasp of fundamental actors and their dynamical behavior or intentions, and how these actors influence current conditions or experiences. The logic that intersectional and multi-scalar viewpoints are necessary to understand the harm from dust emissions is central to the method described here. In recognizing that there is no potential for justice—experienced by land and people—inside of colonial land relations, the abolitionist pillar of CEJ shows why anti-colonial frameworks and research methodologies are necessary to both my proposed method and to meaningfully address multi-scalar dust related health crises (i.e., climate, human, more-than-human). While CEJ does not provide detailed analytical tools to reveal Earth-system functions or statistical PM exposure and heath burden relationships, it does provide essential context for studying patterns in reoccurring outcomes from colonial land relations and related harmful dust emissions.

3.5 Anti-colonial research methods

The persistence of capitalism relies on structures that perpetuate material and discursive harm (paperson, 2017), including academic research, where "*settler perspectives and worldviews get to count as knowledge and research and how these perspectives - repackaged as data and findings - are activated in order to rationalize and maintain unfair social structures*" (Tuck & Yang, 2012). Indigenous and non-Indigenous scholars have developed, written about, and practiced anti-colonial methodologies and frameworks that confront, critique, and reject colonial research that stabilize structures of harm (Carlson, 2019; Johnston, 2022; Liboiron, 2021; Smith, 2021; Tuck & Yang, 2012; Whyte, 2018). These concepts and methods "*centre and change colonial land relations in thought and action*" (Liboiron, 2021, p. 6). Simultaneously, they reject efforts at "*making a colonial empire more inclusive*" or propping up colonial land relations by following themes of "hope" and successful settler multiculturalism (TallBear, 2023). Colonialism operates in many forms, including land theft, dispossession, erasure of Indigenous peoples, and efforts to stabilize and advance settler futures on land. These forms of colonialism sometimes are branded as "decolonizing" projects or efforts, with no intention of actually "*repatriating land to sovereign Native tribes and nations, abolition of slavery in its contemporary forms, and the dismantling of the imperial metropole*" (Tuck & Yang, 2012, p. 31).

In this research, I am particularly interested in understanding how the onset and continuation of colonial land use weakens relationships, leading to ecological instability, dust generation, and the creation of multi-scale harms, as discussed in Section 3.4. Focusing on the functions and components of colonialism facilitates analysis of how long-time-scale dynamics, such as settler colonialism and land exploitation spanning hundreds of years, manifest in short-time-scale behaviors and impacts. In anti-colonial frameworks, this entails the characterization and rejection of colonial land relations (discussed using complex systems concepts in Section 3.1) and the pursuit of anti-colonial land relations, which can be multi-faceted, for example, Indigenous, queer, feminist and Afro-futurist (Liboiron, 2021).

As articulated in *Pollution is Colonialism,* the creation of pollution—including PM created through colonial land acquisition, dispossession and exploitation—are essential processes in colonial land relations, not unintended outcomes (Liboiron, 2021). The harmful emission of dust, as a result of colonial land uses, is a short-time-scale manifestation of colonial land relations, and not a separate issue that can be resolved or minimized without addressing the overarching structure. Therefore, pursuit of technological and regulatory approaches to minimizing harmful effects of dust without addressing colonial land relations merely reproduces the problem at other sites of dispossession and extraction and ensures a stable settler future. Instead, research that addresses the harms from land-use-related dust production must incorporate analyses across multiple scales.

3.6 Critique of approaches to mitigating harms

A common approach to air pollution by governments and many academic researchers is to create risk assessments, which in turn inform risk management (including policies, regulations, etc.) (Vallero, 2014). Fundamentally, this approach yields solutions to environmental pollution-related issues by creating regulations and infrastructural improvements that further stabilize colonial land uses. Harms are minimized if their magnitude is deemed too great, or conversely, harms are tolerated as acceptable risks if an obvious cost-effective solution cannot be found (Liboiron, 2021). One tool to make risk assessments in the research of air quality is a concentration–response function (CRF), which can produce estimates of attributable deaths and/or diseases, years of life lost (YLL), or disability-adjusted life years (DALYs), or change in life expectancy from exposure or a change in exposure to air pollution (Health Risk Assessment of Air Pollution – General Principles, 2016). This method is very effective in using multiple data sources to demonstrate both risk and impact of air pollution (including PM and dust), and helps inform public policy to limit exposure—such as through regulations, city planning and market-based strategies (Craig et al., 2008; Hoffmann, 2019). For example, risk assessments were used by the United States Environmental Protection Agency to create National Ambient Air Quality Standards (NAAQS) through the Clean Air Act (CAA), defining maximum allowable levels of some air pollutants: lead, total PM, sulfur dioxide, and nitrogen dioxide. Additionally, the Biden-Harris administration was recently celebrated for reducing acceptable levels for toxic metal emissions by 67% and mercury by 70% from existing lignite-fired sources (US EPA, 2024b). However, in 2023 it was estimated that 140 million people (roughly 42% of the U.S.) live in areas with pollution levels above the NAAQS (US EPA, 2025). In 2020, it was estimated that it would cost $65 billion to comply with CAA, and that the benefits (e.g., reducing healthcare costs) would be potentially 30 times greater than the initial cost (i.e., ~$2 trillion) (US EPA, 2024a). The lack of short-term investments in complying with established regulations and the inability to address the many other social, economic, and environment costs of air pollution indicate that this regulatory approach is insufficient, and perhaps of marginal significance.

These approaches cannot address structural environmental justice issues of disproportionate exposure and health burdens in marginalized communities or meaningfully address the multi-scalar harms from extractive industries. These regulations assume an acceptable maximum amount of pollution by colonial activities on Indigenous land. Regulations are created with an implicit goal of minimizing harm to increase the stability of the colonial society, while not addressing structural inequalities, grounded in colonial land relations (Liboiron, 2021). The resulting social and political discourse focuses on improving how colonial land use functions, without questioning their legitimacy. The majority of chemicals currently manufactured and emitted into the environment in the US have not been studied for human and ecological impacts (Egeghy et al., 2012; Judson et al., 2009), which is consistent with the hypothesis that air pollution management and chemical regulations follow a colonial model of valuing land exploitation and benefit for the colonial power, at the expense of people and land. Questioning the implied assumptions of risk estimates and foregrounding critiques of structurally harmful land uses could meaningfully advance the contributions of these disciplines to public knowledge, environmental justice, and public policy.

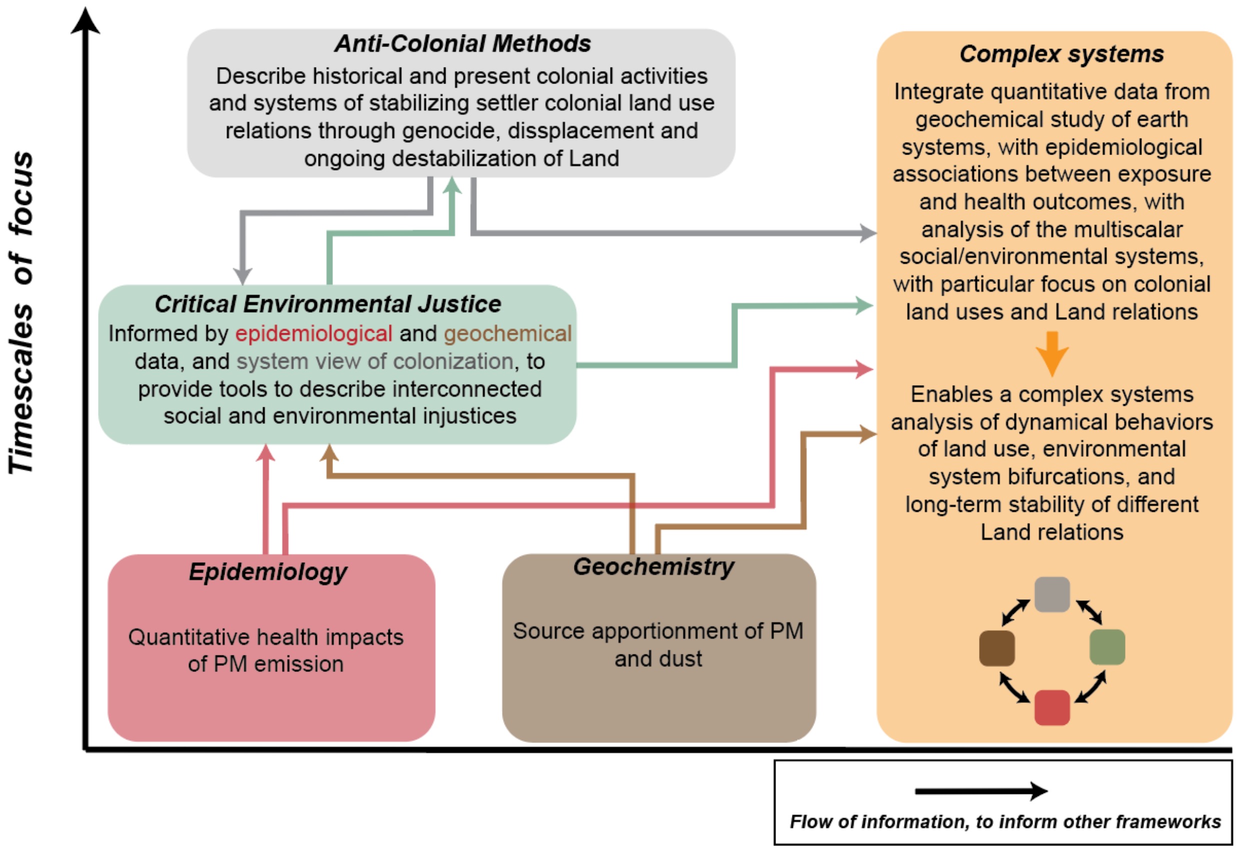

Figure 4.1. A diagram that shows potential interactions between multiple frames and how each addresses a distinct scale of study.

3.7 Proposed multi-scale method for dust research

As outlined in sections 3.1-3.5, and in Figure 1, I combine five distinct frameworks, into a coherent multi-scale analysis of the dynamics of particle emissions and harms. This proposed multi-scale perspective on PM is informed by research from many of the authors cited so far in this study (e.g., Liboiron, 2021; Pellow, 2017; Wolfe, 2006).Their research supports a perspective that rejects the use of environmental and public health research to stabilize and perpetuate mechanisms of injustice.

Concepts from complex systems science can be employed to connect short-term changes to Earth's surface and human health to long-term arcs of cultural discourses, political and economic power, and colonization. Epidemiology and geochemistry strongly influence governmental policies regarding air pollution, and integrating these disciplines with more expansive research frameworks holds a promise for a more complete and meaningful analysis of modern environmental and public health issues, thereby responding to la paperson's call to develop subversive decolonial projects that make use of colonial institutions and are located inside of them (paperson, 2017).

As each of the highlighted frameworks make visible different scales of ecological and human relationships, robust multi-scalar results can equip communities with increased decision-making capacity and agency in employing information, more than is possible with scale-limited geochemical and epidemiological reports of pollution and/or impacts. Research that explores these interconnected frameworks can quantify PM and dust emissions resulting from different land relations and integrate the often hidden and externalized emissions that are necessary to and part of land use.

This multi-scalar method aims to apportion dust and PM emissions to the land use technologies and actors that create them. Paralleling the logic that Dalin et al. (2012) and Rulli et al. (2013) use to trace global flows of water associated with global trade and land grabbing, I aim to connect the emission of dust to the benefactors of its production. Zooming out to longer time scales, I explore how to connect the dynamics of current land use to intergenerational harms from dust and PM emission and exposure, and thereby apportion harm to the beneficiaries of injustice and colonial extraction. A schematic framework for this concept is shown in Figure 2, where research tools from different disciplines are used to investigate multiple types of data and analyze their associations. As shown, my intent is to address and quantify: (1) what are the factors and processes underlying dust emission, (2) where do the material and economic benefits from emissions flow, and (3) what actors or structures benefit from harm resulting from emissions. The answers to these questions are a starting point for connecting sources—and their benefactors—with resulting human and environmental harm, with the aim of pursuing a quantitative accountability framework. I expect that the answers to these questions will be different on long and short time scales, and so might not be reconcilable. Identifying how to assign land uses to contemporary and historical actors, and accurately apportion data of public health impacts to particular sources of dust and PM, will require careful research. This research could be supplemented with land education that considers Indigenous, post-colonial, and decolonizing perspectives on place (Tuck et al., 2014), so as to not perpetuate harmful colonial research or the view of polluted environments as "*wastelands ripe for rescue by ecological settlers*" (Paperson, 2016).

At this stage, I have not developed sufficient methods to fully propagate and quantify land use change and harm across time scales. My intent is to demonstrate the need for such analysis, provide potential frameworks and specific tools to accomplish it, and point to its potential benefit. However, proxies for benefits accrued from dust emissions include the revenues and profits associated with activities that give rise to dust emissions at shorter time scales, and activities that benefit from structures such as white supremacy, settler violence and colonial dispossession at longer time scales. These proxies can be connected to harm at short time scales by apportioning according to toxicity of dust, and additionally at longer time scales by how structures facilitate exposure. An example of the proposed method to study dust emissions and health impacts in the Imperial Valley of Southern California is explored below.

4. Knowledge from multi-scale frameworks in Imperial Valley

As an exploration of existing and potential multi-scale frameworks to attribute harms and benefits from dust emission, as introduced in section 3.7, I discuss the relationship between people and land use in the region that is now the Imperial Valley. I focus on the Imperial Valley as it parallels the study of arid regions in California contained in the previous three chapters. Additionally, the Imperial Valley has been a focal point of intersectional environmental and public health research in Southern California (Johnston et al., 2019) relating to dust exposure, including the exposed lake shore of the shrinking Salton Sea (Jones & Fleck, 2020). The colonial and environmental history of the Imperial Valley is well-documented by scholars, so the focus of this study will be exploring existing knowledge and connections to multi-scale research of dust. After discussing relevant history, I explore the interaction between long-time-scale dynamics (through complexity and anticolonial research) and current short-time-scale dynamics (via CEJ, epidemiology, and geochemistry).

4.1 History of interconnected exploitations

The Imperial Valley, in southeastern California (Figure 2), currently experiences an arid, desert climate with notable presence of the Salton Sea to the north and the Colorado river on its eastern border. At several intervals during the Holocene, a large inland lake—known as Lake Cahuilla—covered the valley, followed by desiccation and the return of a desert climate (Rockwell et al., 2022). People Indigenous to this area have lived in and adapted to this changing climate since time immemorial, with archeological evidence of human presence dating to approximately twenty-thousand years before present (Herrera, 2004). The past 200 years of settler colonial occupation has dramatically changed land-surface stability, the flow of water, the emission of dust, and long-term Indigenous land relationships (Voyles, 2021), leading to distinct environmental injustices and subsequent community resistance.

Written records of the colonial history of the region begin with initial interactions between Spanish colonizers and Indigenous communities in the 1500s, and continue with Spanish missions in the 1700s, multiple successful repulsions of Spanish colonizers by Quechan Indigenous communities, and ongoing United States occupation, which began following the Mexican-American War (Herrera, 2004). Near the end of the 1800s, the majority of European settlers in the valley were seeking gold. Records indicate that miners stole crops and other items from Quechan communities during this period (Fernandez, 2012). Settler presence markedly increased with large-scale manipulation of the Colorado River starting in 1900. The California Development Company began construction of a canal to divert part of the Colorado river into the Imperial Valley for irrigation (Lawrence, 2019). However, a levee was overtopped in 1905 during a flood, causing up to 90,000 cubic feet per second of Colorado river water to flow into the Imperial Valley for a period of two years, forming the Salton Sea, until the breach was fixed (Kennan, 1917). Stable diversion channels were built for crop irrigation, and runoff from agricultural fields has maintained the Salton Sea ever since. Irrigated agriculture initiated a period of intensive settlement and led to extensive agricultural fields covering much of the valley floor (Andres, 2003). Following violent conflict between United States settlers and Indigenous communities (including Quechan, Maricopa, Kumeyaay, Cahuilla, Cocopah, and Pai Pai), the US military established Fort Yuma to overpower Indigenous resistance to colonial presence and encroachment on traditional lands (Von Werlhof, 2004). Following the establishment of Fort Yuma, an unratified land cessation treaty granted the Quechan (and no other tribes) less than 10% of their traditional lands on which they had farmed squash, corn, beans, and melon for thousands of years. The allotment was monitored by the Bureau of Indian Affairs, which allowed non-Indigenous settlers to farm the best land on the allotted area. The 1887 Dawes Act, the 1934 Indian Reorganization Act, the 1953 Public Law 280, censorship of traditional ceremonies, outbreaks of sickness, forced cutting of hair, Canadian corporation gold mining on sacred sites, and Catholic boarding schools all significantly impacted the Quechan community's way of life (Von Werlhof, 2004), while settlers profited off the colonized land and strengthened colonial land-relations.

In the early 1900s, 450,000 acres of land was claimed by settlers for agriculture, overseen by the California Land Development Company and the Imperial Land Company, which built settlements, attracted colonizers and distributed land and water rights (Durazo, 2004). The economic success of the growing cash-crop agricultural industry prompted farmers to search for more laborers. Intensive oppression and violence directed at people Indigenous to the Imperial Valley was rooted in white supremacy, galvanized by the idea that settlers had "tamed" the Imperial Valley. This racist violence was redirected to Japanese, Filipino, Punjabi, Mexican, East Indian, Chinese, and Quechan farm workers, who were the main agricultural labor force of the 20th Century (Fernandez, 2012; McWilliams, 2000). The racial hierarchy evolved to include segregation, denial of land or water rights for BIPOC, wage disparities, suppression of speech, and violent repression of labor unions and ethnocide, although all were subverted (Fernandez, 2012). Farmworker strikes in the late 1920s and 1930s spurred by growing union presence in response to unsafe conditions, low and withheld wages, were met with beatings, kidnappings, arrests, and threats of deportation (Fernandez, 2012; Weber, 1972). Though these union-led resistance movements are seen as having had some success, they did not change underlying racial inequities, which increasingly solidified. In late 1930s, white farm workers from the Midwest, called "dust bowl refugees," demanded and took jobs from BIPOC and migrant workers, and received government financial aid, while BIPOC farmworkers received no benefits (Beltran, 2022). The 1942 Bracero Program further decreased the social and political power of Mexican migrant farmworkers to advocate for their rights (Mandeel, 2014), which led to the increasing criminalization of workers who were the backbone of profitable exports. The rise of immigrant detention centers in the 1940s furthered racial capitalism by using carceral systems to extract labor from migrant farmworkers (Ordaz, 2017). Private corporations created profitable business detaining people; while the threat of detention and constant surveillance in the Imperial Valley was intended as a deterrence, as federal officials sought to criminalize and degrade Mexican workers, regardless of their immigration status (Ordaz, 2017). The increasing control of agricultural land by a smaller number of landowners, who lived outside the county and profited from the national reliance on Imperial Valley food production, furthered the mistreatment of workers and also stymied efforts to address mounting environmental issues into the 1960s (Rudy, 2003).

The county as a whole—including more than 500,000 acres of farmland—relies almost entirely on water fed through the 80-mile-long All-American Canal, which was built in 1934 (Null, 2017). The Imperial Valley is the southern half of the Salton Sink, a basin that has no outflow, as it is below sea level. All water, sediment, sewage, pesticides, etc., accumulates in the basin, especially in lake sediment. Pesticides and industrial and urban pollution flowing into rivers draining to the Salton Sea concentrates extremely high levels of toxins in lake sediments (Sapozhnikova et al., 2004). A wide range of chemicals and pesticides are found, including Chlorate, 2-4-D + 2-4-5-T, commonly known as agent orange (promoted as an effective weed killer, especially to kill native vegetation that was deemed undesirable), organophosphates and organochlorine compounds, such as DDT (Voyles, 2021). By 1971 it was estimated that 5 million tons of pesticides were applied to 500,00 acres of farmland annually (Dunning, 1972). Starting in the 1950s and prior to extreme levels of toxic accumulation, the Salton Sea was known for tourism, recreation, fishing and warm weather vacations close to South California urban centers. This reputation grew throughout the 1960s but rapidly crashed in the 1970s with growing awareness of deteriorating lake conditions and high toxin levels in fish (Voyles, 2021). A series of fish and bird die offs in the 1970s, 1992 and 2005 brought public awareness of the extent of the environmental damage from multiple decades of toxic chemical use and land exploitation (Hurlbert et al., 2007).

Because of new policies and regional droughts in California, Salton Sea water levels started dropping in the 1960s and continue today, leading to shorelines being exposed, with high concentrations of accumulated toxins. This sediment, when dried, is extremely vulnerable to wind erosion, and has been found to be harmful for human health (D'Evelyn et al., 2021). Exposure to shoreline PM has caused major respiratory and cardiovascular health issues throughout the region, significantly impacting the majority Hispanic community (Lawrence, 2019). Regional shoreline dust storms are linked to an increase in mortality, asthma hospitalization, and decreased pulmonary function in adults and children (Johnston et al., 2019). Additionally, 68% of children live within 400 meters of a site where pesticides are currently applied, one in five children have asthma, and the application of pesticides triggers respiratory responses in children, such as wheezing (Ornelas Van Horne et al., 2022). Chronic exposure to harmful environmental conditions in this region has been classified as a state-corporate crime and an environmental injustice, owing to the systemic failure of government or industry to respond to environmental conditions that are causing ongoing and preventable health outcomes (Lawrence, 2019). These exposures also have been classified as environmental racism because of the structural violence and inadequate access to basic public services or health care, based on the affected community's racial makeup and position in society (Rodriguez, 2021). Compounding the issue are insufficient access to healthcare (Juturu, 2021) and poverty. In the 2000 census, median household income in Imperial Valley was $15,500 lower than the state average, with 30% of residents and 44% of children below the poverty level, and unemployment rates fluctuating between 19 to 30% (US Census, 2000; White, 2006). Other compounding issues include the increasing number and severity of wildfires in Southern California and agricultural biomass burning, which serve as particularly harmful sources of dust exposure (Aguilera et al., 2021; Kamai et al., 2023). Additional environmental impacts result from recent lithium mining projects in the Imperial Valley to meet a federal mandate for domestic production of lithium batteries. This new form of extraction continues the pattern of mining and agricultural industries profiting off Imperial Valley land, with few demonstrated benefits for local residents or workers (Buss, 2022).

In summary, the complicated history of settler colonial oppression of both Indigenous communities and the BIPOC agricultural workforce of Imperial Valley has continued with little respite for over one hundred years. Current public health and environmental concerns are inseparable from long-term environmental racism and injustice, and the colonial history of land exploitation (Voyles, 2021). The current public health crises can be traced to willful inaction to address known concerns (Lawrence, 2019), a systemic prioritization of economic gains for landowners over human and environmental health.

4.2 Multi-scale frameworks and existing research

As highlighted in the previous section, environmental, social and colonial histories in the Imperial Valley are deeply intertwined. Specifically, current dust exposure-related health issues are the product of long-time-scale land uses and unjust treatment of workers and residents. Overall, this history reveals connections between long-time-scale trends, intermediate time scale patterns of actions, and short-time-scale dust emissions and harms. Analyses of these interacting dynamics have informed powerful trans-disciplinary research (e.g., Fernandez, 2012; Voyles, 2021), and efforts by Indigenous and non-Indigenous communities to reject the continuation of structural injustices and of efforts to address environmental and human health issues through maintenance of colonial land relations.

Documentation of harms from dust exposure in the Imperial Valley is occurring both through a broad range of statistically-based epidemiological studies (e.g., Miao et al., 2025) and from stories told by affected residents, obtained via community-engaged and community-led research projects resulting from new types of partnerships between academic researchers and community members (e.g., Cheney et al., 2023; London et al., 2020; Mack, 2020; Sinclair et al., 2024; Van Horne et al., 2023). Long-term environmental justice and anticolonial movements (Fernandez, 2012; Sine, 2016; Underhill et al., 2022) have contributed to documentation of colonial dispossession, displacement and extraction. In addition to state and federal air quality monitoring networks, a community led network—Identifying Violations

Affecting Neighborhoods (IVAN)—includes over 40 monitors, and enables local community members to collect, own and use local data for aims defined within their communities (Carvlin, 2018; English et al., 2017). Research projects have addressed different facets of dust emission and human exposure that illuminate connections across scales, including: the variable inflammatory potential of PM from different sources (Mack, 2020) and different sizes (D'Evelyn et al., 2021); multiple PM source apportionment projects (Frie et al., 2019; Mendoza et al., 2010; Watson & Chow, 2001); studies on public health effects of exposure to PM, especially on vulnerable populations (Farzan et al., 2019; Farzan et al., 2024; Ornelas Van Horne et al., 2022, 2024; Zorn, 2024); studies of source-specific PM exposure effects on cardiorespiratory hospitalizations (Kamai et al., 2023; Miao et al., 2025); study of climatic and meteorological conditions (Bradley et al., 2022; Evan et al., 2022; Frie et al., 2019; Johnston et al., 2019; Schwabe, 2022); and advancement of new research frameworks to study environmental injustices (Miao, 2023; Van Horne et al., 2023). Altogether, studies using a wide range of research tools and frameworks, have provided useful information to community members, politicians and industry about the historical and present intersection of environmental and social harms related to dust generation in the Imperial Valley and Salton Sea Basin.

To advance the study of dust in the Imperial Valley beyond these existing projects, I propose the use of the multi-scale framework to quantitively apportion current environmental harms to specific sources and investigate how they relate to long-time-scale dynamics. For example, using a combination of geochemical and epidemiological tools, it is possible to trace the disproportionate impacts of colonial land uses on public health. As of 2023, 40% of the Imperial Valley land-cover is scrubland, 32% is barren desert, 17% is agricultural, 3% is urban, 6% is open water, while less than 1% is industrial (Figure 2) (U.S. Geological Survey, 2023). However, results from my Chapter 3 $PM_{2.5}$ source apportionment study in the Imperial Valley city of Calexico, show that, in contrast to land surface fractional cover, the majority of $PM_{2.5}$ arises from urban, industrial, and vehicle emissions (Figure 2). Calexico is not representative of whole region, and the relative contribution of dust sources varies significantly over the county. Frie et al. (2019) found that Salton Sea playas contributed the majority of dust in northern Imperial Valley, while previous studies in the Calexico area found carbonaceous particles (e.g., from vehicle exhaust and biomass burning) were the most abundant dust sources (Watson & Chow, 2001). Overall, further analysis of existing PM data could demonstrate that colonial land uses (including water and toxic chemical diversions to the Salton Sea) contribute more dust to the ambient atmosphere than undisturbed desert land surfaces. Epidemiological study, including results from Chapter 3, show that Salton Sea sediment, vehicle emissions and biomass burning are more harmful to human health (i.e., a greater association between increased particle concentrations and cardiorespiratory hospitalizations) than desert dust (Berger et al., 2018; Miao et al., 2025). With further study in the Imperial Valley, it may be possible to quantify the extent to which colonial technologies produce more—and more harmful—dust than desert surfaces, which are generally thought to be the most significant source of dust globally (Chen et al., 2018).

However, considering multiple time scales and levels of description, additional kinds of attribution are possible. At fast time scales, it is possible to determine sources of dust and their relative contribution to ambient dust in the atmosphere by employing geochemical tools. Using source profiles, it is possible to attribute dust to industries, companies or entities who create, and benefit economically from dust emission (e.g., through industrial manufacturing, farming, driving, etc). Further, by calculating the statistical population-level associations between acute human health outcomes from exposure and specific sources of dust, the associated health burdens for some populations can be considered alongside the benefits other populations receive by generation or maintenance of that source. These epidemiological methods study correlative associations between exposures and health outcomes, not causal associations. Therefore, this approach cannot produce a definitive attribution of harm to specific actors, but it can identify the statistical significance of the exposure and its cause (i.e., source emission). At a slightly longer time scale, differential exposure and harm experienced can be measured alongside the industries (e.g., alfalfa cash-crop in the Imperial Valley) and demographics (landowners and recipients of food production) that benefit from specific types of land use and associated emissions. These measurements rely on analyses of environmental injustices, stemming from colonialism and white supremacy, that lead to chronic exposure to PM. Further, it is in principle possible to trace the entities and socio-political forces that benefited from water diversion and manipulation systems and agricultural production, as well as and how they are stabilized over decadal time scales through the development of systems that generate dust. Conversely, the majority BIPOC workforce and other demographics structurally experienced harms from dust emission. At the longest scales, focusing on colonial dynamics of land dispossession and displacement of people and extraction from the more-than-human world, it is not possible to directly apportionment dust to individual actors, which are dynamically illegible at these scales. However, by tracing colonial land uses and resulting industries, institutions and societies, it may be possible to produce categorical attributions. This method of accounting would entail separating colonial benefits from dust production into two levels: first, diffuse benefits for regions, cities and institutions; and second, specific actions that make space for settler farmers (e.g., creation of canals and land acquisition) and industries, where beneficiaries are narrower, such as the California Land Development Company. With increasing time scale, cascading levels of association between dust production and beneficiaries becomes more generalized, and quantitative estimates might have larger uncertainties.

Conceptually, the methodology of associating dust emission with multi-scale harms and beneficiaries combines existing knowledge and research in new ways, suggesting a reorientation of the focus of study from short-term harms to systems that produce harm. Given existing tools to analysis the source of PM and harm from PM—which have already been applied in the Imperial Valley—and extensive reporting of the colonial history of the region, a critical next step is to directly address imbalances in profit from settler land uses, and numerous forms of harm. To illustrate the potential for future quantitative analyses, I summarize some of the processes underlying dust generation at multiple scales, and connections between them (Figure 2).

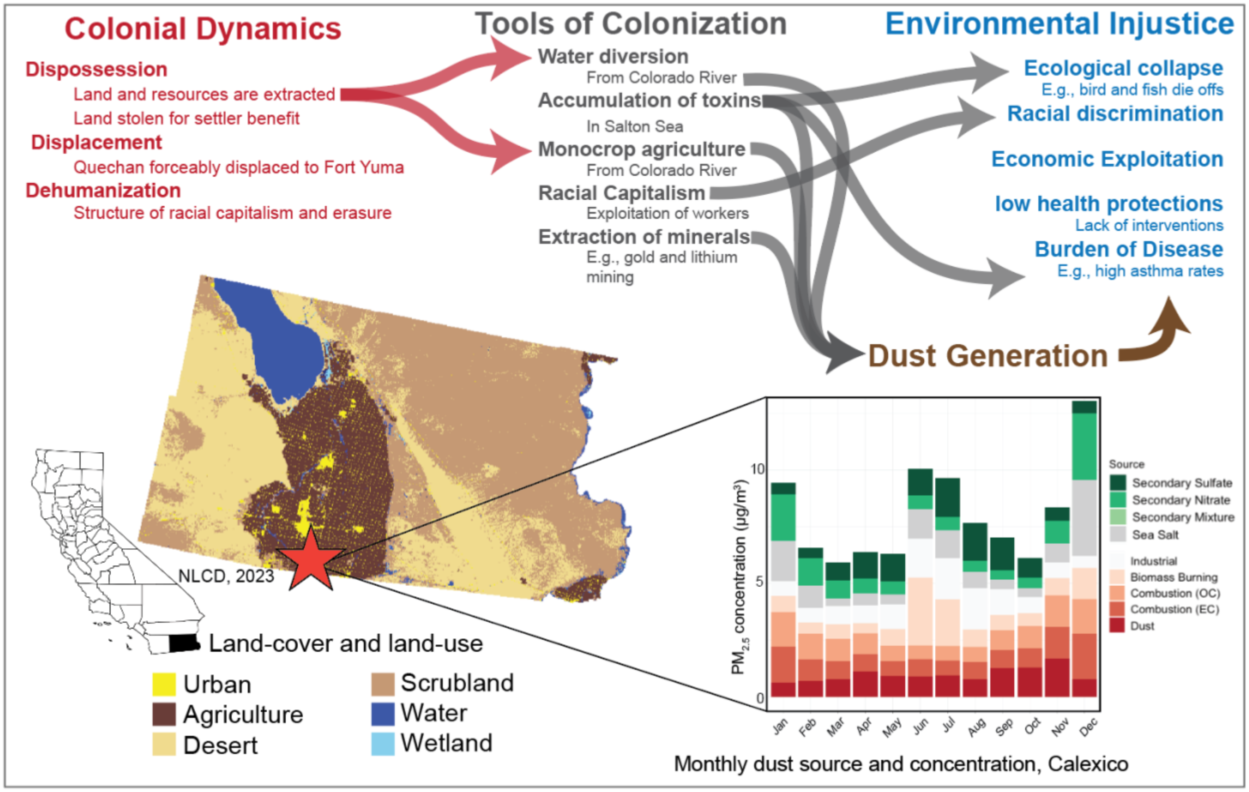

Figure 4.2. A conceptual diagram of the interactions between long-time-scale dynamics, colonialism and short-time-scale processes, dust emission. Arrows connect processes (via context or self-organization) between scales in the Imperial Valley, but are not comprehensive. Land dispossession and displacement of Indigenous people by settlers to extract resources (red), creates context for the use of tools of colonization (gray), that have led to injustices (blue) in the Imperial Valley. A land surface map of the Imperial Valley (U.S. Geological Survey, 2023) shows the Salton Sea in the northwest, and urban and agricultural fields to the south. A bar graph of average monthly $PM_{2.5}$ concentration over the past ten years in Calexico (red star on the map) shows predominance of land-use-related dust, disproportionate to land cover (from Chapter 3 research).

4.3 Limitations

Here I have focused on an analysis of material dynamics associated with dust and health. The dynamics of capitalism and colonialism are strongly dependent on discursive dynamics, which in a world dominated by alternative facts, is becoming increasingly central in interacting with and interpreting the material world. For example, discourses are centrally important to enabling injustices via dehumanization or obscuring or disappearing harms. And so further work is needed to research accountability for creating the discourses that impact harms from dust.

While I have argued for the value in exploring this multi-scale framework, I am still at a preliminary stage of assessing its function. Specifically, measurements and datasets needed to effectively characterize dynamics in each framework and scales need to be identified. Also, I recognize the lack of tools to quantify environmental harm, such as is commonly available in ecological disciplines. Overall, the proposed framework is presently conceptual, with the intention to demonstrate how current approaches to research of dust and PM in geochemical and epidemiological disciplines may benefit from addressing the burden of colonization and environmental injustices, and question how research meant to alleviate injustice might actually strengthen and perpetuate systems that have brought lasting harm life on Earth. Additionally, I acknowledge that while the focus of this framework is creating a method to account for multi-scale harms from PM generation, it must eventually incorporate tracing resistance to the systems that create and perpetuate those harms.

5. Conclusion and outlook

In this chapter, I highlighted dynamics of the dust cycle that are connected to but beyond the scope of Chapters 1-3. With the addition of environmental justice and anticolonial frameworks, I have explored how the tools of geochemistry and epidemiology can be re-purposed to address longer-term and systemic dynamics that underlie dust emission and harms from exposure. To do this, I analyzed the dramatic changes to land use and dust emission that occurred during European colonization, including specific dynamics of colonialism, and those changes that resulted from industrialization. Because each chapter of this dissertation builds off the previous, the inclusion of environmental justice and anticolonial, and complex systems frameworks was a natural next step. The application of my proposed multi-scale analysis methodology to the Imperial Valley maps out an example of future beneficial research pathways for advancing understanding of the role of dust in a rapidly changing world.

Throughout my dissertation I have researched global concerns regarding the role of humans in manipulating Earth system functions, especially the addition of chemically diverse and often harmful dust to ecosystems and toll it has taken on public health. In this final chapter and discussion, I have questioned the role of settler academics in researching modern environmental issues—such as land-use related PM and dust emissions—without explicitly confronting the colonial systems which created the harms.

Especially as climate change and environmental destruction have reached stages with potentially no return, colonial frameworks of control and manipulation cannot be used to explore solutions, which can only perpetuate harms on indigenous communities (Whyte, 2020). This calls for a critical view on methodology and the types of questions asked by researchers and shared with the public. More importantly, it questions the production of knowledge—even if seeking to address injustices—that uses and furthers colonial land relations.

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