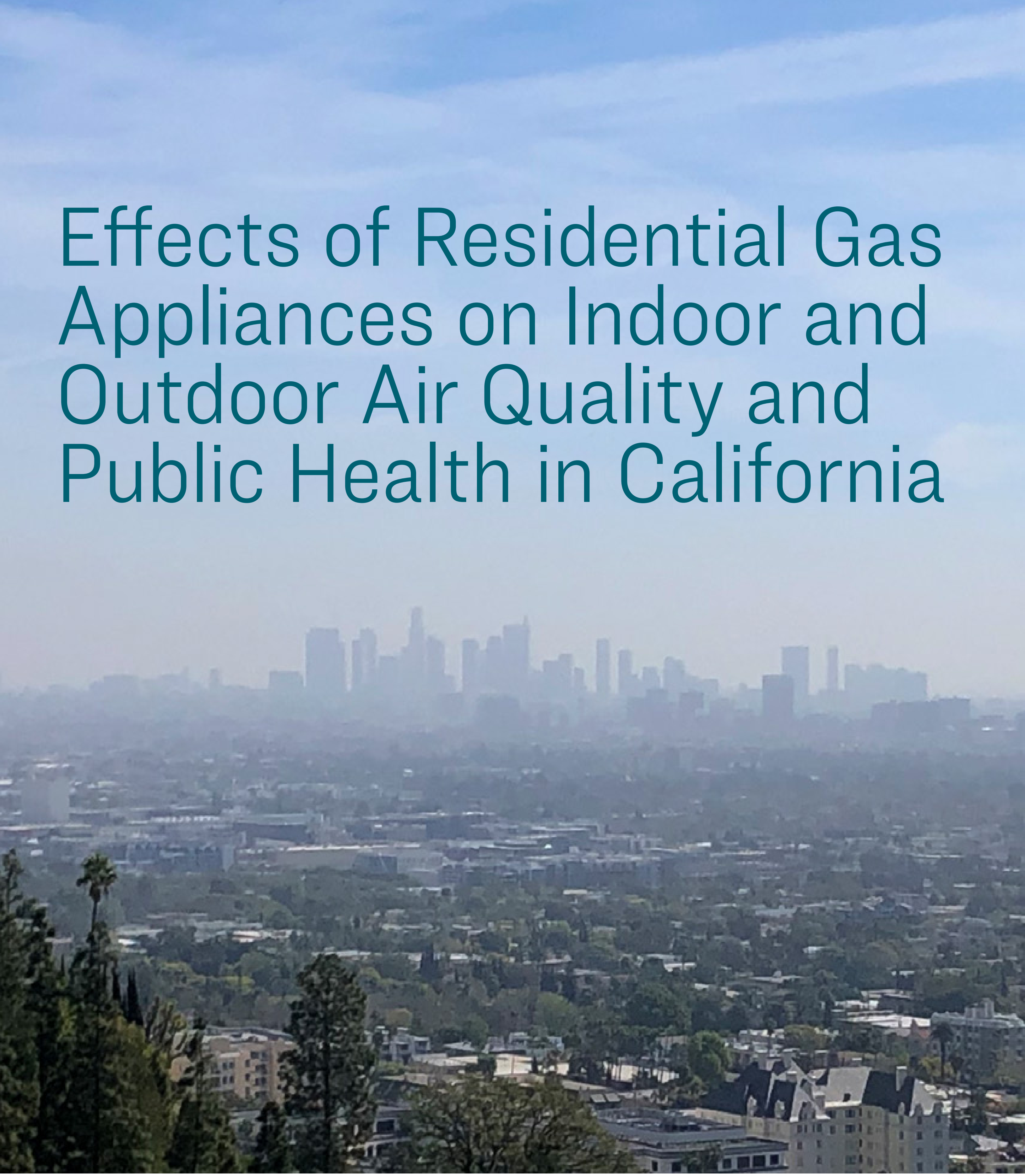
# Effects of Residential Gas Appliances on Indoor and Outdoor Air Quality and Public Health in California



UCLA Fielding School of Public Health
Department of Environmental Health Sciences

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# AUTHORS

**Dr. Yifang Zhu** (Principal Investigator)

**Rachel Connolly**

**Dr. Yan Lin**

**Timothy Mathews**

**Zemin Wang**

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Cover: A view of downtown Los Angeles from Hollywood Hills, blanketed in smog the afternoon of March 5th, 2020. Photo by Kristiana Faddoul, Sierra Club

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# ABBREVIATIONS AND UNITS LIST

| ABBREVIATIONS/ ACRONYMS | DESCRIPTIONS* |
|---|---|
| AB | Assembly Bill |
| ACS | American Community Survey |
| AER | Air exchange rate |
| AHS | American Housing Survey |
| BenMAP | U.S. EPA's Benefits Mapping and Analysis tool |
| BTU | British thermal unit |
| CAAQS | California Ambient Air Quality Standards |
| CARB | California Air Resources Board |
| CDC | Centers for Disease Control and Prevention |
| CEC | California Energy Commission |
| $CH_4$ | Methane |
| CO | Carbon monoxide |
| $CO_2$ | Carbon dioxide |
| COPD | Chronic Obstructive Pulmonary Disease |
| DTSC | California Department of Toxic Substances Control |
| E3 | Energy and Environmental Economics, Inc. |
| EF | Emission factor |
| EPRI | Electric Power Research Institute |
| GHG | Greenhouse gas |
| h | Hour |
| HHRA | Human Health Risk Assessment |
| HQ | Hazard Quotient |
| I/O | Indoor/outdoor ratio |
| IARC | International Agency for Research on Cancer |
| ISA | Integrated Science Assessment |
| J | Joule |
| LBNL | Lawrence Berkeley National Laboratory |

| ABBREVIATIONS/ ACRONYMS | DESCRIPTIONS* |
|---|---|
| MBR | Mass burning rate |
| mg | Milligram |
| ng | Nanogram |
| NAAQS | National Ambient Air Quality Standards |
| NO | Nitric oxide |
| $NO_2$ | Nitrogen dioxide |
| $NO_X$ | Nitrogen oxides |
| µg | Microgram |
| OEHHA | Office of Environmental Health Hazard Assessment |
| ppb | parts per billion |
| ppm | parts per million |
| PAH | Polycyclic aromatic hydrocarbon |
| PERE | Program for Environmental and Regional Equity |
| PM | Particulate matter |
| $PM_{2.5}$ | Fine particulate matter |
| PN | Particle number |
| RASS | Residential Appliance Saturation Study |
| REL | Reference Exposure Level |
| SB | Senate Bill |
| SFH | Single-family home |
| $SO_2$ | Sulfur dioxide |
| UCI | University of California, Irvine |
| USC | University of Southern California |
| UFP | Ultrafine particle |
| US EPA | United States Environmental Protection Agency |
| VOC | Volatile organic compound |
| VSL | Value of a statistical life |
| WHO | World Health Organization |
| WTP | Willingness to pay |

** Links to the associated databases are embedded wherever applicable.*

# EXECUTIVE SUMMARY

The electrification of residential buildings refers to the transition from fossil-fuel-powered appliances to electric technologies. Dozens of cities in California have already passed electrification policies to ensure new constructions within their jurisdictions are built all-electric. State regulatory agencies and utilities are pursuing programs and policies to support residential and commercial building electrification as part of meeting the state's climate and energy goals.

There has been considerable focus on building electrification's potential to reduce greenhouse gas emissions, and less focus on how electrification can also yield significant air quality and public health benefits.

California currently faces a global pandemic in which a rapidly spreading coronavirus disease, COVID-19, can cause severe respiratory illness and even death. New evidence suggests that a small increase in long-term exposure to fine particulate matter ($PM_{2.5}$) leads to a large increase in the COVID-19 death rate; this further establishes the substantial value in protecting the population from the respiratory vulnerability caused by widespread air pollution.

Exposure to the pollutants produced from gas appliances can be detrimental to human health; thus, one significant benefit of replacing natural gas (hereafter referred to as "gas") appliances with electric appliances would be the elimination of indoor air pollution that comes from burning gas indoors. This report aims to better understand the health concerns associated with gas appliance use, as well as the health benefits of phasing out residential gas appliances in California.

To systematically evaluate the impact of gas appliances on indoor air quality (Section 2), we developed an emission factor (EF) database, provided an estimate of indoor air pollutant concentrations due to gas appliance usage, and characterized the associated health impacts. Next, we evaluated the potential health co-benefits resulting from changes to ambient (outdoor) air quality related to residential gas appliance electrification (Section 3). This was accomplished by estimating the total emission of outdoor air pollutants in California due to the use of household gas appliances, the reduction in emissions due to residential building electrification under a modeled transition scenario, the resulting reduction of premature deaths and cases of acute and chronic bronchitis in California, and monetization of those health benefits. A detailed description of the data and methods can be found in Appendix A.

## Key Findings

### INDOOR AIR QUALITY

- Gas appliances emit a wide range of air pollutants, such as carbon monoxide (CO), nitrogen oxides ($NO_x$, including nitrogen dioxide ($NO_2$)), particulate matter (PM), and formaldehyde, which have been linked to various acute and chronic health effects, including respiratory illness, cardiovascular disease, and premature death.
- Under a hypothetical cooking scenario where a stove and oven are used simultaneously for 1 hour, peak concentrations of $NO_2$ from cooking with gas appliances exceed the levels of acute national and California-based ambient air quality thresholds in more than 90% of modeled emission scenarios.
- Concentrations of CO and $NO_2$ resulting from gas cooking are the highest for apartments, due to a smaller residence size. This presents an additional risk for renters, who are often low-income.
- Increases in indoor air pollutant concentrations can be driven by insufficient ventilation. Surveys show that less than 35% of California residents use range hoods when cooking — and many homes in the U.S. are lacking range hoods or ventilation altogether.
- The use of kitchen appliances for supplemental heating can increase exposure risks, and there is evidence this disproportionately affects low-income households, though more data on the frequency of use is needed to quantify the risk to various populations.

- Environmental justice communities disproportionately experience poor housing conditions which can be detrimental to health. Concerns related to gas appliance use include: the presence of old and unmaintained appliances in households, smaller and overcrowded residences where air pollution can reach higher concentrations, and challenges faced by renters to control appliance choices or afford maintenance. These populations already face cumulative effects associated with health and environmental injustices more broadly, and gas appliance issues can compound this. There are significant data gaps regarding equity and the health effects of gas combustion on low-income and minority populations, which should be further explored to facilitate a just transition to a low-carbon future.
- Better regulations and safeguards are needed to protect residents from exposure to indoor air pollution from gas appliances. Along with replacing gas kitchen appliances with electric alternatives, increasing the frequency of range hood use and improving the efficacy of ventilation technology would also reduce exposure and protect public health.

OUTDOOR AIR QUALITY

- Gas appliances are also a source of outdoor air pollution, and literature shows that the pollutants released by combustion can lead to illness and premature death.
- Using the EFs developed in this study's indoor air quality analysis, and assuming all indoor emissions are transported to the outdoor environment, we find that approximately 12,000 tons of CO and 15,900 tons of $NO_X$ (see Figure 3-1 in Section 3.2.1) were emitted to outdoor air from the use of residential gas appliances in California in 2018.
- If all residential gas appliances were immediately replaced with clean electric alternatives, the reduction of outdoor $NO_X$ and $PM_{2.5}$ would result in 354 fewer deaths, as well as 596 fewer cases of acute bronchitis and 304 fewer cases of chronic bronchitis annually in California (Table 3-1). This is equivalent to approximately $3.5 billion in monetized health benefits over the course of one year. These numbers only account for exposures from outdoor air as a result of residential electrification; a full exposure assessment accounting for indoor exposures would increase the total health benefits and the associated economic benefits of residential electrification.

In summary, this report contributes to a growing body of research quantifying the air quality and health impacts from the use of gas appliances in households, and highlights several potential benefits, both health-related and economic, of residential electrification throughout the state of California. While this report provides an estimate of emissions, and resulting emission reductions from discontinuing the use of gas appliances in residences, it does not consider the full spectrum of costs and benefits associated with residential building electrification. Policymakers and stakeholders are encouraged to use this report, alongside existing research on building decarbonization, electrification, and other related topics, as a tool to develop stronger regulations and protections that limit indoor and outdoor air pollution from gas appliances, and to support new policy development to improve public health, particularly for communities disproportionately burdened by pollution from fossil fuels.

# 1 INTRODUCTION

## 1.1. CALIFORNIA'S GAS CONSUMPTION AND TRANSITION TO CLEAN ENERGY

Natural gas (hereafter referred to as "gas") is one of California's primary energy sources.[1] It is a fossil fuel consisting of mostly hydrocarbons, the majority of which is methane ($CH_4$) - a potent greenhouse gas (GHG). In 2018, more than 2.1 trillion cubic feet of gas were consumed in California, which accounts for 7.1% of gas consumption within the entire United States.[1]

In California, gas is used to fuel power plants and certain industrial processes, and in buildings for heating and cooking. In residences, common gas-powered appliances include stoves, ovens, furnaces, water heaters, clothes dryers, and fireplaces. Results from the American Housing Survey (AHS) indicate that more than 90% of California households use gas for at least one purpose, and almost 70% of households use gas for cooking.[2]

While demand for gas in the power sector is expected to drop dramatically as the state implements Senate Bill (SB) 100, which calls for 100% carbon-free electricity, there is no current statewide policy to address the gas that is burned inside California's buildings, even though consumption by residential and commercial buildings accounts for 31% of gas use within the state.[1] The residential sector alone accounts for more than 20% of the state's gas use.[1] Research also indicates that residential appliances alone constitute 15% of California's $CH_4$ emissions from gas,[3] and overall, buildings are responsible for an estimated 25% of all GHG emissions in California.[4,5] Two-thirds of these are caused by onsite combustion of fuel, including gas.[5] While this report focuses on California, it is worth noting that the climate effects of gas use in buildings are of national significance. The United States Environmental Protection Agency (US EPA) reported that in 2017, gas consumption comprised 89% of direct, fossil-fuel carbon dioxide ($CO_2$) emissions from the residential and commercial sectors.[6]

California is a national leader in clean energy and climate policy, and has mainly pursued new policies and programs to promote building electrification (i.e., the transition from fossil-fuel-powered appliances to electric technologies) as a climate mitigation strategy.[4,7,8] State and local agencies have not, for the most part, regulated gas appliances or promoted electrification to explicitly improve air quality and public health, although agencies and reports have noted cleaner air and improved health outcomes as a co-benefit of building decarbonization.[9]

This research was commissioned to inform the potential air quality and health benefits of stricter regulation of gas appliances with a goal of improving the state's air quality and population health, and to better understand the co-benefits of building decarbonization.

Much recent research surrounding the use of gas as an energy source focuses on emissions of GHGs, such as carbon dioxide ($CO_2$) as a result of combustion, as well as the leakage[i] of $CH_4$.[3,14] Organizations such as Energy and Environmental Economics, Inc. (E3) and the Electric Power Research Institute (EPRI) have conducted comprehensive research for the California Energy Commission (CEC) and utility companies on decarbonization and building electrification. Findings indicate that California residential building electrification is a cost-effective GHG mitigation tool under many circumstances, and would often result in reduced consumer household energy costs.[4,7]

i. Though gas has been considered a transitional fuel because it emits less GHGs than other fossil fuels[10], evidence indicates that the leakage of methane negates the climate benefits of burning gas.[11] In addition, the use of fracking for the extraction of gas results in methane leaks and incurs severe environmental and health costs; the chemical additives involved in fracking are highly toxic.[11–13]

## 1.2. GAS APPLIANCE USE AND ASSOCIATED AIR QUALITY AND HEALTH OUTCOMES

Another key area in the literature explores the extent of criteria pollutants, such as carbon monoxide (CO), nitrogen dioxide ($NO_2$), and particulate matter (PM) produced from gas combustion, and the health co-benefits from GHG reduction tactics due to the reduction of these criteria air pollutants, which have a more localized and substantial impact on public health than GHGs.[15] Gas has been marketed as a relatively clean fuel because it emits less criteria pollutants compared to other fossil fuels, such as coal and oil,[16,17] as well as the burning of biomass.[18,19] However, there are significant risks associated with the burning of gas in residences, due to the indoor emission of pollutants, such as CO and formaldehyde (from incomplete combustion), as well as nitrogen oxides ($NO_X$) such as $NO_2$ (caused by the oxidation of nitrogen during combustion). Other hazardous compounds emitted from the burning of gas inside homes include volatile organic compounds (VOCs), sulfur oxides, and PM.[20] The resulting indoor air pollution can have adverse effects on human health, as Americans spend almost 90% of their time indoors,[21] and a substantial body of literature has established that both indoor and outdoor air pollution are notable threats to human health.[22–24] While many studies have been published on the various sources and effects of indoor air pollution, the extent of the effects of residential exposure to combusted gas on human health merits additional analysis.

Many studies have measured emissions and concentration levels of $NO_2$, $NO_X$, PM, formaldehyde, and other compounds while appliances were in use. Overall, studies have reported higher concentrations of CO and $NO_2$ in homes that use gas rather than all-electric appliances. Although electric appliances do not generate emissions from combustion, they can produce emissions from other sources, such as the cooking of food, or dust on the heating surface.[23,25–28] There are also numerous studies on cooking activities, which also provide evidence of emissions from gas stove and oven use, although some emissions are related to the cooking style and type of food being cooked, and not the fuel that is being used.[26,28,29] Additionally, studies on the association between gas appliance use and health have mixed results, in part due to study design limitations, but also due to a lack of data on quantified exposures.[30,31] The nature of this uncertainty is described in further detail in Section 2.1. Studies have also demonstrated associations between gas stove use and increased respiratory symptoms for household

residents, particularly children.[32–34] Notably, children had lower odds of suffering from asthma and bronchitis in households where adults used ventilation when operating gas stoves.[35] The California Department of Public Health recommends increasing outdoor ventilation and using exhaust fans when cooking with gas stoves, and avoiding the use of illegal, fuel-burning, unvented gas space heaters.[36]

Environmental health burdens associated with gas appliance use can disproportionately affect low-income individuals, who are often renters with less control over appliance installation and maintenance,[37–39] and typically living in smaller units, which can result in elevated pollutant concentrations.[40,41] These issues compound the cumulative health effects in environmental justice communities, populations which currently — and historically — have borne a disproportionate burden of environmental health risks.

## 1.3. SCOPE OF RESEARCH

Considering existing knowledge, as well as specific knowledge gaps, this research builds upon work from other organizations, including the Lawrence Berkeley National Laboratory (LBNL), the California Air Resources Board (CARB), the CEC, E3, and EPRI, to conduct a wide-ranging evaluation of the effects of gas appliance use on both indoor and outdoor air quality, the associated health effects and susceptibility of populations, and the potential benefits of electrifying residential gas appliances throughout the state. We endeavor to consider how these issues impact low-income and environmental justice communities throughout the report.

This report is a synthesis of relevant literature and data, incorporating literature primarily focused on the 21st century in California, and including new secondary analyses and modeling to the extent possible. Our work is novel in its combination of varying approaches of evaluating gas appliances, including: (1) modeling a variety of gas cooking exposure scenarios and conducting sensitivity analyses accounting for pollutant spillage into the indoor environment from four types of appliances; (2) an evaluation of exposure and vulnerability considerations, including equity-related concerns associated with gas appliance use; (3) an in-depth aggregation of current data on the health impacts of pollutants associated with gas combustion; (4) a quantitative assessment of outdoor emissions resulting from gas appliance use; and (5) a health impact and monetary valuation assessment of exposure to those outdoor emissions. This assessment aims to help the general public and policymakers to better understand the potential health impacts associated with gas appliance use, as well as the health benefits of phasing out residential gas appliances in California.

In the rest of the report, we separate our analysis into two main sections: indoor air quality and health effects, and outdoor air quality and health effects. We start with a detailed background section, then present results and discussion for each section. We conclude with a summary of our findings. The data and methodology can be found in Appendix A.

# 2 INDOOR AIR QUALITY AND HEALTH EFFECTS

Section 2 is focused on indoor air quality and health effects. Section 2.1 provides background information from a literature review, and Section 2.2 contains the results and discussion. The objectives of this section are to:

- quantify emission factors of CO, $NO_2$, and $NO_X$ from gas appliances in California;
- evaluate the impact of gas appliance usage on indoor levels of CO, $NO_2$, and $NO_X$, as well as their associated health effects; and
- qualitatively assess the vulnerability of specific populations to indoor air pollution exposures from gas appliance usage, through an equity lens.

In this section of the report, we quantitatively focus on three pollutants: CO, $NO_2$, and $NO_X$. We do not include other pollutants, such as formaldehyde and PM [including ultrafine particles (UFP, particles less than 0.1 micrometer in size), and fine PM with an aerodynamic diameter less than or equal to 2.5 micrometers ($PM_{2.5}$)], due to data paucity and feasibility. However, to the extent possible, we do qualitatively assess the emissions and associated health effects of these additional pollutants. Therefore, formaldehyde and PM are mentioned throughout the background (Section 2.1), and exposure is assessed in various portions of the Results & Discussion (Section 2.2).

## 2.1. BACKGROUND

This section provides general background information from the literature, and further findings based on modeling are presented in the results section. In this literature review section, we:

- **explore** the relationship between gas appliance use and indoor air quality by describing findings from relevant research studies and reports (2.1.1);
- **define** factors associated with pollutant emissions (2.1.2) and resulting indoor air concentrations (2.1.3), such as appliance ventilation and maintenance, and house volume and air exchange rate (AER);
- **clarify** the significance of the relationship between indoor air pollution and human health, particularly with respect to environmental justice communities (2.1.4); and
- **identify** current knowledge gaps and the contribution of this report (2.1.5).

### 2.1.1. THE RELATIONSHIP BETWEEN GAS APPLIANCE USE AND INDOOR AIR QUALITY

***Formation of combustion pollutants***

Combustion pollutants are produced from the use of gas appliances, including water heaters, stoves, ovens, furnaces and other indoor heating devices, such as gas fireplaces. Notable pollutants include CO, $NO_2$, $NO_X$, formaldehyde, and PM, including UFP and $PM_{2.5}$, though there are several other pollutants associated with residential gas combustion (see Table B-1 in Appendix B regarding pollutants emitted from gas combustion and their associated concentrations in the indoor environment). Although we were unable to quantitatively analyze all the pollutants emitted by combustion appliances, Table B-1 illustrates the wide range of pollutants produced. It excludes the pollutants subjected to subsequent quantitative analysis in this report — CO, $NO_2$, and $NO_X$ — pollutants with known health effects that also have enough publicly available combustion emission data to conduct analysis.

Combustion-related pollutants are primarily formed by the processes of incomplete combustion or oxidation. Incomplete combustion occurs when there is insufficient oxygen available to complete the combustion of fuel, resulting in byproducts such as CO and formaldehyde. In order to facilitate "complete" combustion, the proper amount of gas and air must be supplied at the correct pressure. However, incomplete combustion and its associated byproducts are unavoidable, even under ideal conditions.[42] Regarding oxidation, a prevalent example resulting in combustion pollution formation is the oxidation of nitric oxide (NO) to $NO_2$. At high temperatures, NO is formed by the combination of nitrogen with oxygen then oxidized to become $NO_2$.

***Overview of types of gas appliances***

Out of all commonly used gas appliances, water heaters and home heating devices such as furnaces are responsible for the majority of gas use in households, and thus, emit a larger proportion of combustion pollutants than gas kitchen appliances (stoves and ovens).[3,15,42–44] However, kitchen appliance emissions have a more significant effect on indoor air quality, as the heating appliances are vented outdoors and those emissions are generally considered to be outside the building envelope. Depending on the type of appliance and associated features, pollutants are either emitted directly into the living space, mitigated with ventilation technology such as range hoods, or directly vented outdoors (typically water heaters and furnaces). Ventilation effectiveness, which usually depends on appliance quality and maintenance, is discussed further in Section 2.1.2. There are regulations surrounding the use of gas appliances in households, including requirements for heating devices and water heaters to be vented outdoors, and prohibiting the sale of unvented gas space heaters. We discuss these regulations further in Section 2.1.2.

***Pollutant emissions and indoor air concentrations associated with gas appliance use***

Organizations such as LBNL and CARB have conducted research on the topic of gas appliance emissions in California specifically. They have studied the effectiveness of ventilation technology (including the use of range hoods), issues such as backdrafting (the backward movement of exhausted gases through the venting system) into residences from ventilation ducts and pollutant spillage (described in more detail in Section 2.1.2), and increasing the energy efficiency of appliances, and have modeled indoor air quality and population exposures resulting from the use of these appliances. [27,44–53] Additionally, researchers from other institutions have conducted various studies on these topics over the last several decades, from measurement of indoor air concentrations to simulating concentrations from gas appliance use, both in California and in other regions.[25,26,54–60] Studies measuring pollutant emissions or indoor air concentrations have consistently found that the use of gas appliances can result in concentrations of pollutants — particularly $NO_2$ — at concentrations above the level of ambient (outdoor) air quality standards.

Specifically, studies of California residential buildings have examined the association between gas appliances and measured indoor levels of air pollutants, including CO, $NO_2$, $NO_X$, $PM_{2.5}$, UFP, polycyclic aromatic hydrocarbons (PAHs), and formaldehyde.[44–46,48–50,52] While the majority of research focuses on cooking appliances, such as stoves and ovens, studies have also measured pollutant emissions or resulting indoor air pollutant concentrations from the use of heating appliances, such as furnaces, space heaters, water heaters, and fireplaces.[44,54,56,58,60–63] Furthermore, although many studies have measured $PM_{2.5}$ and UFP emissions from cooking with various types of food and cooking oil, these particulate emissions were often attributed to the food and cooking method rather than the operation of gas appliances.[55,57,64]

Several studies have found gas stove usage results in both peak and weekly average $NO_2$ concentrations exceeding the level set by both the chronic California Ambient Air Quality Standards (CAAQS) ambient annual average limit of 57 micrograms per cubic meter ($\mu g/m^3$), and the acute National Ambient Air Quality Standards (NAAQS, set by the US EPA) 1-hour limit of 188 $\mu g/m^3$ or 100 parts per billion (ppb). Please refer to Section 2.2.2 for relevant considerations regarding the use of outdoor air quality standards to assess exposure to indoor air quality.[27,45,46,49,52] Studies of California residential buildings have reported $NO_2$ levels in excess of these standards in kitchens and bedrooms, suggesting elevated concentrations throughout the entirety of the home during a single instance of gas cooking, especially in homes using gas stoves with pilot lights, or without venting range hoods.[49,50,52] Research from earlier decades on unvented gas space heaters measured $NO_2$ and $PM_{2.5}$ concentrations above standards, but these types of heating devices are no longer legally sold in California.[58,60,65,66]

Research has found that CO is a lesser potential health concern than $NO_2$ if appliances are operating properly. Gas stoves have been associated with increased levels of indoor CO in California homes, but these increases in concentrations are generally negligible,[27,49,51,52] with only a small portion of homes exhibiting CO concentrations above the CAAQS 1-hour standard of 23 milligrams per cubic meter (mg/m$^3$).[45,46] However, CO concentrations above the CAAQS 8-hour standard of 10 mg/m$^3$ have been reported during preparation of a full meal and under broiling conditions, without range hood use (though these were peak values and these concentrations did not persist for an entire 8-hour period).[27] Furthermore, CO emissions may rise to higher concentrations under conditions where appliance ventilation mechanisms fail or are not used, or the stove is misused for heating residences, and we address the former in Section 2.1.2. CARB reports that CO is responsible for 13 to 36 deaths from non-fire-related CO poisoning in California each year since 2000.[67]

As mentioned at the beginning of Section 2, although we do not include PM or formaldehyde in our quantitative assessment, it is important not to overlook these pollutants when considering the effect of gas appliances on air quality. Similarly to CO, studies measuring $PM_{2.5}$ emissions found that increases attributed solely to gas kitchen appliances (with no cooking of food involved, though sometimes a pot of water was heated) were negligible.[49,52] One caveat mentioned previously is that cooking can be a significant source of exposure to $PM_{2.5}$ due to heating and combustion of food and cooking oil, resulting in indoor concentrations far in excess of the NAAQS 24-hour threshold of 35 μg/m$^3$.[27,55,57] Furthermore, studies have measured substantial peak UFP concentrations during gas stove cooking, both with and without food.[28,30,44,48,57,68,69] All studies including tests of gas stoves used without food demonstrated elevated UFP concentrations.[28,44,68] Emissions from episodic sources such as cooking, with either gas or electric stoves, constitute a majority of indoor UFP concentrations.[30,48,68,70–72] UFP concentrations during episodes of cooking without a range hood are comparable to those found outdoors on high pollution days.[28] Since no government standards for UFP concentrations currently exist, and the health effects of UFPs are not yet fully characterized, it is challenging to regulate these smaller particles. Though we do not quantitatively evaluate UFP in this report, we discuss the health effects of UFP exposure in Section 2.2.4. Studies have also estimated particle emissions from other types of gas appliances, such as water heaters and home heating devices, but most assess particle emissions in the units of particle number (PN, which better reflects UFP levels) and not $PM_{2.5}$. Based on CARB's annual projections of county-level, estimated total emissions for $PM_{2.5}$ from residential gas combustion, and as seen for other pollutants as well, water heaters and home heating appliances have significantly higher overall emissions than gas cooking appliances. However, water heaters and home heating appliances are vented outdoors (outside the building envelope), as mandated by regulations.[73]

Gas appliances also emit formaldehyde,[27,44,62] but some studies did not find a statistically significant association between gas appliance use and indoor formaldehyde concentrations.[45,46,74] A CARB analysis reported formaldehyde concentrations far above the acute Reference Exposure Level (REL) of 55 μg/m$^3$

set by the California Office of Environmental Health Hazard Assessment (OEHHA) during gas cooking, both with and without food.[27] An REL is the maximum concentration at which not to expect any adverse, non-cancer health effects at each given exposure duration (acute, 8-hour, or chronic). However, an LBNL study of California homes found that although 95% of homes tested had formaldehyde concentrations above the OEHHA chronic REL, these levels were not statistically significantly associated with gas appliances.[45] In addition, gas appliances emit acetaldehyde,[27,44] a highly toxic and carcinogenic VOC similar to formaldehyde, with recent research indicating low levels emitted from gas stove burners.[44] Due to the lack of emission data and statistically significant evidence reported in the primary literature, we did not include formaldehyde or acetaldehyde in our quantitative analysis.

Besides experimental research, several simulation studies have modeled gas appliance emissions and reported exposures to indoor pollutants, including CO, $NO_2$, and formaldehyde.[47,53,59] Simulation studies specific to California residential buildings found that gas stove emissions comprise up to a third of total weekly average concentrations of indoor CO and $NO_2$, and even conservative estimates of indoor CO and $NO_2$ concentrations may frequently be in excess of the 1-hour and 8-hour CAAQS standards for CO and the 1-hour NAAQS standard for $NO_2$. [47,53] One study estimated that in a typical winter week, 12 million and 1.7 million Californians may be exposed to $NO_2$ and CO levels (respectively) in exceedance of acute, ambient air quality standards.[47] Furthermore, the study estimated that formaldehyde emissions from gas cooking appliances alone would lead to exposures exceeding the OEHHA acute RELs (for approximately 50% of homes) and chronic RELs (for less than 10% of homes), depending on the season.[47]

### 2.1.2. FACTORS INFLUENCING EMISSIONS TO THE INDOOR ENVIRONMENT

Apart from the frequency of appliance use, as well as trends toward reduced heating (and increased cooling) demand for California buildings, there are several important factors influencing the quantity of emissions to indoor residences.

***Appliance ventilation conditions***

The effective removal of combustion products generated by gas appliances is a core element of health and safety in buildings. Home heating devices and water heaters must have their exhausted gases moved through the appliance, out of the venting apparatus, and into the outdoors.[75] The National Fire Protection Association 54: National Fuel Gas Code provides safety requirements for the ventilation of gas appliances and requires that gas space heaters and water heaters be vented to the outdoors, while the California Health and Safety Code prohibits the sale of unvented gas space heaters, and mandates the existence of ventilation equipment above stoves and ovens.[76,77] Even with such legislation in place, unvented or inadequately vented gas cooking appliances are present in some California homes.[78,79]

One significant concern regarding appliance ventilation failure is pollutant backdraft and resulting spillage, which put residents at greater risk of CO poisoning. Backdraft refers to the backward movement of exhausted gases through the venting system, and spillage refers to the resulting leakage of exhausted gases from the appliance into the indoor environment, which leads to the buildup of pollutants inside the home.[80] Although the frequency of backdrafting and spillage events is not well-established, this has led to excessive CO exposure, which has severe consequences: The Centers for Disease Control and Prevention (CDC) estimated 393 deaths in the United States from unintentional, non-fire-related CO poisoning associated with consumer products in 2015.[81] In California specifically, the Tracking California program (previously known as the California Environmental Health Tracking Program) estimated 643 emergency department visits due to non-fire-related CO poisoning in 2016.[82]

One main cause of backdraft and spillage is depressurization, which happens when air removed from the house by weather-related forces, open doors and windows, or mechanical appliances such as exhaust fans and furnaces, results in a lower air pressure indoors as compared to the outdoor environment.[80,83] Depressurization interferes with the mechanisms of combustion appliances, resulting in backdrafting and spillage. Depressurization is usually periodic rather than continuous,[83] although research has observed instances of continuous depressurization.[84] A literature review conducted by LBNL found that while up to 50% of appliances tested were at risk of backdrafting, few instances of “sustained” backdrafting or spillage were recorded.[80] There are several challenges associated with monitoring for backdrafting and spillage in homes.[80] Due to the existing limitations, questions regarding the frequency, duration, and severity of backdrafting and spillage events remain to be answered.

Low-income and elderly residents may face increased risk of CO poisoning from gas combustion appliances. A 2016 LBNL report on wall furnaces in apartments did find that backdrafting can occur frequently in small residences when kitchen and bathroom exhaust fans are


on a high setting, though this study had a relatively small sample size (16 apartments) and highlighted the need for additional research.[85] Nonetheless, this points to the potential for added risk for residents, including elderly populations, who live in smaller rental apartments, are often low-income, and face existing challenges with the burdens of appliance maintenance.[86,87]

Considering the uncertainty surrounding improper ventilation, one of our two sensitivity analyses (See Section 2.2.2 and Appendix A.1.2 for details) involved a scenario to account for the potential of indoor emissions from appliances, such as water heaters and home heating devices, that are designed to solely emit combustion pollutants outdoors. Of course, the location of the appliances, which can vary from wall furnaces in living rooms to water heaters in designated mechanical closets, are not aspects we were able to control for, but are important considerations for future research.

***Appliance maintenance considerations***

Maintenance issues can have a pronounced impact on the emissions produced by combustion appliances, as well as on ventilation efficiency. Old appliances that have not been maintained are at risk of ventilation failure, resulting in potentially dangerous levels of pollutants being emitted into the indoor environment.[44,61,80] Other problems requiring maintenance include heat exchanger failures and blocked flues in furnaces.

Appliance tuning, which refers to various aspects of appliance maintenance, can also have a substantial impact on emissions. Well-tuned appliances often emit substantially less CO than poorly tuned appliances, sometimes differing by an order of magnitude.[80,83,88–91] However, there are only limited studies on appliance maintenance and safety mechanisms, and these topics warrant further research.

### 2.1.3. FACTORS INFLUENCING COMBUSTION POLLUTANT CONCENTRATIONS IN AN INDOOR ENVIRONMENT

There are several factors that significantly affect the indoor air pollutant concentrations resulting from combustion.

***Range hoods and capture efficiency***

Gas stoves often lack adequate exhaust ventilation. The low-rise residential building ventilation code ASHRAE 62.2 requires the installation of range hoods in kitchens, with minimum airflow and maximum noise levels, but it is estimated that only half of new homes in the United States comply with this standard.[78] Furthermore, a study of California homes, using data from a real estate website, approximated that 47% of homes had combination microwave/range hoods, which do not meet the airflow and noise level requirements of ASHRAE 62.2, while 7% of homes had no range hoods at all.[79] A 2014 LBNL report highlighted a specific need for the development of over-the-range microwaves that meet the ASHRAE 62.2 requirements.[92] The body of research on the use and effectiveness of range hoods is growing.

Research on kitchen range hoods has demonstrated their potential to reduce exposure to pollutants emitted by combustion appliances, as well as evaluated noise levels, since sound is one reason why people often elect not to use range hoods.[45,52,57,68,92–97] Range hoods differ considerably in their ability to remove pollutants from the indoor environment and can be assessed using capture efficiency as a measure of overall effectiveness. Capture efficiency refers to the proportion of pollutants emitted from an appliance that are removed by the range hood before they enter the indoor environment.[52,93] The capture efficiency of range hoods is often above 50%, though it varies widely depending on the cooktop burner used (typically lower for front burners), as well as coverage and quality of the hood.[52,57,68,93,95] In one study of California residences, the use of range hoods resulted in significant reductions in air pollutant concentrations within the home.[52]

While studies have shown that range hoods can significantly reduce exposure, they are infrequently used and not always available or appropriately sized or installed.[78,79,92] Small-scale survey results show that less than 35% of California residents use range hoods when cooking,[98] while a CARB study of California homes found that 54% of participants did not use their range hood.[96] As mentioned previously, studies have shown that the excessive noise produced by many range hoods and fans is a primary reason for the lack of range hood use.[92,96,97] It is important to note that increased awareness of the need for ventilation during cooking and encouragement of range hood use may reduce exposures to pollutants emitted by combustion appliances for those with properly sized, installed, and maintained hoods. However, renters sometimes do not have range hoods installed, or existing hoods are not vented outdoors and may not meet standards, therefore putting renters at heightened vulnerability to exposure to air pollutants from gas cooking appliances.[99] Due to the infrequency of range hood use, our analysis assumed that there is no significant range hood use as a health-protective conservative assessment, though it is still useful to consider our estimates in the context of conditions involving range hoods as well, with varying levels of capture efficiency.

***Residence size and ventilation***

The size and ventilation of an indoor space are primary determinants of indoor air quality. In smaller residences, indoor air pollutants are distributed across a smaller space and thus, are more concentrated.[100–102] The volume of an indoor space is also a major factor in the determination of AER, which is expressed as the number of indoor air volumes replaced with outdoor air per hour.[102]

Ventilation (and AER) significantly influence indoor air quality. Inadequate ventilation has been associated with higher concentrations of indoor air pollutants, including $NO_2$, $PM_{2.5}$, and VOCs, as well as adverse health outcomes.[103–106] In fact, a recent study of commercial buildings in California determined that such buildings rarely meet ventilation standards.[107] There are reported challenges associated with meeting ventilation standards in multifamily housing as well.[108] A dilemma that has emerged in recent years, particularly with climate change considerations, is the dichotomy between promoting energy efficiency and improving indoor air quality. The tightening of building envelopes — essentially, residential air-sealing — has the potential to save billions in energy bills and reduce infiltration of outdoor air pollutants,[109] but it also decreases ventilation, degrading indoor air quality.[110] More energy-efficient buildings with tightened envelopes have, in some cases, been associated with adverse health outcomes due to worsened indoor air quality,[111,112] though a recent study on green buildings found several health benefits for individuals who moved from conventional housing to green-renovated housing.[106] Due to the crucial role of ventilation in determining indoor air quality, developments in building energy efficiency should be balanced with the preservation of indoor air quality.

### 2.1.4. WHY THIS ISSUE MATTERS: INDOOR AIR QUALITY, HUMAN HEALTH, AND ENVIRONMENTAL JUSTICE

As mentioned in the introduction, considering that people in the U.S. spend almost 90% of their time indoors, indoor air quality and human health are closely linked.[21] Many studies have assessed the health impacts of various indoor air pollutants.[30,113,114] We discuss each primary combustion pollutant (CO, $NO_2/NO_X$), as well as the pollutants we do not quantitatively evaluate (PM and formaldehyde) and their associated relationships with human health in detail in Section 2.2.4.

In the context of household gas appliances, the potential transition from gas to all-electric home appliances could benefit low-income households and environmental justice communities by improving both indoor and outdoor air quality. These communities face disproportionate air-pollution burdens[115] and limited access to clean energy resources.

While many issues related to gas appliance use and vulnerable populations are challenging to quantify without primary data collection, we aim to aggregate

as much of the relevant quantitative and qualitative information as possible on this topic as it connects to environmental justice and equity. A few key equity considerations related to gas appliance use, which we explore further in Sections 2.2.2 and 2.2.3, are as follows:

- SUPPLEMENTAL USE OF COOKING APPLIANCES FOR HEATING RESIDENCES. Though the frequency is not well-established, some research indicates that low-income and minority residents may disproportionately use kitchen appliances for the purpose of heating their residences (instead of using designated heating devices).
- HOUSING CHARACTERISTICS: TENURE, QUALITY, RESIDENCE SIZE, AND APPLIANCE MAINTENANCE. Residences occupied by low-income populations are often older, and use older, less efficient, and unmaintained appliances. These older appliances may not be regularly maintained due to the cost required and a lack of available funds to repair them, or lack of landlord attention.[37–39] Low-income residences are also likely to be smaller in size and have inadequate ventilation, resulting in higher indoor pollutant concentrations.[40,41]
- TIME-ACTIVITY PATTERNS. Time-activity patterns, or the amounts of time spent performing various activities throughout the day, substantially affect exposure to pollutants in various environments. Notably, children in low-income families may spend a greater amount of time at home and indoors than other populations.
- CUMULATIVE EFFECTS: HEALTH AND ENVIRONMENTAL JUSTICE IN CALIFORNIA. Residents in environmental justice or "disadvantaged" communities [defined by SB 535 as the top 25% scoring tracts in OEHHA's CalEnviroScreen tool, used for assessing environmental justice vulnerability] face some of the worst air quality in the state. Gas appliance emissions add to the persistent outdoor air pollution and can compound existing environmental burdens, placing low-income residents and people of color at even greater risk of adverse health effects from air pollution.

### 2.1.5. KNOWLEDGE GAP AND CONTRIBUTIONS TO THE LITERATURE

Based on the literature review, there is a clear need to: (1) aggregate information on related studies of gas appliances, indoor air quality, equity, and health; and (2) conduct data analysis to provide additional clarity on these issues through quantitative estimations.

While there is clear evidence of a relationship between indoor air quality and health, and combustion falls under that domain, there is some inconclusive literature related to gas appliance use and specific health effects. The broader relationship between $NO_2$ and adverse health effects is well-established,[116] but a recurrent theme in the literature is the uncertainty regarding the link between indoor $NO_2$ exposures from gas combustion and respiratory illness.[30,31,113,117] Challenges to the clarification of this relationship include the variabilities between appliances, use activity patterns, and home size and ventilation.[118] Studies have also highlighted the uncertainty regarding the relationship between residential indoor concentrations and personal exposure.[119] While several studies investigating gas appliances and asthma exacerbation produced mixed results, evidence supports a clearer association between gas appliances and asthma and respiratory symptoms in children,[33,120,121] with one meta-analysis reporting that children living in homes using gas for cooking have a 42% higher risk of having asthma.[33] While we did not estimate the association between specific health symptoms and use of gas appliances, our literature review and analysis aim to clarify the relationship between pollutants associated with gas appliance use and human health.

As described earlier, there are a limited number of recent studies that simulate and measure indoor air pollutant concentrations resulting from the use of gas appliances, and many are focused entirely on gas stovetop ranges. We used similar methods and data as some of those studies to conduct our analysis, but we included multiple types of appliances and conducted a detailed literature review on the use of gas appliances, related pollutants, and human health.

We modeled pollutant emissions, concentrations, and exposures resulting from gas appliance use in different housing types in California and linked these exposures to potential health effects via comparison to state and national standards. To our knowledge, there are no existing literature review and secondary analysis studies that tie together indoor air quality modeling for various pollutants, housing types, and low-income vulnerability in California.

## 2.2. RESULTS AND DISCUSSION

### 2.2.1. EMISSION FACTOR DATABASE

***Results of statistical analyses***

To model the effects of gas appliances on indoor air quality, we first created an emission factors (EF) database for residential gas appliances (see Appendix A, Section A.1.1 for details). Our regression models, designed to predict EFs in units of ng/J (nanogram/Joule), found that there are significant differences in EFs among gas appliance types.

Unsurprisingly, the EFs of gas appliances have declined over time, likely due to the technological advances of appliances and pollutant capture technology, which reduce emissions. Consistently, as the year of the publication from which EFs were gathered became more recent, the ng/J emissions decreased (e.g., a paper in 1995 would report higher emissions than a paper published in 2009, with a statistically significant difference); this indicates that emissions have reduced over time. For $NO_X$, there is a statistically significant increase in EFs for appliances designed to be vented outdoors (e.g., water heaters and home heating devices).

***EF and emission rate estimations***

As described in Appendix A, the EFs of gas appliances in a unit such as ng/J do not reflect the amount of pollutants released during the consumption without accounting for the mass burning rate (MBR, in J per time period) of different gas appliances. The emission rate in µg/hour (µg/h), or the amount of pollutants released in a specific time period during the usage of different gas appliances, is the product of the EF in ng/J and the MBR in J/h, since both factors affect the rate that combustion byproducts are released into the air.

EFs in ng/J, and emission rates in µg/h (the amount of pollutants released in a specific time period during the usage of different gas appliances) for each appliance category and pollutant were calculated as described in Appendix A.

**Table 2-1:** Mean emission factors (EF) and emission rates (ER) for each appliance type.

| Appliance Type | CO (mean) | | $NO_2$ (mean) | | $NO_X$ (mean) | |
|---|---|---|---|---|---|---|
| | EF (ng/J) | ER (µg/h) | EF (ng/J) | ER (µg/h) | EF (ng/J) | ER (µg/h) |
| Gas Stove | 52 | 670,000 | 10 | 130,000 | 38 | 440,000 |
| Gas Oven[ii] | 92 | 1,700,000 | 8.3 | 150,000 | 36 | 640,000 |
| Gas Water Heater[iii] | 18 | 3,200,000 | 3.4 | 490,000 | 25 | 2,300,000 |
| Gas Heater | 16 | 1,300,000 | 5.3 | 320,000 | 37 | 1,600,000 |

*Note: Values correspond to total emission factors and rates when the appliance is turned on, regardless of whether an appliance is vented outdoors (meaning not all these emissions travel indoors).*

ii. Separate EFs were calculated for stoves and ovens, but throughout this report we combined the two for most analyses (using a sum of emission rates), due to the nature of existing data (e.g. the RASS and CARB State Implementation Plan data). More specifics available upon request.

iii. This analysis incorporates both tankless and storage water heaters, which do have significantly different emissions for CO; tankless water heaters have higher emissions of CO. We did not control for these differences in our analysis. These higher emissions also occur for formaldehyde, which we did not quantitatively assess in this report.[44]

**Table 2-2:** Average indoor air concentrations by appliance – peak (highest concentration) vs. time-averaged 15-minute cooking, 1-hour cooking, and 2-hour cooking scenarios.

| Appliances | Pollutant | Average Peak Concentration (µg/m³) | Average Time-weighted Concentration (µg/m³) | Average Time-weighted Concentration (µg/m³) | Average Time-weighted Concentration (µg/m³) |
|---|---|---|---|---|---|
| Duration | | Peak | 15-minute cooking | 1-hour cooking | 2-hour cooking |
| Location | | Kitchen | Entire Residence | Entire Residence | Entire Residence |
| Stoves and ovens | CO | 18,000 *(16 ppm)* | 950 *(0.83 ppm)* | 2,600 *(2.3 ppm)* | 4,900 *(4.2 ppm)* |
| | $NO_2$ | 1,600 *(860 ppb)** | 16 *(9 ppb)* | 37 *(19 ppb)* | 64 *(34 ppb)** |
| | $NO_X$ | 6,700 *(3,600 ppb)* | 43 *(23 ppb)* | 130 *(69 ppb)* | 250 *(130 ppb)* |
| Stoves only | CO | 5,600 *(4.9 ppm)* | 550 *(0.48 ppm)* | 1,000 *(0.9 ppm)* | 1,700 *(1.5 ppm)* |
| | $NO_2$ | 750 *(400 ppb)** | 12 *(6.4 ppb)* | 22 *(11 ppb)* | 34 *(18 ppb)* |
| | $NO_X$ | 2,800 *(1,500 ppb)* | 26 *(14 ppb)* | 62 *(33 ppb)* | 110 *(58 ppb)* |

*Note: Values marked with * exceed acute CAAQS (for average peak concentration) for CO and $NO_2$, or 8-hour CAAQS for CO/chronic CAAQS for $NO_2$ (for time-weighted concentrations).*

Descriptive statistics from the results of our EF calculations are listed in Table 2-1. Kitchen appliances have higher EFs in ng/J for all pollutants, as compared with other gas appliances, but energy usage for water heaters and home heating devices is much higher (see Figure B-2 in Appendix B), which is why resulting emissions are higher for water heaters and home heating devices.[3,15,42,44,45] This is consistent with previous studies that have observed higher emissions from water heaters and home heating devices. Residential water heating results in the highest level of emissions of each of these pollutants.

## 2.2.2. INDOOR AIR QUALITY IMPACTS AND SUSCEPTIBILITY

***Indoor air quality model results***

As described in Appendix A, a mass-balance model[122] was used to estimate indoor air concentrations of CO, $NO_2$, and $NO_X$ under various scenarios of kitchen appliance use, including peak concentrations in the kitchen and time-weighted concentrations throughout the entire home (considering that cooking only occurs for a small portion of the day). This is described further in Appendix A (Section A.1.2), but the model produced an output with the highest concentration value, representing the emissions while cooking; we averaged these to establish the peak concentration levels presented in Table 2-2. For peak concentration levels, we used kitchen-specific volumes, and for values weighted by appliance usage time, we used entire residence volumes, under the assumption that pollutants would mix into the residential space over time. We are conservatively assuming there is no range hood use, and that all kitchen appliances are unvented.

Table 2-2 provides an overview of the average concentrations calculated using the model. We used a range of 15 minutes of cooking to 2 hours of cooking to represent a spectrum of potential cooking patterns.

***Defining exceedances of air quality thresholds***

Throughout this section, we used California (CAAQS) and U.S. EPA (NAAQS) ambient (outdoor) air quality **standards** as a metric for health effects from exposure.[iv] These standards are the maximum allowable concentration of a pollutant present in outdoor air that will not have a known, adverse impact on human health and are developed to apply to long-term, ambient outdoor air quality, averaged over time periods. It is not possible to actually exceed these outdoor standards in an indoor environment due to the technical definition.

Therefore, we apply target **thresholds** using the standards as a guide to provide context for indoor air quality. We refer to three different types of thresholds based on the standards: 1) acute (1-hour for $NO_2$ and CO), 2) 8-hour (for CO), and 3) chronic (annual mean for $NO_2$ — there is no annual mean standard for CO). When we use the term "acute," we are referring to 1-hour standards. For CO, we refer to 8-hour standards directly as such. For $NO_2$, when we use the term "chronic," we are referring to the annual mean standards.

When an exceedance is referenced in this report, it means that the modeled indoor air concentration is higher than the threshold levels based on the standards in Table B-7. When we refer to the percentage of exceedances, we are discussing the percent of our modeled indoor air quality estimates that exceed thresholds (please see Appendix A for additional details). We evaluate the indoor air quality exceedances of the

iv. An underlying assumption is that concentrations and exposures are directly proportional.

**Table 2-3:** Concentration and exposure scenarios.

| Pollutant | Scenario(s) | Time-Averaging | Location | Comparison |
|---|---|---|---|---|
| CO | Peak (assuming 1 hour of elevated concentration for exceedance to apply) | None | Kitchen | *Acute:* 1-hour thresholds |
| $NO_2$ | Peak (assuming 1 hour of elevated concentration for exceedance to apply) | None | Kitchen | *Acute:* 1-hour thresholds |
| CO | 15-minutes, 1-hour, and 2-hours of cooking | 8-hour | Entire residence | *8-hour* thresholds |
| $NO_2$ | 15-minutes, 1-hour, and 2-hours of cooking | 24-hour | Entire residence | *Chronic:* annual mean thresholds |

CAAQS and NAAQS thresholds, overall and for separate residence types, for CO and $NO_2$ (there are no applicable standards for $NO_X$).

We compare the peak concentrations (which are direct model outputs without any time-averaging, reflecting kitchen concentrations) from our indoor air quality model to the acute $NO_2$ and CO thresholds, under the assumption that exceedances of the thresholds for our estimated peak concentrations only apply under a scenario where cooking occurs for an extended period of time and the air quality levels in the kitchen remain elevated for an entire hour (considering the ambient air quality acute 1-hour threshold described above). We compare the modeled 8-hour time-averaged CO concentrations to the 8-hour CO thresholds, and the 24-hour time-averaged $NO_2$ concentrations to the chronic $NO_2$ thresholds, under three cooking-time scenarios (15 minutes of cooking, 1 hour of cooking, and 2 hours of cooking. This is laid out in Table 2-3.

We focus on the California CAAQS and U.S. NAAQS in this report, but we also note that Canada has existing indoor residential air quality guidelines for $NO_2$ that are more stringent than the U.S. thresholds we discuss throughout the report (i.e., 170 μg/m$^3$ for 1-hour, and 20 μg/m$^3$ for 24-hour).[123] Thus, the $NO_2$ results we present here can also be considered through the more health-protective lens of the Canadian standards. The Canadian CO standards are similar to the CAAQS and NAAQS thresholds. All of these standards can be found in Table B-7 in Appendix B.

***Findings: exceedances of air quality thresholds***

As shown in Table 2-2, for the use of both stoves and ovens simultaneously, the 2-hour use of kitchen appliances results in an average of the time-weighted NO concentrations of 64 μg/m$^3$, exceeding the chronic (annual mean) $NO_2$ CAAQS threshold of 57 μg/m$^3$. For 2 hours or less of stove use alone, the average household does not exceed any chronic thresholds.

Exceedances resulting from the use of both kitchen appliances simultaneously, as well as from stoves individually, are summarized in Table 2-4, with percent exceedances for both the CAAQS and NAAQS (see Appendix B, Table B-7 for each threshold level). As mentioned previously and in Appendix A, the kitchen peak concentrations are compared with the acute thresholds, and the modeled 15-minute, 1-hour, and 2-hour cooking, entire home concentration estimates have been time-averaged (over 24 hours for $NO_2$ and 8 hours for CO), and are compared with the chronic threshold for $NO_2$ and 8-hour threshold for CO.

When both stoves and ovens are used simultaneously, 18.7% of peak CO concentrations inside the kitchen exceed acute CAAQS, and 11% of 2-hour and 4.5% of 1-hour cooking averages throughout the home exceed 8-hour CAAQS. For CO, less than 1% of 8-hour concentrations based on a 15-minute cooking scenario results exceed the 8-hour CAAQS.

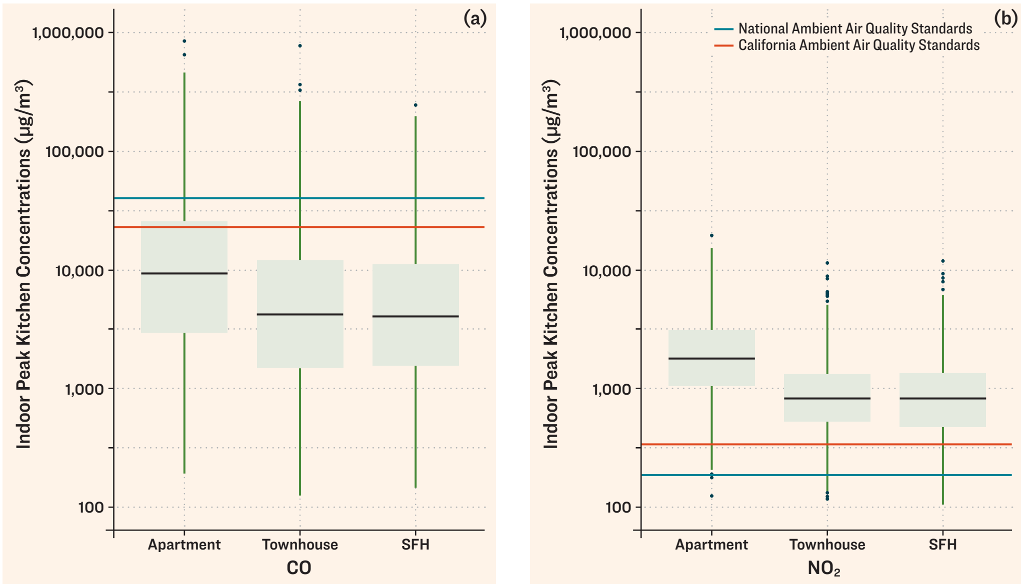
Results for $NO_2$ are even more noteworthy, particularly for peak concentrations. Again, when both stoves and ovens are used at the same time, more than 90% of

**Table 2-4:** Percent exceedances of air quality thresholds by appliances used and cooking time intervals.

| Appliance | Pollutant | Acute | | 8-hour for CO and Chronic for $NO_2$ | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | % of Peak Exceeding CAAQS | %of Peak Exceeding NAAQS | % of 15-minute Use Exceeding CAAQS | % of 15-minute Use Exceeding NAAQS | % of 1-hour Use Exceeding CAAQS | % of 1-hour Use Exceeding NAAQS | % of 2-hour Use Exceeding CAAQS | % of 2-hour Use Exceeding NAAQS |
| Stoves and ovens | CO | 19% | 11% | 0.4% | 0.4% | 4.5% | 4.5% | 11% | 11% |
| | $NO_2$ | 93% | 99% | 2.0% | 0.1% | 15% | 2.5% | 45% | 15% |
| Stoves only | CO | 4.4% | 2.6% | 0.1% | 0.1% | 1.0% | 1.0% | 2.7% | 2.7% |
| | $NO_2$ | 61% | 81% | 1.7% | 0.4% | 5.0% | 1.1% | 15% | 4.1% |

*Note: In the peak scenarios, when comparing concentrations to air quality thresholds and acute exposures, we assume cooking time occurred for the entire 1-hour period that the acute threshold applies.*

**Figure 2-1:** Peak concentrations in the kitchen resulting from usage of stoves and ovens simultaneously, by pollutant [(a) CO and (b) $NO_2$] and housing type.




peak $NO_2$ concentrations inside the kitchen exceed acute CAAQS. Additionally, 45% of 2-hour and 15% of 1-hour cooking averages throughout the home exceed chronic CAAQS, though only 2% of 15-minute cooking concentrations exceed chronic CAAQS.

When stoves are used independently, and resulting emissions and concentrations are lower, CO exceedances are again much less significant, while for $NO_2$, exceedances of acute NAAQS occur up to 80% of the time for peak concentrations, and exceedances of chronic CAAQS occur up to 15% of the time for the longest cooking time (2 hours), both based on the most stringent thresholds.

Keeping in mind the air quality thresholds, these percentages represent the frequency that air quality can no longer be considered safe (i.e., having no known, adverse impact on human health).

Figure 2-1 and Tables 2-5 and 2-6 provide housing-type-specific assessments. Figure 2-1 depicts average peak concentrations inside kitchens for each housing type, along with acute, ambient air quality standards for CO and $NO_2$. Variations in peak concentrations shown in Figure 2-1 are a result of variations in residence volume and associated ventilation rates. Our findings for apartments, townhouses, and single-family homes (SFHs) in Tables 2-5 and 2-6 are a disaggregation of the data in Table 2-4 and serve to inform the public about the variations in indoor air concentrations by building type. It must be noted that these findings are based on averages of housing volume and ventilation found in the literature, and may not represent all homes; therefore, these should only be used as general indicators. While Table 2-2 and Table 2-4 include a 15-minute cooking scenario, we did not include this particular scenario in the housing-specific tables (2-5 and 2-6) for simplicity. If the 15-minute cooking scenario is of further interest, Tables 2-2 and 2-4 can be used to extrapolate the data.

To appropriately interpret these tables and figures, we refer to housing estimates in California. Throughout the state, approximately 58% of residences are SFHs, 30% are apartments, and 9% are attached homes, such as townhouses.[2] The remainder (~3%) are primarily mobile homes, but because those make up such a small percentage of the California housing stock, we

**Table 2-5:** Average peak (kitchen) and time-weighted, 8-hour average (entire home) CO concentrations from use of gas kitchen appliances in various residence types, and percentage of scenarios in which concentrations exceed air quality thresholds.

| Residence Type | Acute - Peak | | 8-hour | | | |
|---|---|---|---|---|---|---|
| | Peak Conc. in Kitchen (µg/m3) | % of Cases Above 1-hour Standards | 8-hour Conc. 1-hour Cooking Entire Home (µg/m3) | % of Cases Above 8-hour CAAQS 1-hour Cooking | 8-hour Conc. 2-hour Cooking Entire Home (µg/m3) | % of Cases Above 8-hour CAAQS 2-hour Cooking |
| Apartment | 28,000 | 27.6% | 3,900 | 8.2% | 7,400 | 18.2% |
| Townhouse | 13,000 | 12.8% | 2,000 | 2.5% | 3,500 | 6.9% |
| SFH | 12,000 | 12.2% | 1,800 | 1.8% | 3,300 | 5.8% |

**Table 2-6:** Average peak (kitchen) and time-weighted 24-hour average (entire home) $NO_2$ concentrations from use of gas kitchen appliances in various residence types, and percentage of scenarios in which concentrations exceed air quality thresholds.

| Residence Type | Acute - Peak | | Chronic | | | |
|---|---|---|---|---|---|---|
| | Peak Conc. in Kitchen (μg/m3) | % of Cases Above Acute Standards | Time-weighted Conc. 1-hour Cooking Entire Home (μg/m3) | % of Cases Above Chronic CAAQS 1-hour Cooking | Time-weighted Conc. 2-hour Cooking Entire Home (μg/m3) | % of Cases Above Chronic CAAQS 2-hour Cooking |
| Apartment | 2,400 | 98.3% | 46 | 27.2% | 85 | 65.8% |
| Townhouse | 1,100 | 90.8% | 31 | 8.4% | 52 | 31.9% |
| SFH | 1,100 | 87.0% | 33 | 12.5% | 56 | 33.9% |

do not include them in this analysis. However, this is an important area for additional study. There are existing challenges with the provision of utilities in mobile home parks; the California Public Utilities Commission approved a Mobile Home Park Utility Upgrade Program in 2014, which is still operating.[124]

These concentrations and exceedances are comparable to the findings in previous studies, both modeled and measured, many of which are discussed in Section 2.1.1. One notable finding is that exceedances are all higher for apartments, primarily due to smaller residence sizes.

A 2012 analysis conducted by LBNL measured pollutant levels in 155 California homes and found that approximately 10% of residences studied had chronic (6-day, in the case of this study — typical passive monitoring methods only capture multiday $NO_2$ averages) indoor air concentrations of $NO_2$ exceeding chronic CAAQS, which are comparable to our 1-hour cooking results of 15% exceedances for stoves and ovens used simultaneously, and 5% for just stove use (Table 2-4).[45] They had a low number of exceedances for CO, which aligns with our findings (Tables 2-4 and 2-5) that CO exceedances are much less frequent than $NO_2$. A modeling analysis of Southern California homes found a similar pattern between CO and $NO_2$ exposures.[53]

We did find a high percentage of peak exceedances of acute (1-hour) thresholds for $NO_2$ when both stoves and ovens are used (Table 2-4 and 2-6), and associated exposures apply, assuming the household cooked for the entire hour, but this is not necessarily a typical exposure scenario; it depends on cooking habits in the home. When only stoves are used (Table 2-4), the risk assessment more closely matches existing literature on emissions from stovetop ranges. A simulation-based study of California residences, which incorporated seasonality, found that among homes using gas stoves with no ventilation (the same conditions as our analysis), 55-70% exceeded NAAQS $NO_2$ standards.[47] Though we found that almost all of our estimated concentrations exceeded the 1-hour $NO_2$ NAAQS threshold for concurrent stove and oven use, we found 80% exceedances when solely stoves are used (Table 2-4), which is comparable.

Overall, our analysis echoes the assertions of many existing studies that exceedances of regulatory standards for $NO_2$ may be frequent and are a cause for concern. Additional information on the pollutant levels measured and simulated in the literature is provided in Section 2.1.1.

***Sensitivity analysis: using kitchen appliances for heating residential spaces and the impact of improper ventilation***

In situations where designated heating devices are insufficient in a residence, kitchen appliances are sometimes used as heating devices. Additionally, as discussed in Section 2.1.2, ventilation technology does not always work exactly as designed. Due to the lack of existing data sources, we were unable to quantify the frequency of occurrences of the use of kitchen appliances for heating residential spaces, or the frequency of improper ventilation that results in pollutant spillage to the indoor environment. We have provided a brief analysis of existing knowledge to cover the spectrum of this potential issue.

Again, while it is challenging to quantify the frequency of these types of occurrences, there are many circumstances under which backdrafting and spillage of combustion pollutants can occur, and evidence suggests this may happen in homes across the United States (see Section 2.1.2). Under these various scenarios where appliances designed to be vented outdoors are improperly vented and combustion gases spill into the interior, peak exposures rise significantly, particularly because when used, these types of appliances have higher hourly emissions than kitchen appliances (see Table 2-1). One key assumption for this portion of the analysis is that the pollutant spillage is occurring for the entire time the appliance is operating. Due to the conservative nature of this assumption, we included several capture efficiency scenarios; we have provided this range of scenarios to account for various percentages of combustion pollutants traveling indoors instead of out through ventilation. These concentrations are for each appliance used individually, as opposed to concurrently.

Compared to the average peak concentrations presented in Tables 2-2, 2-5, and 2-6, the concentrations resulting from improper ventilation presented in Table 2-7 are elevated in several cases. For water heaters, these concentrations commonly exceed the acute CAAQS threshold for CO (two out of four capture efficiency scenarios) and $NO_2$ (all four scenarios). For $NO_2$, concentrations for home heating device use also exceed the acute CAAQS threshold for all but the highest capture efficiency scenario. These represent peak concentrations throughout the entire home, not just the kitchen (since there is not a specific, designated room for these other appliances); therefore, these are modeled on a larger volume than peak concentrations in the other tables. It is important to note that if the appliance of concern is in a small room, concentrations in that room will be significantly higher than what is recorded in Table 2-7.

There is limited information on the supplemental use of cooking appliances (stoves and ovens) for heating residences (meaning appliances not designed for heating are used to heat spaces in the home). When gas cooking appliances are used for heating, there are two particularly important considerations: the greatly increased duration of use, compared to when they are used for cooking, and that they constitute an unvented heating source. These factors elevate indoor combustion pollutant levels. A study investigating the relationship between respiratory illness in children, gas stove use, and ventilation found that in homes where adults used the stoves for both cooking and heating, as opposed to solely for cooking, children had significantly higher odds of being diagnosed with asthma and experiencing other respiratory symptoms.[35]

Under a scenario where kitchen appliances are used for supplemental heating for approximately 4 hours, time-weighted exposures in the entire home rise by a factor of 4.8 for $NO_2$ as compared to 1 hour of cooking, and the chronic CAAQS threshold is exceeded in 90% of instances. The 8-hour CO concentrations would rise by 2.8 times in this scenario. If only the stove is used for both cooking and supplemental heating, time-weighted exposures for $NO_2$ rise by a factor of 4.0 and exceed the CAAQS threshold 51% of the time when supplemental heating occurs. On average, the 8-hour CO concentrations would rise by 2 times when compared to 1 hour of cooking.

Though frequency of use for these purposes is not well established, particularly in recent literature and datasets, there are reasons for equity-related concern. A report based on the National Health and Nutrition Examination Survey (NHANES) III (1988-1994) found that use of a gas stove or oven for heating was highest among adults in the Southern U.S., with lower-income households engaging in such use of combustion appliances approximately twice as often as higher-income households.[125] In recent years, the U.S. Census Bureau's American Community Survey (ACS) started collecting data on supplemental heating devices, but found no clear income- or race-based trends. In each income group, 2.6-5.4% of households reported using cooking stoves for supplemental heating. Significantly fewer households use gas ovens with the oven door open for heating, as opposed to cooking stoves.

There have been a few articles written on the use of gas stoves for supplemental heating, though several date to the early 1980s. This behavior has been identified as a contributor to elevated indoor $NO_2$ levels in low-income housing — particularly in combination with poor ventilation and small apartment size[30] — and the use of gas stoves for heating is linked with childhood asthma.[118] In the previously mentioned NHANES III of more than 8,000 children, those who lived in homes where a gas stove or oven was used for heat were more likely to have

**Table 2-7:** Mean peak CO and $NO_2$ exposures in the entire home associated with pollutant backdrafting/spillage and various capture efficiencies.

| Appliance | Pollutant | Concentration with 0% Capture Efficiency (µg/m³) | Concentration with 25% Capture Efficiency (µg/m³) | Concentration with 50% Capture Efficiency (µg/m³) | Concentration with 75% Capture Efficiency (µg/m³) |
|---|---|---|---|---|---|
| Heating devices | CO | 11,000 | 8,200 | 5,600 | 3,000 |
| | $NO_2$ | 730* | 550* | 370* | 190 |
| | $NO_x$ | 3,900 | 2,900 | 2,000 | 990 |
| Water heaters | CO | 31,000* | 24,000* | 16,000 | 8,100 |
| | $NO_2$ | 1,400* | 1,100* | 710* | 360* |
| | $NO_x$ | 6,400 | 4,800 | 3,200 | 1,600 |

*Note: Values marked with * exceed acute CAAQS for CO and $NO_2$.*

clinically diagnosed asthma.[118,121] In a 1980s study of 700,000 residents in New York City, approximately 50% of those with gas stoves, but no gas heating appliances, were found to use gas stoves for supplemental heating.[126] A study of patients with symptoms of CO exposure found that use of gas stoves for heating was a significant predictor of high carboxyhemoglobin levels, indicating CO poisoning.[127] In a sample of residences in the United Kingdom, use of gas ovens for heating was significantly related to $NO_2$ concentrations in both kitchens and bedrooms.[128] However, in a study of low-income public housing in Boston, supplemental heating with stoves was not a significant predictor of indoor $NO_2$ concentrations, though gas stove heating behavior was only assessed via an initial survey and not during the environmental sampling period.[129] Researchers have called for further research and education on the impact of supplemental heating with gas stoves, and have identified the improvement of heating technology as a means of limiting the use of gas stoves for heating.

### 2.2.3. ADDITIONAL EXPOSURE AND VULNERABILITY CONSIDERATIONS

This section includes a brief, qualitative discussion of other factors associated with exposure and associated risk.

***Synergistic effects of multiple air pollutants***

There is potential for synergistic effects of exposure to multiple pollutants, including our pollutants of interest — CO, $NO_2$, and $NO_X$ — meaning that exposure to multiple pollutants at one time may not be a direct sum of the individual health impact of exposure to each pollutant.[130] The health effects of pollutants such as $PM_{2.5}$, $NO_X$/$NO_2$, and CO are evaluated separately in our report and others, but these exposures occur simultaneously, and concurrently with other pollutants not included in our analysis, such as heavy metals, PAHs, and VOCs (both during cooking, and due to other sources of indoor exposures).[27,119] We are not able to account for this in our analysis, but it is an important consideration.

***Body weight, inhalation rates, and gender***

Body weight and inhalation rates play a major role in determining the effects of personal exposure to airborne contaminants. Higher inhalation rates result in greater exposures, and lower body weights increase the effects of exposure, due to a higher dose per unit of body weight. Body weight and inhalation rate are also correlated, and thus, the two factors should be considered in conjunction.[131] Inhalation rates increase with body weight, with substantial increases found when comparing people of normal weight and people who are overweight; additionally, males have slightly higher inhalation rates than females.

Body weight and inhalation rate considerations are most important in regard to children, who are particularly susceptible to the adverse health effects of air pollution. Children perform a substantially higher level of daily physical activity than adults, which culminates in a far greater intake of air into the lungs.[132] Furthermore, children breathe 50% more air per unit of body weight than adults, due to having a greater lung surface area to body weight ratio.[133]

We did not incorporate body weight and inhalation rates into our quantitative assessment, but the California Department of Toxic Substances Control (DTSC) releases and regularly updates a set of Human Health Risk Assessment (HHRA) notes[134] focused on different aspects of risk assessment, and HHRA Note 1 is entirely focused on default exposure factors, including body weight and inhalation rate. Additionally, the 2011 US EPA Exposure Factors Handbook offers a comprehensive overview of inhalation rates for children and adults according to age, gender, body weight, and activity level.[131] According to this handbook, domestic tasks such as cooking are considered light intensity activities.

In terms of gender, the variations between men and women related to body weight and inhalation rate apply, but in regard to gas appliances, there is an additional consideration regarding which gender spends more time cooking and thus, is in closer proximity to gas appliances being used in the kitchen. Surveys have indicated that women do spend more time preparing meals than men[v], resulting in additional exposure to combustion pollutants in households with gas appliances.[137,138] A 1991 CARB study of children's activity patterns also found the prevalence of potential exposure to fumes from a gas-powered oven to be consistently elevated for children of all ages and genders.[139]

***Housing characteristics: tenure, quality, residence size, and appliance maintenance***

Disadvantaged populations disproportionately experience poor housing conditions that can be detrimental to health.[140] A recent State of the Nation's Housing report from Harvard University found that more than half of the nation's low-income population

v. We do not expand upon it in this report, but it is important to note that the gender disparities in cooking frequency as well as associated exposures for children are international issues, particularly in countries where traditional cookstoves are used to burn solid fuels, leading to significant environmental health concerns.[135,136]

live in high-poverty neighborhoods.[141] Housing quality related to safety is substandard in many low-income residences,[142,143] and is inextricably linked with public health concerns; also, appliance age and maintenance fall under this umbrella of housing safety considerations.[40,144] Lower-income families are often renters: The median income for renters in California is $50,000 annually, while the median income for home owners is $90,070 annually.[2] Lower-income families often do not have control over, or incentive to engage in, appliance replacement and maintenance; additionally, these families have less disposable income to spend on resolving maintenance issues. We can refer to the sensitivity analyses in 2.2.2, which highlight a potential health concern associated with old and unmaintained appliances: improper ventilation resulting in spillage of combustion pollutants into the living space. Lack of maintenance may result in improper ventilation occurring for extended periods of time. Thus, there is a need for future research on the sociodemographic trends in households with appliance maintenance and ventilation concerns.

Housing size variations are accounted for within the indoor air quality model; a smaller home with the same ventilation rate as a larger home will have higher concentrations of pollutants due to having lower volume. Our findings from Section 2.2.2 demonstrate that exceedances for CO and $NO_2$ are all higher for apartments, primarily due to smaller residence sizes.

Additionally, individuals in the kitchen and other rooms will be in closer proximity to gas-burning devices. This may have air quality and health implications for low-income populations living in smaller homes. Furthermore, low-income residences are more likely to be overcrowded,[145] which may affect cooking frequency. “Overcrowding” is defined as occurring when there is more than one person living in a residence as there are rooms in the residence (including rooms such as the living room and kitchen). According to the California Department of Housing and Community Development, California’s overcrowding rate is 8.4%, which is more than double the U.S. average of 3.4%.[145]

***Time-activity patterns***

Research has shown that children in low-income families spend more time in the home,[146] and are thus exposed to indoor air quality issues in the home more often than children from families with a higher socioeconomic status. Limited literature indicates that this may be particularly due to lower participation in after-school programs, resulting in greater exposure to indoor air pollutants in the home.[147,148] This is an area for future study.

***Cumulative impacts: health and environmental justice in California***

As we briefly touched upon in the background section, low-income and environmental justice communities are often disproportionately affected by adverse environmental conditions, and historically, have less access to clean water and air, as well as to clean energy resources. Many environmental issues in disadvantaged communities are externalities that these communities do not have control over, and these issues contribute to health disparities.

Existing research has explored the cumulative effects faced by vulnerable communities, finding that there are many complex, nuanced relationships between environmental and social factors that can result in significant (and potentially nonlinear) health effects on the population.[115,149] This includes the exacerbation of the effects of harmful environmental exposures — such as air or water pollution — and the enhancement of psychosocial stress experienced in impoverished neighborhoods. There are also potential synergistic effects from exposure to multiple pollutants, and multiple stressors, that need to be explored further to be fully understood.[115,149]

Research suggests that regulatory interventions must consider different elements of cumulative effects to reduce environmental inequities and associated health disparities.[115] It is critical to note that any air quality impact from the use of gas appliances compounds upon preexisting, complex, and adverse environmental and health burdens in these communities.

### 2.2.4. HEALTH EFFECTS OF AIR POLLUTION

In this section, we present the existing evidence surrounding each pollutant's relationship with gas appliance use, and describe the specific acute and chronic health impacts associated with exposure. We address indoor exposures specifically, but this section is also generally applicable to outdoor, ambient exposures to these pollutants. Table 2-8 summarizes the health effects described in this section.

#### *2.2.4.1. Nitrogen oxides*

$NO_X$, predominantly consisting of NO and $NO_2$, are widespread gaseous pollutants. $NO_2$ is primarily formed from the oxidation of NO.[155] Existing research evaluates the human health impacts of both $NO_X$ and $NO_2$, but much of recent literature focuses on $NO_2$, particularly since a growing body of evidence indicates that it leads to premature mortality. Therefore, we focus specifically on $NO_2$ in this section. The 2016 US EPA Integrated Science Assessment (ISA) on the health effects of $NO_X$ found the literature to be suggestive of a causal relationship between chronic $NO_2$ exposure and respiratory effects, cardiovascular effects, cancer, and mortality, though it did not make an absolute determination.[116]

As mentioned in Section 2.1.1, combustion appliances — specifically, gas cooking appliances — have been found to increase indoor $NO_2$ levels above acute CAAQS and NAAQS thresholds.[119] Studies have observed higher $NO_2$ exposures in homes with gas stoves compared to those with electric stoves.[27,221] When cooking with gas, peak concentrations of $NO_2$ in the kitchen can reach levels far in excess of the CAAQS 1-hour $NO_2$ threshold.[34] Individuals who cook with gas can be exposed to high levels of $NO_2$ due to close proximity to the stove. These peak concentrations of $NO_2$ are comparable to those reported in Section 2.2.2.

Though exposure to $NO_2$ has been linked to adverse health outcomes, there is some mixed evidence regarding the association between indoor $NO_2$ exposure from combustion appliances and specific respiratory health impacts.[31,33] Studies from the last several decades have found a robust association between $NO_2$ from gas cooking and increased risk of respiratory illness

**Table 2-8:** Overview of health effects of main studied pollutants.

| Pollutant | Health Effects | |
|---|---|---|
| | Acute | Chronic |
| Nitrogen oxides ($NO_X$) | Decreased lung function, asthma exacerbation, respiratory infection,[118,120,150–153] stroke[154] | Premature mortality,[155–158] lung and breast cancer,[156,159] cough, shortness of breath, asthma, wheezing, respiratory illness in children[33,33,91,117,120,160–163] |
| Carbon monoxide (CO) | Death, brain damage, seizures, memory loss, dementia, headaches, dizziness, nausea[164–168] | Brain and heart toxicity,[164,169–173] heart failure and cardiovascular disease,[167,174–176] low birth weight[177] |
| Fine particulate matter ($PM_{2.5}$) | Stroke, increased blood pressure[154,178–180] | Premature mortality,[22,181] bronchitis, asthma onset and exacerbation,[185–189] low birth weight and preterm birth[190–194] |
| Ultrafine particles (UFP) | Increased blood pressure[179,195] | Cardiovascular disease,[196,197] neurological disorders[198,199] |
| Formaldehyde | Respiratory/eye/skin irritation, sneezing, coughing, nasal congestion,[103,200,201] drowsiness, chest tightness, shortness of breath,[103,200,202] asthma exacerbation,[203,204] death (higher doses)[205] | Cancer,[103,172,206–210] asthma and bronchitis in children,[200,211,212] damage to respiratory system,[205,211,213–219] headaches, sleep disorders, memory loss,[202,205] birth defects, low birth weight, spontaneous abortion[205,213,218,220] |

in children, such as asthma, wheezing, and other respiratory symptoms.[33,91,120,162,163] $NO_2$ exposure from gas appliances is implicated in many other respiratory symptoms, including cough, lung obstruction, and shortness of breath.[33,117,160,161] Women may be at higher health risk from $NO_2$ exposure, due to greater susceptibility and higher frequency of cooking compared to men.[117,138,160] Research suggests that due to the widespread use of gas for cooking, $NO_2$ exposure from gas appliances has a substantial public health impact, particularly in children, as described in Section 2.2.3.[33]

The respiratory effects of acute $NO_2$ exposure more broadly include decreased lung function, asthma exacerbation, and increased risk of respiratory infection.[118,120] Children are at the highest risk of health effects from $NO_2$ exposure.[118,120,222] Short-term $NO_2$ exposures above the CAAQS 1-hour standard are associated with lung inflammation, particularly in individuals with asthma or chronic obstructive pulmonary disease (COPD).[150–153] Acute $NO_2$ exposure is also associated with increased risk of hospital admission and mortality from stroke.[154]

Chronic $NO_2$ exposure is suspected to be a driver of air-pollution-related mortality and is associated with premature death.[156,222] Studies have observed a relationship between chronic $NO_2$ exposure and all-cause, cardiovascular, respiratory, and lung cancer mortalities, with greater risks among populations with preexisting diseases.[155–158] Chronic $NO_2$ exposure also increases the risk of lung and breast cancers,[156,159] and evidence also suggests impact to pregnancy outcomes, such as low birth weight.[223–225] A substantial body of evidence supports an independent effect of $NO_2$ on mortality, and epidemiological research on this burgeoning topic is accumulating quickly.[155–158] Because $NO_2$ is ubiquitous and large populations are exposed, even small increases in $NO_2$ may have extensive public health consequences.[155–158,222]

#### *2.2.4.2. Carbon monoxide*

Although exposure to dangerous levels of CO is preventable, many instances of CO poisoning still occur in homes,[226] resulting in estimated expenses of $1.3 billion annually in the United States.[227,228] As mentioned previously, the Tracking California program estimated 643 emergency department visits due to non-fire-related CO poisoning in 2016,[82] and CARB estimates there have been 13-36 non-fire-related CO poisoning deaths in California annually since 2000.[67] Although CO emissions from gas appliances can be negligible,[27,49,51,52] and most of the CO concentrations presented in our results in Section 2.2.2 are below the state and national 8-hour standards of 10,000 µg/m³, dangerously high CO exposures from gas appliances may occur due to mechanical and ventilation failures.[164,229] Excessive CO exposures, often associated with gas appliances, have been found to cause severe damage to brain tissue,[164–168] and can result in long-term or permanent neurological symptoms such as seizures, memory loss, and dementia.[228–232]

CO exposure has diverse, acute human health effects, with symptoms ranging from headaches, dizziness, and nausea at low concentrations, to neurological damage and death at high concentrations.[165,168] CO is an insidious pollutant; because it is tasteless, odorless, and induces nonspecific symptoms, CO exposures often remain undetected by both victims and medical professionals.[233]

While the health effects of acute exposure are well-established, the long-term impact is not as well-studied or understood. Chronic exposure to low concentrations of CO was found to be associated with adverse health effects on multiple organ systems, with substantial evidence demonstrating toxic effects on the brain and heart.[164,169–173] The World Health Organization (WHO) suggests potential toxic effects of chronic CO concentrations above 6.9 mg/m³.[172] Increases of 11.5 mg/m³ in ambient CO levels are associated with increased risk of hospital admission for congestive heart failure, particularly in the elderly,[167,174,175] whereas increases of slightly more than 1 mg/m³ in 1-hour maximum CO concentrations are associated with increased risk of cardiovascular-disease-related hospitalizations.[176] In addition, children are especially vulnerable to CO exposure due to their developing nervous systems and high metabolic rates,[229] and exposure to ambient CO levels close to those of the WHO threshold listed previously in this report is associated with an increased risk of low birth weight.[177] There are other risks associated with CO exposure during pregnancy as well.[234]

#### *2.2.4.3. Particulate matter*

PM is a leading cause of worldwide mortality and morbidity, and there is evidence that $PM_{2.5}$ pollution adversely affects cardiovascular and respiratory health through a myriad of pathways.[22,181] Recent research, based on the Global Burden of Disease project, found that $PM_{2.5}$ led to approximately 8.9 million deaths in 2015, which is higher than previous estimates.[24] PM concentrations are often higher indoors than outdoors and come from a variety of sources, including cooking, household aerosol products, office equipment, and transportation of outdoor pollution into the indoor environment.[182]

Cooking with combustion appliances can be a significant source of $PM_{2.5}$ and UFP exposure,[26,27,64,235] though studies of $PM_{2.5}$ exposure from these types of appliances have shown a decrease over the last several decades, likely due to technological advances resulting in reduced emissions. Although both gas and electric stoves generate particle emissions, gas stoves have been found to produce greater particle exposures.[26,28] Cooking methods, and the type of food being cooked, can also have a substantial impact on $PM_{2.5}$ emissions, and the use of cooking oils with higher smoke temperatures has been identified as a means of reducing $PM_{2.5}$ emissions.[236] As mentioned previously, many of these experimental tests involved food, and the $PM_{2.5}$ concentrations observed cannot solely be attributed to the appliances. Additionally, as mentioned in Section 2.1, PM emissions from gas water heaters and home heating devices are significantly higher than PM emissions from kitchen appliances.

Short-term exposure to $PM_{2.5}$ is associated with cardiovascular and cerebrovascular disease, including strokes and increases in blood pressure.[154,178–180] An increased risk of hospital admissions and mortality for stroke has been observed per 10 μg/m$^3$ short-term increase in ambient $PM_{2.5}$.[154]

$PM_{2.5}$ also has well-established, chronic health effects.[182] An extensive body of evidence supports a significant association between $PM_{2.5}$ and all-cause mortality.[22] $PM_{2.5}$ exposure is associated with an increased risk of cardiovascular and respiratory mortality, with a greater increase in risk than seen for $NO_2$.[156–158] The impact of $PM_{2.5}$ pollution also includes increased emergency room visits and general hospital admissions,[182–184] and chronic $PM_{2.5}$ exposure is linked to certain cardiovascular diseases and chronic respiratory conditions as well, such as bronchitis and asthma onset and exacerbation.[185–189] Chronic $PM_{2.5}$ exposure is particularly harmful to pregnant women and children. Long-term $PM_{2.5}$ exposure during pregnancy is associated with increased risk of low birth weight and preterm birth per 10 μg/m$^3$ increase in $PM_{2.5}$.[190–194] A study of more than 600,000 births in California over a 7-year period revealed a significant association between $PM_{2.5}$ exposure and low birth weight.[237] One recent study found that the dose-response relationship between preterm birth and $PM_{2.5}$ is linear at lower pollution levels, suggesting increased risk even at low concentrations.[190]

Evidence indicates that combustion processes produce large amounts of UFPs.[48,57,236] UFPs can also be formed by nucleation, where low volatile gas phase species are converted to aerosol phases. Nucleation events can be provoked by the operation of gas appliances, where combustion processes produce gaseous emissions such as CO and $NO_2$.[44,238,239] In an LBNL study of combustion appliance emissions, the vast majority of particles emitted by gas stoves were found to be in the ultrafine range,[44] and a study of residences in Northern California found cooking to be the greatest source of indoor UFPs.[48] A chamber study demonstrated substantial UFP number concentrations of more than 300,000 particles/cm$^3$ emitted by both gas and electric stoves,[240] while a residential study of gas stoves in Taiwan recorded $PM_{2.5}$ concentrations of up to 100 μg/m$^3$ and UFP emissions of up to 1,400,000 particles/cm$^3$.[241]

Burgeoning research indicates that UFPs significantly affect human health, though regulatory intervention to control emissions of these particles is particularly challenging, due to their small size.[198] Emerging evidence posits UFPs are potentially more toxic and harmful than $PM_{2.5}$ on a per unit mass basis.[179,242]

Chronic exposure to UFPs is associated with increases in markers related to cardiovascular disease risk.[196,197] Both ambient UFP concentrations and UFP emissions from indoor sources have been found to increase blood pressure in adults and children.[179,195] UFP exposure also has pronounced respiratory effects: A study of five European cities over a 10-year period found an association between UFP and respiratory hospitalizations during warm periods, with the strongest effects seen among children 0-14 years old.[243] Researchers suggest that UFPs may contribute to neurological disorders as well.[198,199]

Exposure to PM has well-established acute and chronic health effects, and though we were not able to quantify indoor residential exposures to $PM_{2.5}$ and UFPs in this report, due to limited available data, PM exposure due to indoor gas appliance operation should be considered in future air pollution and health effect studies.

#### *2.2.4.4. Formaldehyde*

Formaldehyde is a part of a larger family of VOCs, which are common indoor pollutants with sources including building materials, carpeting, paint, furniture, personal care products, and combustion.[30,244,245] Newer residential buildings have been found to produce greater formaldehyde and VOC emissions than older buildings.[200,244] Additionally, recent evidence suggests that infiltration of outdoor formaldehyde contributes substantially to indoor concentrations.[246,247] Formaldehyde is an extremely prevalent pollutant, and a study of California homes found that 95% had formaldehyde levels above the OEHHA chronic REL.[45]

Formaldehyde has been formally established as a human carcinogen by regulatory agencies such as the WHO and the International Agency for Research on Cancer (IARC), and noted potentially carcinogenic by other agencies, with evidence of it causing nasopharyngeal cancer and, to a lesser extent, leukemia.[103,172,206–210] Apart from its carcinogenic effects, formaldehyde is a sensory and respiratory irritant with both acute and chronic non-cancer health effects.[103,200,201] However, formaldehyde exposures often occur in conjunction with large numbers of other VOCs and indoor air pollutants; thus, identifying the direct health effects of formaldehyde has proven challenging.[200,213,248]

Formaldehyde can be formed as a byproduct of combustion processes, due to incomplete combustion.[103,247] Although most existing literature focuses on formaldehyde emissions as a result of cigarette smoking, wood combustion, and off-gassing from building materials, a number of studies have investigated the effects of formaldehyde formation due to gas appliances and residential cooking activities.[27,44–46,74] Although the operation of gas appliances has been found to result in formaldehyde emissions, the concentrations measured in such studies were often below the OEHHA chronic REL of 9 μg/m$^3$.[27,44] Additionally, several studies did not indicate any significant contribution of gas appliance use to indoor formaldehyde concentrations.[45,46,74] These results are consistent with research showing that building materials are the primary sources of indoor formaldehyde emissions.[30,244,245] Preliminary evidence in mice suggests intermittent exposures to higher concentrations of formaldehyde are more damaging than constant low-level exposures, as the dose-response relationship between formaldehyde and its impact on human health is not linear.[214] This may be of importance when considering formaldehyde emissions from gas appliances, as these exposures are acute and unpredictable, as opposed to chronic and stable (as in the case of formaldehyde emitted from building materials).

Respiratory irritation is the most common, acute effect of formaldehyde exposure, along with related symptoms such as dry skin, sneezing, coughing, eye irritation, and nasal congestion.[200,202,205,207,211,249] Formaldehyde exposure is also associated with a range of nonspecific symptoms, including drowsiness, chest tightness, and shortness of breath.[103,200,202] Relatively low formaldehyde concentrations are associated with increased risk of asthma and chronic bronchitis in children.[200,211,212] Formaldehyde also increases sensitivity to allergens in asthmatics, even at the WHO-recommended maximum, 30-minute average concentration.[203,204] Acute formaldehyde poisoning at higher doses is associated with severe symptoms, including fever, vomiting, abdominal pain, and in extreme circumstances, death.[205]

Chronic formaldehyde exposure is an issue of concern as well, as effects have been found to increase over time. In addition to increasing cancer risk, chronic exposure to formaldehyde results in a multitude of symptoms, including reduced lung function, tremors, and damage to the nasal passages.[205,211,213–219] The relationship between chronic formaldehyde exposure and poor respiratory health may be particularly important for children.[250] Additionally, chronic formaldehyde exposure has neurotoxic effects, causing symptoms such as headaches, sleep disorders, and memory loss.[202,205] Formaldehyde is also a reproductive and developmental toxicant associated with birth defects, low birth weights, and spontaneous abortion.[205,213,218,220]

While exposure to formaldehyde can result in life-threatening, adverse health conditions, it remains unclear whether formaldehyde is a significant concern related to gas appliance use. Since formaldehyde is a known carcinogen, this topic demands further research.

### 2.2.5. ASSUMPTIONS AND LIMITATIONS

Due to the limited scope of this project, we did not conduct any primary data collection; we only analyzed existing literature and datasets. While we used as many relevant data sources as we could access, data paucity was a major limitation for this report. Particularly for conducting future quantitative analyses with regard to equity, the development of additional, publicly available databases to include more detailed and higher spatial resolution data would be a significant asset.

There are other factors associated with exposure that we were unable to control for, including the location of appliances throughout the home (water heaters and home heating devices), and seasonality (which affects ventilation, as well as the ambient pollutant concentrations used in the indoor air model).[29,47] There were also challenges associated with determining a standard residence volume and ventilation rate for each residence type, so these values are based on estimates from primary literature, and public data and reports.

Details on our calculations of volumes and ventilation rates are included in Appendix A. We also did not assess any exposures or other dangers associated with electrification, as we focus on combustion pollutants in this report.

There are also limitations associated with the use of an indoor air quality model. This model assumes the pollutant of interest reaches a steady state, which is more appropriate for emissions occurring over a consistent period, not for analyzing short-term emissions. Our analyses also operated under various assumptions about the time spent using kitchen appliances. We approximated applicable time periods, but we also wanted to provide varying assessments considering different amounts of time using the appliances, so that readers of this report can gain a better understanding of the implications of their own appliance-use habits. Additionally, as mentioned previously, the 1-hour (acute) thresholds compared to our peak kitchen pollutant concentrations only apply to exposures under a scenario where air quality levels remain elevated for an entire hour.

Finally, there are indoor air quality issues associated with the use of gas cooking appliances that will remain despite the implementation of electrification, and we do not account for this. Some PM emissions are associated with cooking oils and foods, and there are no mitigation methods for this, other than the use of ventilation devices such as range hoods. We do not claim that the transition to electric appliances would make a substantial difference in terms of emissions from cooking oils and food.

This report does not compare the benefits and costs of electrification versus improving range hood use and efficiency in terms of reducing indoor air pollution. This is an important consideration that needs to be included in any full-scale assessment of indoor air pollution mitigation techniques. We touched briefly upon range hoods in Section 2.1.3.

# 3 OUTDOOR AIR QUALITY AND HEALTH EFFECTS

Section 2 discussed the indoor air quality issues and resulting health effects associated with the use of residential gas appliances. This portion of the report covers an equally relevant realm: how the use of these appliances affects outdoor (ambient) air quality, the extent to which residential building electrification would reduce ambient exposures to the pollutants of concern, and the resulting premature mortality and morbidity reductions throughout the state. Section 3.1 provides background information from a literature review, and Section 3.2 contains the results and discussion. The objectives of this section are to:

- quantify the total emissions of CO, $NO_2$, and $NO_X$ due to gas appliances across California;
- model changes in ambient $PM_{2.5}$ due to reduced emissions of $NO_X$ and $PM_{2.5}$ from a hypothetical, residential building electrification scenario; and
- estimate the potential reduction in mortality associated with the modeled scenario.

In the results and discussion portion of this outdoor air quality section of the report, as described in these objectives, we included various quantitative assessments of four pollutants: CO, $NO_2$, $NO_X$, and $PM_{2.5}$. In Section 3.2.1, we assessed the total emissions of CO, $NO_2$, and $NO_X$ based on our EFs calculated in Section 2. In Sections 3.2.2 and 3.2.3, we considered only two pollutants: $NO_X$ and $PM_{2.5}$. We first estimated reductions in secondary $PM_{2.5}$ levels due to a calculated reduction in $NO_X$ and resulting nitrate $PM_{2.5}$. We then incorporated CARB data on $PM_{2.5}$ emissions from residential gas appliances to estimate the total reduction in $PM_{2.5}$ from the replacement of gas appliances, representing changes in primary and secondary (nitrate) $PM_{2.5}$ from gas use. We then assessed health impacts from reductions in those estimated ambient $PM_{2.5}$ levels. This is described in detail in Appendix A.

## 3.1. BACKGROUND

In this literature review section, we discuss the following relevant topics:

- current electrification research as it relates to criteria air pollutants, and related policy and implications of electrification more generally (3.1.1);
- the relationship between gas appliances and outdoor air quality (3.1.2);
- resulting outdoor air quality, health, and environmental justice implications (3.1.3); and
- the identified knowledge gap we aim to fill (3.1.4).

### 3.1.1. CALIFORNIA'S ELECTRICITY LANDSCAPE

***Electricity and criteria air pollutant emissions***

Residential building electrification has multiple potential co-benefits, spanning the domains of air quality, health, and climate change mitigation.[251,252] Reducing air pollutant emissions through electrification would improve air quality and promote public health while also limiting production of GHGs. The coupling of electrification with decarbonizing electricity generation represents an ideal scenario, producing these associated co-benefits. Of course, even without complete, wide-scale decarbonization of electricity generation, decreasing the carbon-generating proportion of the power mix will be conducive to climate change mitigation.[253]

There are several existing research studies on air quality and health co-benefits from electrification.[254–256] A recent modeling study predicted that achieving the California Executive Order S-3-05 target of reducing GHG emissions to 80% below 1990 levels by 2050 through a focus on electrification would result in significant public health benefits.[256] A comprehensive decarbonization approach, prioritizing electrification and clean, renewable energy sources, with 85% electrification in commercial and residential sectors, would reduce 2050 emissions of $NO_X$ by 34% and $PM_{2.5}$ by 33%.[256] Furthermore, the pollutant reductions would result in the avoidance of an estimated 12,100 premature deaths annually by 2050, due to changes in ambient ozone and $PM_{2.5}$, with a monetized estimated value of $109 billion.[256] This particular scenario, focused on the implementation of clean, renewable energy and high levels of electrification, had significantly more health co-benefits than a scenario focused on combustible, "renewable" fuels. Similarly, a study modeling the air quality impact of Assembly Bill (AB) 32[257] (The Global Warming Solutions Act of 2006, which facilitated the enactment of the Cap-and-Trade Program), predicted cumulative emissions reductions of approximately 15% for $NO_X$ and 1% for $PM_{2.5}$ throughout California.[254] Air quality improvements due to AB 32 were predicted to avoid approximately 880 premature deaths per year by 2030, with an estimated monetized value of $5.4 billion.[254] Another California-focused study on reaching the 2050 GHG emissions targets predicted air-pollution-associated premature mortality reductions of up to 2,760 deaths per year by 2050, with an estimated monetized value of up to $20 billion annually.[255]

A 2018 CEC report investigating future decarbonization scenarios reported high levels of building electrification to be an effective, relatively low-risk, and low-cost GHG mitigation strategy as compared to other mitigation measures, and a key factor in reducing gas consumption.[7] On a related project in 2019 for the CEC, the E3 group and the Advanced Power & Energy Program at University of California, Irvine (UCI) evaluated the air quality and health effects of electrification; they evaluated multiple scenarios to reduce GHG emissions from 1990 levels by 80% by the year 2050, including a high building-electrification scenario, and a no building-electrification scenario.[258] Similar to the CEC report, they also found that building electrification has the potential to be low risk and low cost, in this case compared to the

widespread use of renewable gas.[258] One portion of this project involved using the U.S. EPA's Benefits Mapping and Analysis tool (BenMAP) to conduct a health-impact analysis for a high building-electrification scenario. Building electrification was projected to result in lower $PM_{2.5}$ concentrations, particularly in winter, and the BenMAP analysis reported health savings of approximately $200 million over 10-day episodes in summer and winter, due to mitigation of ozone and $PM_{2.5}$ from the high building-electrification scenario.[258] Reductions in secondary $PM_{2.5}$ formation from lower $NO_x$ emissions from gas appliances were found to have a major impact on health savings.[258] The building-electrification scenarios modeled by UCI indicate substantial co-benefits to air quality and human health as a result of reductions in $NO_x$, $PM_{2.5}$, and ozone. However, these scenarios were designed to reduce GHG emissions rather than target criteria pollutant reductions.[258]

EPRI prepared a 2019 report, commissioned by the CEC, investigating the air quality implications of electrification in California across multiple sectors.[15] It estimated that electrification would result in substantial reductions of $PM_{2.5}$ and ozone across California, with monetized health benefits estimated to be $108 billion per year in 2050.[15] It also found that electrification of residential and commercial stationary sources resulted in the majority of $PM_{2.5}$ reductions (61%), with significant impacts from the reduction of residential wood combustion.[15]

One aspect to keep in mind throughout this analysis, which will be mentioned again in the Results and Discussion section, is that electricity generation at gas power plants emits both GHGs and criteria air pollutants. Even if all residential gas appliances were transitioned to electric appliances, the electricity required to power these appliances must still be generated by some form of fuel, and gas power plants currently produce almost half of the electricity generation in the state. Therefore, in order to avoid increased emissions from gas power plants, building electrification must be based on the preface that the electric power system will continue to decarbonize and shift to clean energy. As California increasingly builds and relies upon zero-carbon electricity sources such as wind and solar energy, which is state-mandated by the 100% Clean Energy Act of 2018, or SB 100, the overall GHG and air quality benefits of electrification will increase (This law enacted the Renewables Portfolio Standard (RPS), which mandates that 100% of electricity sold must be generated from zero-carbon energy sources by 2045[259]. This is discussed further in Section 3.2.2.

Overall, GHG emissions reductions and in particular, electrification, offer immense co-benefits with regard to air quality, health, and economic value, with the largest benefits predicted in densely populated, metropolitan areas.[7,15,254–256]

***Building electrification in California: policy and economic implications***

California is a national leader in clean energy and climate policy. Though local emissions of criteria pollutants are a byproduct of combustion processes (the focus of this report), there is a substantial body of research on the relationships between energy sources and climate change mitigation,[3,42,251,260–262] and climate change mitigation continues to be a main driver of policy that affects electrification status throughout the state. Together, SB 32 (which extends AB 32), SB 100, California's B-55-18 Executive Order to achieve carbon neutrality by 2045, and other forthcoming bills provide a strong legislative framework for mitigating climate change, with aggressive targets for reducing GHG emissions.[257,263] Research has identified wide-scale electrification of multiple energy-consumption sectors, including residential buildings, as an important requisite for achieving California's GHG emissions goals; [4,7,8] this transition will require policy support.[264]

Unlike the national landscape, where approximately 25% of all homes in the United States are all-electric,[265] roughly 90% of California's homes consume gas for various fueling purposes. The majority of California homes use gas for heating and cooking: Recent AHS surveys estimated that 64% of California homes used gas as their primary heating fuel, and that 67% of homes used gas as their primary cooking fuel.[2] One example of a scenario with high rates of building electrification, as described in the recent report by E3, finds that more than 7 million existing California residences will need to be retrofitted with electric technologies.[4] A different report posed another scenario showing that if gas were entirely phased out at an accelerated pace, there would be more than 13 million residential buildings in California retrofitted by 2045.[266]

In July 2019, Berkeley became the first city in California to introduce legislation to phase out the use of gas piping in new buildings, with limited exceptions.[267] Since then, roughly 30 cities and counties have adopted ordinances supporting or requiring the construction of all-electric buildings.[268]

A study modeling the impact of future building electrification found that all-electric homes performed better than mixed-fuel buildings, in terms of both GHG emissions reductions and abatement costs associated with the construction of buildings compliant with the Title 24 California Building Standards.[269] In particular, the electrification of space and water heating appliances presents an opportunity for substantial GHG emissions reductions, which aligns with our results from Section 2 regarding criteria pollutant emissions.[260,270,271]

Additionally, a recent study investigating the consequences of residential building electrification in California predicted several load-distribution effects to the electrical grid.[4] Building electrification would result in more efficient utilization of the power grid as characterized by an improved load factor, which is the ratio of average- to peak- electricity demand. The report forecasted changes in seasonal electricity demand, with higher overall winter electricity loads and slightly lower peak summer electricity loads.

Transitioning to electric heat pumps would provide effective heating and cooling in buildings, and the recent maturation of heat pump technologies has been identified as an efficient and beneficial component of future electrification.[4,272,273] This is all particularly relevant as California temperatures are rising. Projected increases in the intensity, frequency, and duration of heat waves[274] could result in higher air conditioning adoption, increased cooling demands, and decreased heating demands.

There are several economic considerations related to a transition from gas to electricity. Fugitive $CH_4$ emissions produced when gas appliances are not operating (e.g., emissions from pipe leaks) are estimated to incur an economic cost to consumers of approximately $30 million annually.[3] Research has also found that building all-electric reduces construction costs considerably, and lowers energy bills overall.[275] One research group projected consumer energy bill savings for a range of appliances, and found that despite higher capital costs in certain scenarios, the majority of households will have both bill and life-cycle savings as a result of building electrification.[4] However, one consideration identified by a recent report is that as demand for gas falls, cost for gas customers increases significantly.[258] In a wide-scale electrification scenario, this may result in low-income gas consumers requiring rate protection or financial assistance for transition. This report also predicted higher utility bills for mixed-fuel homes than for all-electric homes after year 2030.[258] This may have multifaceted effects that constitute equity concerns; higher utility bills for gas-using homes will further encourage electrification, but low-income consumers who rent their homes and do not own their gas appliances, or are unable to afford purchasing electric appliances, may bear a disproportionate burden of transition costs. One report addressed other barriers to residential electrification throughout the state, highlighting that, despite the fact that all-electric homes have lower maintenance costs (and other considerable benefits, including no direct emissions), the up-front costs of purchasing high-efficiency, electric appliances are higher.[15] Savings were highest for homes with the greatest heating and cooling demands, such as larger SFHs.

Policy intervention providing incentives for replacing gas appliances with electric appliances may make the transition to electrification in California more equitable. For example, tariffs for all-electric homes offer lower rates for electricity to offset their higher electricity consumption.[276] Financing programs providing low- or no-interest loans for electric appliances could provide a means for making electrification economically feasible, especially in disadvantaged communities.[276]

### 3.1.2. RESIDENTIAL GAS APPLIANCES AND OUTDOOR AIR QUALITY

***Relationship between indoor and outdoor air quality***

While much of scientific literature focuses on the transport of outdoor pollutants into indoor environments, emissions from residential gas appliances also transport outdoors, through the ventilation system or through open windows and other pathways.[277] The relationship between indoor and outdoor air can be characterized by the indoor/outdoor (I/O) ratio. The I/O ratio is influenced by factors such as natural and mechanical ventilation and the tightness of the building envelope: Generally, closed windows lead to low I/O ratios, while well-ventilated environments have higher I/O ratios.[277]

***Pollutant chemistry***

Atmospheric chemistry is important when considering certain criteria pollutants. $NO_X$ is heavily involved in the formation of ground-level ozone.[278] Ozone is a secondary pollutant produced by a complex chemical reaction between $NO_X$, VOCs, and sunlight.[278] Ozone is a risk factor for all-cause, cardiovascular, and respiratory mortality,[279,280] and the global burden of ozone exposure is estimated at almost half a million deaths per year,[281] although a recent study estimates that this number could actually be as high as 1.2 million.[282] Contributions to ozone associated with $NO_X$ emissions from gas appliance use are outside the scope of this study.

### 3.1.3. WHY THIS ISSUE MATTERS: OUTDOOR AIR QUALITY, HEALTH, BUILDING ELECTRIFICATION, AND ENVIRONMENTAL JUSTICE

In Section 2, this report discussed indoor air quality effects from the use of gas appliances in residences;

however, air pollution from residential gas combustion affects outdoor air quality as well. The relationship between outdoor air quality and public health is an important equity issue that merits greater public awareness and policy development. As introduced in Section 2.1.4, low-income communities and communities of color often have poor air quality and are burdened with associated negative health effects. More than 40% of all fossil-fuel power plants in California are disproportionately concentrated near disadvantaged communities, including in the San Francisco East Bay, the Sacramento area, Bakersfield, and the Los Angeles South Bay.[283] Furthermore, data from the exposure mapping tool CalEnviroScreen demonstrate that the most socially vulnerable communities, such as those with high poverty, unemployment, and poor health status, also suffer the highest cumulative burden of pollution exposure.[283,284] These communities often have less access to healthcare and may only seek out medical professionals when in dire need, further exacerbating the mortality and morbidity effects experienced due to increased pollutant exposure.[15]

Apart from these important considerations regarding outdoor air quality, health, and environmental justice as they relate to gas appliances and electrification, there are numerous other equity considerations related to the potential transition to electrification. These equity considerations are not a focus of this report but must not be overlooked.

The University of Southern California (USC) Program for Environmental and Regional Equity (PERE) prepared a 2019 report for the Climate Equity Network offering guiding principles for a just and equitable transition to a low-carbon future.[283] The report drew attention to the need to protect and prioritize disadvantaged communities throughout the process of alleviating the effects of climate change. EPRI's electrification report (discussed previously) also highlights equity considerations.[15] EPRI conducted interviews and hosted stakeholder meetings with environmental justice groups to discuss the results of their electrification analysis, receiving input from the communities affected.[15] Beyond simply acknowledging injustices of the past, both the USC PERE and EPRI emphasize the need for disadvantaged communities to play a role in the transition to electrification and renewable energy, stating that these marginalized populations should be included in the decision-making process as a means of advancing equity.[15,283]

A 2019 Greenlining Institute report focused on equitable building electrification, developing a five-step framework for California to ensure that environmental and social justice communities are at the forefront of this transition.[285] The steps outlined in this report include working closely with communities to identify needs and make community-driven decisions, identifying methods and metrics for data tracking, ensuring allocation of necessary funding, and successfully influencing outcomes.[285] As mentioned previously, policy intervention, such as providing incentives for replacing gas appliances with electric appliances, can help make the transition to electrification in California more equitable.[276] While there are existing low-income

energy programs throughout the state, Greenlining's report identifies that there are significant distributive justice shortcomings in terms of benefit allocation; for several reasons outlined in more detail in their report, program benefits are not maximized for households in need.[285] These reports and many others have highlighted the importance of equity in decision-making regarding California's energy future.

### 3.1.4. KNOWLEDGE GAP AND CONTRIBUTIONS TO THE LITERATURE

Our study contributes to a growing body of recent research on the potential impact of expanding electrification throughout the state. Large-scale research projects modeling the future of electrification in California have considered the impact across economic sectors.[15,254–256] These studies reported significant reductions in criteria air pollutant emissions, and associated health and monetary benefits, in addition to reduced GHG emissions. Our approach is novel in that it isolates the emissions and health effects of gas appliances in the residential sector, providing estimates of criteria pollutant emissions and their resulting effects on outdoor air quality and health.

Our focus on the emissions associated with residential gas appliances may serve as a benchmark to be used by future models of a potential switch to electrification. This report offers a quantitative approximation of the contribution of residential gas appliances to overall air pollutant levels in California. We anticipate that this analysis will contribute to a more developed understanding of how residential activity impacts air quality on a larger, statewide scale.

## 3.2. RESULTS AND DISCUSSION

### 3.2.1. CONTRIBUTION OF GAS APPLIANCES TO OUTDOOR AIR POLLUTION IN CALIFORNIA

***Total emissions of pollutants from gas appliances throughout the state***

Using our calculated EFs and CEC data on gas consumption, we estimated total annual emissions of $NO_X$ (which includes $NO_2$) and CO for 2018 (See Appendix A, Section A.2.1 for additional details). We found that residential gas appliances emitted approximately 15,900 tons of $NO_X$ (with a confidence interval of 15,500 to 16,300) and 12,000 tons of CO (with a confidence interval of 10,800 to 13,100) in 2018. In comparison, CARB's annual estimates for residential gas appliance use were approximately 16,000 tons for $NO_X$ and 9,000 tons for CO for 2018. There is no specific estimate for $NO_2$ provided by CARB for comparison here, but we do present $NO_2$ results separately in Figures 3-1 and 3-2. Since the numbers are similar in magnitude, indicating consistency between our estimates and CARB's estimates, we chose to extract $PM_{2.5}$ gas combustion emission estimates from CARB, and use them for the remainder of the outdoor air quality and health analysis in this section to develop more comprehensive estimates for total pollutant reductions and mortality impacts. This is discussed in Appendix A in more detail. Figure B-4 in Appendix B depicts total emissions of the three studied pollutants by county from gas appliance use, calculated using the EFs reported in Section 2.

This report's findings indicate that emissions from residential gas appliances account for approximately 3% of total $NO_X$ emissions in California (Figure B-5 in

**Figure 3-1:** Estimated state-wide emissions of pollutants (a) CO, (b) $NO_2$, and (c) $NO_X$ by gas appliance type.

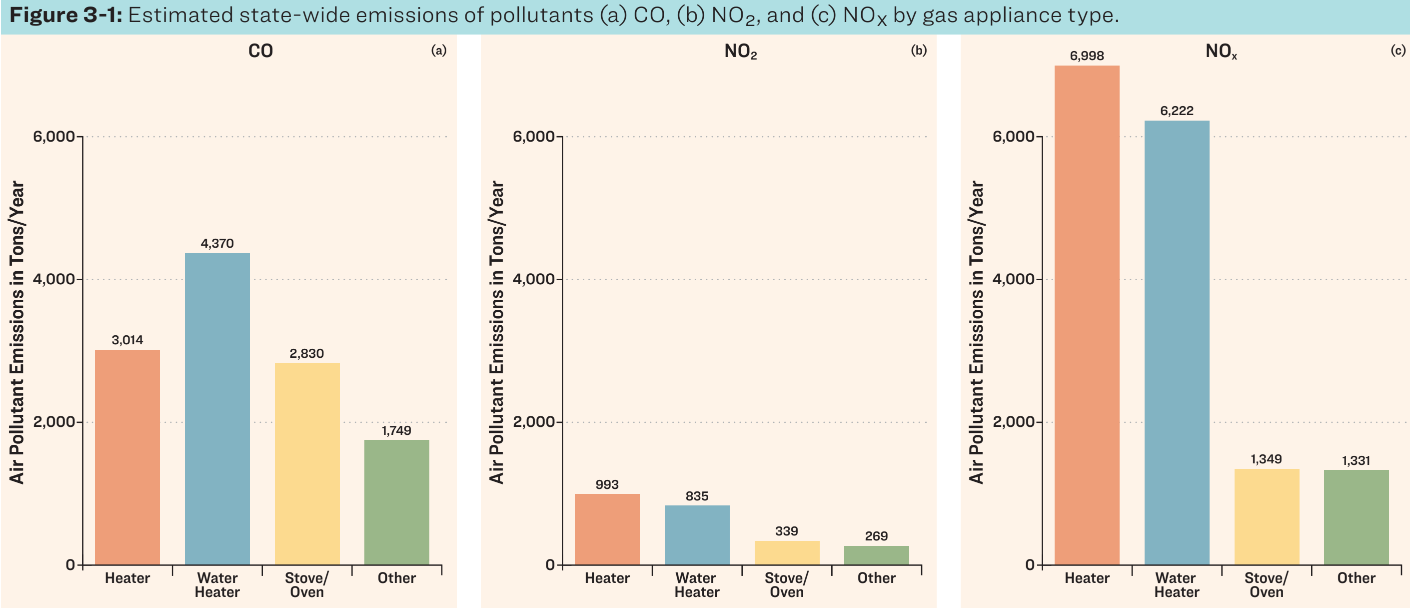
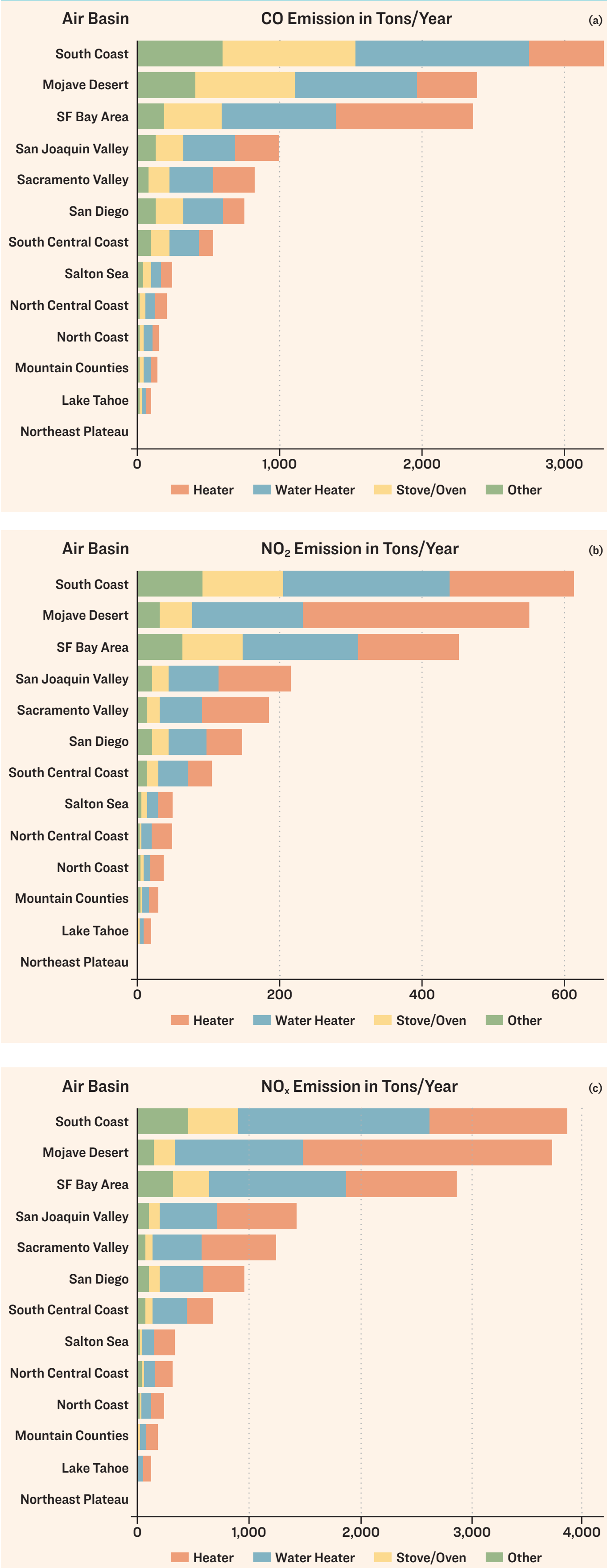


**Figure 3-2:** Estimated emissions of (a) CO, (b) $NO_2$, and (c) $NO_X$ in air basins from gas appliances by type.




Appendix B shows $NO_X$ emissions from gas appliances by county, as compared to $NO_X$ emissions from all sources).[73] Among all counties in California, Los Angeles County has the highest total $NO_X$ emissions, as well as the highest $NO_X$ emissions from gas appliances (3,900 tons/year). As of 2019, 34 million Californians live in counties that are not in compliance with state or federal ambient air quality standards for ozone and/or $PM_{2.5}$.[286] Considering $NO_X$ contributes to ambient $PM_{2.5}$, gas appliances have the potential to add to this pollution burden.

***Comparison of emissions from various types of appliances***

Our analysis indicates that gas water heaters and home heating devices, such as furnaces, are responsible for the bulk of outdoor air pollution from gas appliances. Gas water heaters contribute the most to CO emissions (36.5% of all CO emissions come from residential gas appliances) when compared with other types of gas appliances, while gas heating appliances emitted the most $NO_X$ (44% of all $NO_X$ emissions from residential gas appliances) in California for 2018 (Figure 3-1). This is associated with the relative EFs in ng/J of each pollutant for each appliance type, as well as the percent distribution of use of these appliances, extracted from the Residential Appliance Saturation Study (RASS).[287]

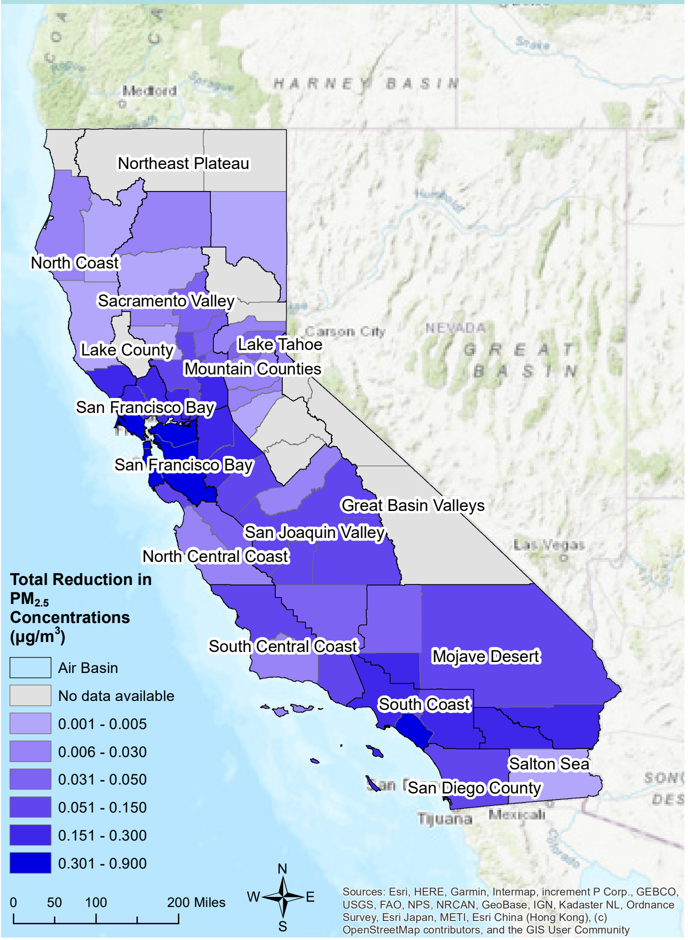
This section also shows the estimated apportionment of emissions for each gas appliance type by air basin, as depicted in Figure 3-2. Variations by air basin result from different usage profiles for the climate zones designated in the RASS (see Appendix A, Section A.2.1 for details). The two air basins not included in this figure (Great Basin Valleys and Lake County) did not have any available gas consumption data.

Moving forward, this report only discusses $NO_X$, as ambient CO is not used in the health-impact calculations and $NO_2$ is a component of $NO_X$. As described previously, we assess the contribution of $NO_X$ to $PM_{2.5}$ using our calculated EFs. We also assess the independent contribution from $PM_{2.5}$ emissions from residential fuel combustion to ambient $PM_{2.5}$, using data extracted from a CARB database. We did not develop EFs for $PM_{2.5}$ in Section 2 due to data paucity.

### 3.2.2. EMISSION REDUCTION DUE TO RESIDENTIAL BUILDING ELECTRIFICATION

We simulated an electrification scenario in which 100% of gas appliances were replaced with clean-energy electric appliances, under the assumption that all emissions described in Section 3.2.1 are eliminated. As mentioned in the beginning of Section 3, we first

**Figure 3-3:** Total reduction in ambient $PM_{2.5}$ concentrations in California from elimination of gas appliances, by county in 2018.




estimated reductions in secondary $PM_{2.5}$ levels, based on our calculated reduction in $NO_X$ (Section 3.2.1) and resulting nitrate $PM_{2.5}$. We then incorporated CARB data on $PM_{2.5}$ emissions from residential gas appliances to estimate the total reduction in $PM_{2.5}$ from replacement of gas appliances, representing changes in primary and secondary (nitrate) $PM_{2.5}$ from gas appliance use. This scenario is described in detail in Appendix A. Overall, this scenario suggests a reduction in the ambient $PM_{2.5}$ concentration by an average of 0.11 μg/$m^3$ per county (see Appendix A.2.3 for details).

Appendix B shows county data for total $PM_{2.5}$ and $NO_X$ emissions, and the estimated emission reductions with building electrification per county. Figure 3-3 shows the geographic distribution of emission reductions due to residential building electrification.

As discussed in Section 3.1.1, there are existing emissions from power plants due to electricity generation.[73] Gas accounts for approximately half of all electricity generation in California,[288] and thus, if the fuel sources of electricity generation were to remain the same, gas usage would increase (and associated emissions from power plants would increase) if the new electric load is not powered by renewable energy resources. However, utilities are making progress to ramp down electricity production from gas and deploy clean energy on the grid, in accordance with the state's zero-carbon requirements. Additionally, taking into consideration California law SB 100 — which requires all of the state's electricity to be generated by zero-carbon resources by 2045 — there will be increasingly less dependence on nonrenewable resources from power plants, and an increased clean energy portfolio that contributes to reduced emissions from power plants.[259] Our analysis does not account for any increases in gas used for electricity generation as a means of looking beyond the transition period to zero-carbon resources.

### 3.2.3. REDUCED MORTALITY (DEATH) AND MORBIDITY (DISEASE) DUE TO ELECTRIFICATION

In this section, we assess the human health impact from emission reductions in the ambient $PM_{2.5}$ levels due to building electrification described in Section 3.2.2. Using the U.S. EPA's BenMAP community edition tool (BenMAP-CE), we estimated all-cause mortality impacts, acute bronchitis impacts, and chronic bronchitis impacts[vi] due to the reduction in $PM_{2.5}$ from the modeled electrification scenario for the year 2018, as described in Section 3.2.2. As described in the Data and Methods section (Appendix A, Section A.2.3), we incorporated impacts from the reduction of both primary and secondary (nitrate) $PM_{2.5}$ from the conversion of $NO_X$ to secondary $PM_{2.5}$.

For the year 2018 (as described in Section 3.2.2), the improvement in outdoor air quality from residential building electrification alone would reduce approximately 354 deaths (all-cause mortality), 304 cases of chronic bronchitis, and 596 cases of acute bronchitis in California (see Table B-5 for confidence intervals for mortality). The most affected counties are the higher-population areas, i.e., Los Angeles County and Orange County, due to the nature of the concentration-response function.

To estimate the monetized benefits of reduced all-cause mortality, we used a Value of a Statistical Life (VSL) estimation in BenMAP, which is commonly used in health impact assessment. For acute and chronic bronchitis, we used a Willingness to Pay (WTP) function, explained in more detail in Appendix A. The mortality reductions

vi. Mortality impact applies to the population aged 30-99; acute bronchitis impact applies to the population aged 8-12; and chronic bronchitis impact applies to the population aged 27-99.

result in estimated monetized benefits of almost $3.3 billion. For reductions of acute bronchitis and chronic bronchitis cases respectively, benefits were estimated at $310,000 and $150 million respectively, in 2019 dollar-values. The total estimated, monetized benefits for all health effects addressed (i.e., all-cause mortality, acute bronchitis, and chronic bronchitis) were estimated to be close to $3.5 billion dollars (see Table B-6 for confidence intervals for mortality).

A summary of all health impact and valuation results is shown in Table 3-1, and annual monetary values for the five California air basins with the highest monetary benefits are shown in Table 3-2. There are additional tables in Appendix B describing health effects and resulting monetary benefits by county and by air basin. Tables B-3 and B-4 [vii] in the appendix include an approximation of mortality and valuation results by air basin for nitrate $PM_{2.5}$ alone and all $PM_{2.5}$. For morbidity and the remainder of this discussion, this report only refers to impacts from total $PM_{2.5}$.

**Table 3-1:** Annual health impacts and monetized benefits from outdoor air quality improvements in a residential electrification scenario.

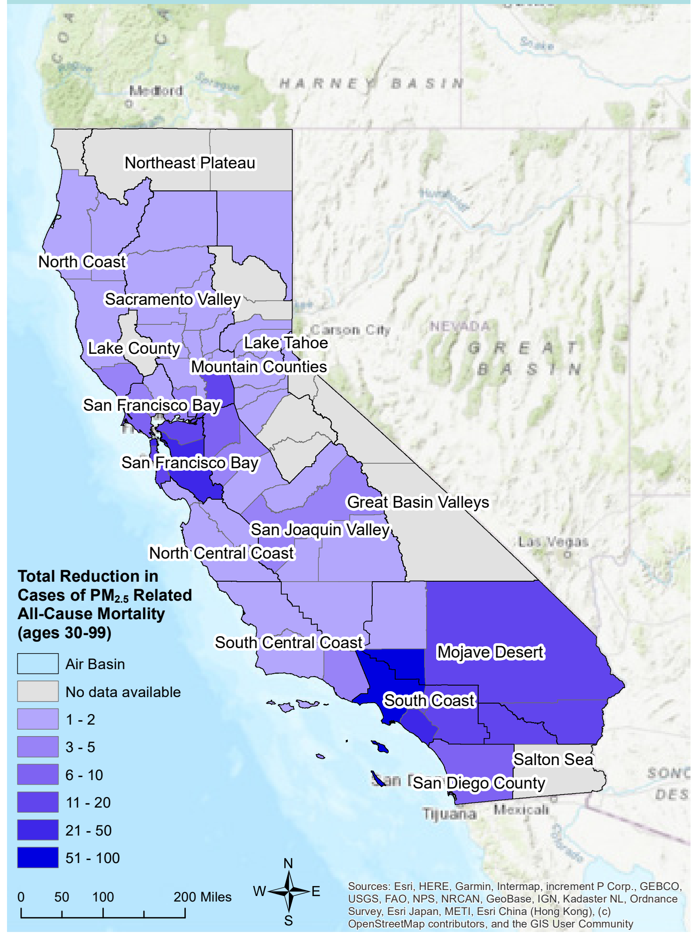
| Health Impact | Avoided Mortality and Morbidity Cases (Annual) | Monetized Benefits (Annual) |
|---|---|---|
| All-Cause Mortality (ages 30 - 99) | 354 | $3.3 billion |
| Acute Bronchitis (children ages 8-12) | 596 | $0.3 million |
| Chronic Bronchitis (ages 27-99) | 304 | $150 million |
| **Totals** | — | **$3.5 billion** |

**Table 3-2:** Estimated annual monetization of health benefits from the electrification scenario by air basin for the five air basins with the highest benefits throughout the state.[viii]

| Air Basin | All $PM_{2.5}$: Mortality Valuation (Annual) | Acute Bronchitis Valuation (Annual) | Chronic Bronchitis Valuation (Annual) |
|---|---|---|---|
| San Francisco Bay Area | $1.2 billion | $100,000 | $58 million |
| South Coast | $1.0 billion | 97,000 | $46 million |
| Mojave Desert | $0.6 billion | 57,000 | $26 million |
| Sacramento Valley | $0.2 billion | 16,000 | $7 million |
| San Joaquin Valley | $0.2 billion | 18,000 | $6 million |

A geographic depiction of mortality reductions by county is provided in Figure 3-4.

**Figure 3-4:** Total reduction in annual cases of $PM_{2.5}$ related all-cause mortality by county in 2018.




We can compare our findings with other recently released reports. As previously mentioned, EPRI released an analysis for the CEC on the air quality and health impacts of a high electrification scenario in California. In their analysis, EPRI found that electrification of multiple sectors in 2050 would result in $108 billion in annual health benefits for California from reductions in $PM_{2.5}$ and ozone.[15] They also found significant, unexpected impacts from the reduction of residential wood combustion — reductions in winter $PM_{2.5}$ from wood-burning are equivalent to reductions from all other sources combined. More than half of EPRI's reported benefits would occur in the South Coast Air Basin. The analysis involved developing a reference and electrification scenario for 2050 based on emission inventories for non-road, stationary, on-road, and power-sector sources, pulled from multiple databases and models. Though there are other challenges in comparison

vii. We separated an air pollution impact of a county evenly across different air basins, in cases where the county is spread over different air basin areas.

viii. The values in this table are rounded to two significant digits. Please see Table B-4 for detailed results.

here as well, considering EPRI's analysis is for 2050 and ours is for 2018, and our analysis only incorporates $PM_{2.5}$ and residential building electrification, and is only including the effects of gas, it is reasonable to assume that our estimated benefits are a small percentage of EPRI's. Another general point of comparison is CARB's GHG inventory cited by the EPRI report, which suggests that the residential sector was responsible for approximately 6% of GHG emissions in 2017, most of which resulted from gas combustion.[289] This can be compared to our residential sector findings of 3% ($3.4 billion) of the monetized benefits EPRI estimated for high electrification of multiple sectors; however, again, our analysis only accounted for $PM_{2.5}$, and not ozone. Thus, they are even more comparable.

A Massachusetts Institute of Technology study published in early 2020 assessed premature mortality from cross-state air pollution in the U.S., accounting for $PM_{2.5}$ and ozone emissions and resulting health effects from exposure.[290] This analysis found that the residential and commercial sectors, which included residential combustion of all fuels as well as other sources (such as waste treatment), were responsible for more than 6,000 premature deaths in California in 2018.[290] Our analysis accounts for a small subset of these mortality rates. We only assess $PM_{2.5}$ and residential gas consumption in our analysis; we do not include other fuels or emission sources, as the 2020 study did. Our findings account for approximately 6% of these premature deaths attributable to the residential and commercial sectors. Another key finding from that particular study: The premature deaths caused by those two sectors is double the number caused by electric power generation processes. The authors of that study clarify that this is a direct result of significant emission reductions in electric power generation processes since 2005.[290]

We were not able to assess outdoor air quality and resulting health effects at the census tract level due to data paucity, but conducting future analyses at that spatial level would enable us to draw quantitative conclusions about the relationship between gas appliance use, electrification, and environmental justice.

### 3.2.4. ASSUMPTIONS AND LIMITATIONS

There are several limitations associated with the analysis presented in Section 3. Data limitations restricted the geographical scale for this analysis, as much data (e.g., on energy use) are only available at the county level and higher. The RASS data we used to estimate total emissions is from 2009, and therefore may not be entirely representative of current usage proportions of different appliances. Due to the limited scope of this project, we were not able to conduct dispersion or photochemical modeling for the distribution of $PM_{2.5}$ emissions and resulting ambient changes, and instead used a back-of-the-envelope calculation for estimating changes in $PM_{2.5}$ at the county level that would result from eliminating residential gas appliances. We also assumed all indoor emissions from gas appliances eventually traveled outdoors, which is a health-protective, conservative assessment. We also did not account for heating demand trends or seasonality in this analysis, although prior studies have separately evaluated winter and summer seasons. Considering these factors, this is a simplified analysis and should be considered a conservative approximation.

Regarding the health impact analysis, it is important to note that the BenMAP software accounts for ambient outdoor $PM_{2.5}$ changes and does not assess population time-activity patterns or personal exposures, which also significantly contribute to health effects.

Finally, there are several facets of electrification that we were unable to include in this report due to the scope of this project. There are some pollutant emissions from the use of residential electric appliances, though not to the same extent as those produced by combustion of gas appliances; we did not account for these electric appliance emissions in this analysis. This report only focuses on residential buildings and does not include an assessment of the health and monetary impacts from the electrification of commercial buildings. Also, it does not assess the costs and potential adverse impacts of residential electrification as some other reports do, such as EPRI's report (on electrification of multiple sectors), which we have discussed and cited.[15]

# 4 CONCLUSION

While California is a leader in clean energy and climate policy, many regions of the state have poor air quality, particularly in state-identified disadvantaged communities. As part of the state's strategy to improve air quality and public health, one area of focus can be reducing emissions from gas appliances through methods such as building electrification. This report presents the adverse health effects resulting from the residential use of gas appliances and outlines the potential benefits of transitioning residential gas appliances to all-electric appliances. These benefits are not only related to GHG emission reductions, but also are related to improving indoor and outdoor air quality, as well as subsequent health and economic effects from pollutant reductions.

The indoor air quality analysis for this report found that concentrations of CO and $NO_2$ during cooking events can exceed the levels set by national and California-based ambient air quality standards, occurring much more often for $NO_2$ than CO. Under a cooking scenario where the stove and oven are used simultaneously for an hour, acute exposures to $NO_2$ from cooking with gas appliances exceed the levels of national and California-based ambient air quality thresholds in more than 90% of modeled emission scenarios. Concentrations of CO and $NO_2$ resulting from gas cooking are the highest for apartments, due to smaller residence sizes. This presents an additional risk for renters, who are often lower income than homeowners. Considering the well-known dangers of CO poisoning, and that acute and chronic exposures to $NO_2$ are associated with respiratory illness and mortality, this is a serious concern that should not be overlooked. We echo other researchers in this space with the recommendation that proper ventilation technology, such as effective, low-noise range hoods, be implemented to reduce exposure and protect public health.

Regarding outdoor air quality, this report indicates that under a 2018 scenario where all residential gas appliances were transitioned to electric, the reduction of secondary nitrate $PM_{2.5}$ (from $NO_X$) and primary $PM_{2.5}$ would result in 354 fewer deaths, and 596 and 304 fewer cases of acute and chronic bronchitis, respectively. The reduction in associated negative health effects is equivalent to approximately $3.5 billion in monetized health benefits for just one year.

However, these health and monetary benefits will not be realized at the pace or scale needed without policymaker support. Decision-makers at state and local agencies that regulate air quality all have important roles to play in determining the best course of action for reducing pollution from gas appliances, and doing so in a way that prioritizes and protects those most burdened by air pollution — namely, low-income and environmental justice communities. Implemented strategically, new policies to reduce air pollution from residential buildings will yield significant health benefits, improve the quality of life for Californians, and reduce greenhouse gas emissions.

# APPENDICES

## Appendix A: Data and Methods

### A.1. INDOOR AIR QUALITY & HEALTH EFFECTS (SECTION 2 IN THE REPORT)

#### A.1.1. EMISSION FACTOR DATABASE

We developed an EF database of CO, $NO_2$, and $NO_X$ for different gas appliances, including stoves and ovens, heating devices, and water heaters. In cases where EFs were not available for some pollutants (due to data paucity and/or feasibility), such as $PM_{2.5}$, UFP, and formaldehyde, a qualitative analysis of related emissions and associated health impacts was conducted.

***Aggregating appliance characteristics from online resources***

First, we summarized real-world (measured) EFs of gas appliances from existing peer-reviewed or grey literature. While most previous studies reported the EF of gas appliances in a unit such as ng/J, those values do not reflect the amount of pollutants released during the consumption without accounting for the MBR in J/h of different gas appliances. Hence, we also summarized the MBR of various gas and electric appliances from approximately 15 main appliance brands using online resources, including websites for companies such as Home Depot, Lowe's, Amazon, etc. Our internet search terms to select brands and extract information included, "gas and electric ovens," "cooktops," "popular gas appliances," "range," "gas range," "electric range," "furnace," "water heater," and "fireplace." We gathered information regarding different models for each brand, including price(s), heat output in British thermal units (BTUs — which were converted to J/h), and specification characteristics.

***Extracting emission rates from primary literature and determining significant explanatory variables***

We acknowledge that the EF (ng/J) can be influenced by many factors, such as appliance age, location, and ventilation conditions. Thus, we collected information on these parameters in conjunction with EFs from the aforementioned literature. We performed a multiple linear regression analysis to quantify the contribution of different factors to the EFs of CO, $NO_2$, and $NO_X$, with EFs and various factors as dependent and independent variables, respectively. We ran three models using RStudio software, with emission rates of CO, $NO_2$, and $NO_X$ as the outcome variable.

$$Log\ (Emission\ Factor) = \beta_0 + \beta_1\alpha + \beta_2\chi + \beta_3\delta + \beta_4\phi + \beta_5\eta + \beta_6\lambda \quad (1)$$

where, $\alpha$ = appliance directly vented (yes/no), $\chi$ = energy use (J/h), $\delta$ = appliance age, $\phi$ = appliance type, $\eta$ = laboratory or residence setting, and $\lambda$ = year of study.

The EFs from literature used as the outcome variable in the regression analysis, along with the associated covariates, were primarily extracted from a 2009 report produced by LBNL in California, which was ideal since it provided very detailed information on sampling methods and results.[44] Since this dataset is the most recent source and specific to our study area, this was our primary data source for the quantitative analysis. We also gathered emission rates from other papers, some of which dated to the 1970s (so, covering the last 50 years); we gathered as much data as was feasible during the timeframe. We only used data from the United States, and most data used was California-based, but we needed to use several other studies as well, to optimize the regression models. Of course, there have been technological advances that have reduced emission rates over time, and this was factored into the regression model. Appliance ages spanned from 1-20 years old. Most of the data used fit the regression line well.

Based on the results of normality tests, we log-transformed the data. We ran the model with multiple specifications, including with and without the oldest data points, and with various appliance groupings, to ensure our model was optimally fitted. The final sample sizes (emission rates with values for each covariate in the regression equation), including all appliance types, were approximately 55 for each pollutant.

The regression models all had $R^2$ values of ≥ 0.6 (see Figure B-1 in Appendix B), which indicates that the models fit well, and the dependent variables of pollutant EFs in ng/J are highly predictable from the data. Three supplemental figures with the regression lines, J/second distribution (prior to conversion to J/h) and predicted emission rates are included in Appendix B (Figures B-1, B-2, and B-3).

We identified factors that are significantly associated with EFs. Then, by reviewing existing literature and databases, we obtained or assigned appropriate values for those factors to better reflect the real-world scenario in California, and used bootstrapping statistics to simulate the distribution of each factor — and thus, EFs in California, accordingly, with 1000 bootstraps for each factor. This nonparametric technique involved resampling data to estimate data distributions and is widely used. The MBR values, in J/h, were gathered from online resources and converted from BTU/h. The ventilation characteristics we used for stoves and ovens were based on the general assumptions that stoves and ovens are not vented, and heating devices and water heaters are vented [44] (assumed due to existing regulations). According to California Health and Safety Code, gas-fueled, unvented space heaters cannot be sold,[76] the CEC mandates direct venting of water heaters to outside spaces,[291] and the California Mechanical Code regulates the venting of other fuel-burning devices as well (additional information on these types of regulations is provided in Section 2.1.2 of the report).[292] The age of appliances was gathered from the RASS. We calculated EFs for stoves and ovens separately, but combined them for much of this analysis under the assumption that the devices were operating at the same time and were entirely vented into the kitchen, though we do include some separate considerations for stove use only. We did not adjust for variable range hood use, due to survey results showing low rates of range hood use in California.[98] Therefore, this aspect of the analysis is more conservative.

***Developing emission rate database***

With available data on EFs (ng/J) and MBR (J/h), we derived a new emission rate database for $NO_2$, $NO_X$, and CO in a unit of ng/h, which we converted to µg/h, which reflects the emission rate, or the amount of pollutants released in a specific time period during the usage of different gas appliances. To validate our results, we compared our calculated emission rates to the WHO's household fuel combustion emission rate targets[293] as well as primary literature. We used Microsoft Excel to conduct the bootstrapping and initial model setup. The equations used for calculating EFs and emission rates are listed here:

$$\begin{gathered} EF\ (\text{in a unit of ng/J}) = f(\alpha, \chi, \delta, \phi, \lambda) * \text{prediction uncertainty} \\ \text{Emission rate (in a unit of ng/h)} = BTU * f(\alpha, \chi, \delta, \phi, \lambda) * \text{prediction uncertainty} \end{gathered} \tag{2}$$

where, $\alpha$ = appliance directly vented (yes/no), $\chi$ = energy use (J/h), $\delta$ = appliance age, $\phi$ = appliance type, and $\lambda$ = year of study.

## A.1.2. INDOOR AIR QUALITY IMPACTS AND SUSCEPTIBILITY

We estimated the impact of the gas appliances that are not vented to the outdoors (stoves and ovens) on indoor air quality using our developed EF database and a mass balance model:[122]

$$C = p\left(\frac{Q}{Q+kV}\right)C_O + \frac{S}{Q+kV} \tag{3}$$

where, V = volume of the indoor space ($m^3$), Q = ventilation rate ($m^3$/h), S = emission rate (µg/h), $C_0$ = outdoor concentration (µg/$m^3$), C = indoor concentration (µg/$m^3$), k = deposition rate ($h^{-1}$), and p = penetration factor (unitless).

We used values for deposition rate and penetration factor from recent journal articles.[47,52] For outdoor concentrations, we used average California values from the EPA's Air Data portal for 2018.[294] The methods for determining volume and ventilation rate, which varied for different housing types, are described later in this section.

For this analysis, we defined two indoor environments, one with the use of gas appliances and one without, to determine the contribution of gas appliances to indoor air pollution, the latter of which represents an electrification scenario; we assumed there were no emissions of combustion pollutants with the use of electric appliances. Under the assumption of a steady state, we calculated the increment of indoor levels of CO, $NO_2$, and $NO_X$ due to gas appliance use by comparing C in models with or without emissions (S) from gas appliance usage. The elevation of pollutants' concentrations due to gas appliances was weighted by the time of their usage (e.g., time-activity patterns) over 24 hours to estimate the contribution to chronic exposures. We modeled the use of kitchen appliances under three cooking scenarios: 15 minutes, 1 hour, and 2 hours per day.[ix] These timeframes

ix. We took this approach because we believe it is valuable to compare cooking times of 15 minutes, 1 hour, and 2 hours, since one objective of this report is to help the public understand these concerns, and considering that some households may cook more or less than others, we wanted to provide this range.

were chosen based on a cooking appliance use survey indicating a total daily cooking time of around 1 hour for breakfast, lunch, dinner, and other meals, for both stoves and ovens, though we included multiple timeframes to account for a wide range of cooking patterns.[98] For CO, we also weighted the elevated concentrations over 8 hours for comparison with the national 8-hour standard of 10 mg/m$^3$ for both California and the EPA.[295,296] The model also produced an output with the highest concentration value, representing the emissions while cooking; we used this to establish peak concentration levels. For peak concentration levels, we used kitchen-specific volumes, and for values weighted by usage time, we used entire residence volumes under the assumption that pollutants would mix into the residential space over time.

We modeled the increments of CO, $NO_2$, and $NO_X$ due to gas appliance use in three common residential building types: SFHs, apartment buildings, and townhouses. As shown in Equation (3), the relationship between C and S is dependent on other building parameters associated with indoor air quality, suggesting differential impacts of gas appliances in different building types. Therefore, we collected data on building design parameters (e.g., air exchange rate and ventilation, residence volume) by housing type in California from regulatory standards, AHS, primary literature, and various other reports on building ventilation and other factors. To estimate kitchen volume, we assumed kitchens occupied 10% of the house volume. This is near the lower end of the range found in our literature review, and hence, another conservative assumption. We used bootstrapping to simulate the distributions of the various housing parameters, and incorporated them into Equation (3) to finalize our indoor air quality estimates. These findings were simplified into three boxplots, by pollutant (Figure 2-1).

For assessing the impact these concentrations and associated exposures may have on health, we stopped evaluating $NO_X$ separately, since $NO_2$ is established to be the primary health concern out of the nitrogen oxides, as well as a main combustion pollutant with established ambient air quality standards.

As noted in Section 2.2.2, we used California (CAAQS) and U.S. EPA ambient (outdoor - NAAQS) air quality standards as a metric for health effects from exposure. These standards are the maximum allowable concentration of a pollutant present in outdoor air that will not have a known, adverse impact on human health and are developed to apply to long-term, ambient outdoor air quality, averaged over time periods. It is not possible to actually exceed these outdoor standards in an indoor environment due to the technical definition. Therefore, we apply target thresholds using the standards as a guide to provide context for indoor air quality. We refer to three different types of thresholds based on the standards: 1) Acute (1-hour for $NO_2$ and CO), 2) 8-hour (for CO), and 3) chronic (annual mean for $NO_2$; there is no annual mean standard for CO). When we use the term "acute," we are referring to 1-hour standards. For CO, we refer to 8-hour standards directly as such. For $NO_2$, when we use the term "chronic," we are referring to the annual mean standards.

When an exceedance is referenced in this report, it means that the modeled indoor air concentration is higher than the threshold levels based on the standards in Table B-7. When we refer to the percentage of exceedances, we are discussing the percent of our modeled indoor air quality estimates that exceed thresholds. We evaluated the indoor air quality exceedances of the CAAQS and NAAQS thresholds, overall and for separate residence types, for CO and $NO_2$ (there are no applicable standards for $NO_X$).

To assess acute exposures of these two pollutants, we compared peak concentrations to acute (1-hour) CAAQS and NAAQS. This is like calculating a hazard quotient (HQ) as is done in risk assessment, which would be a ratio of the concentrations to the established standards. If the HQ is less than 1 (essentially, if the maximum concentration does not exceed the standard), adverse health impacts are not expected. We compared the peak concentrations to 1-hour CAAQS of 339 μg/m$^3$ (180 ppb) for $NO_2$ and 23 mg/m$^3$ (20 ppm) for CO and 1-hour NAAQS (US EPA standards) of 188 μg/m$^3$ (100 ppb) for $NO_2$ and 40 mg/m$^3$ (35 ppm) for CO.[295] Exceedances of the thresholds for our estimated peak concentrations (based on 1-hour standards) only apply under a scenario where cooking occurs for the entire hour and the air quality levels remain elevated.

To assess chronic exposures, we compared chronic exposure concentrations for $NO_2$ to the annual mean CAAQS (57 μg/m$^3$ or 30 ppb) and NAAQS (100 μg/m$^3$ or 53 ppb) as well.[295,296] For CO, we compared our 8-hour averages to the 8-hour threshold of 10 mg/m$^3$ (9.0 ppm) for both CAAQS and the NAAQS.[295] All acute, chronic, and 8-hour standards are listed in Table B-7. We also qualitatively discussed long-term health impacts of the pollutants (Section 2.2.4) as described in the literature, since chronic impacts are less well-established for $NO_2$ and CO (though the epidemiological literature on $NO_2$ and mortality is expanding, which we discuss in detail in the results section). More information on this is provided in Section A.1.3.

We conducted two sensitivity analyses: we estimated indoor air concentrations due to potential use of kitchen appliances for supplemental heating, and we estimated concentrations resulting from improper ventilation systems for appliances required to be vented outdoors (within the results, we discuss peak concentrations in the entire home due to the ventilation issues).

We evaluated exposure susceptibility qualitatively to the extent possible (Section 2.2.3), including equity considerations. There is insufficient data to quantitatively estimate exposure disparities in different groups. It is important to note that risk is highly dependent on exposure parameters (e.g., inhalation rate and body weight). Additionally, different populations' exposures are affected by their activity patterns as well as by the environmental concentrations of pollutants. We qualitatively discussed the exposure levels of populations with different characteristics.

### A.1.3. HEALTH EFFECTS OF AIR POLLUTION

We reviewed previous human research studies focusing on pollutants that could be emitted by gas appliances. Specifically, controlled-exposure experimental and time-series epidemiologic studies were reviewed to evaluate the adverse health effects of short-term exposures, while cross-sectional studies were reviewed for long-term effects. In Section 2.2.4, we summarized potential outcomes associated with CO, $NO_2$, and $NO_X$ at all levels, including those comparable to our modeled concentrations. Additionally, we summarized the current literature on the health impacts of exposure to PM and formaldehyde.

## A.2. OUTDOOR AIR QUALITY & HEALTH EFFECTS (SECTION 3 IN THE REPORT)

### A.2.1. CONTRIBUTION TO TOTAL EMISSIONS OF OUTDOOR AIR POLLUTANTS IN CALIFORNIA

We estimated the total emissions of outdoor air pollutants from gas appliances in California, at the county and state-wide levels. To do this, we used the EF database created in Section 2 in combination with total gas consumption (from the CEC) to calculate the total emissions of CO, $NO_2$, and $NO_X$ in tons/year.[297] Since consumption patterns related to appliances vary by region (e.g., in some regions 40% of the energy may be used by water heaters, but in a different area, more energy is designated to heating devices than water heaters), we used the RASS relative appliance energy usage splits by climate zones to estimate emissions in each county by assigning counties to each climate zone. Total emissions of each pollutant by county are depicted in Figure B-4. We compared the contributions from gas appliances to $NO_X$ emissions with other sources in California, which is shown in Figure B-5. With the data of energy consumption by different gas appliances in California,[287] we also estimated the emissions by appliance type, and the result is depicted in Figure 3-1.

It is also important to note that for this second section, we incorporated emissions from all types of appliances, while for the indoor air quality evaluation, we primarily considered emissions from appliances that are not vented to the outdoors (e.g., stoves and ovens), and also conducted a sensitivity analysis, incorporating scenarios in which venting technology for water heaters and home heating devices failed to transport all combustion pollutants outdoors. For this section, we operated under the conservative assumption that all indoor emissions are transporting outside.

### A.2.2. EMISSION REDUCTION DUE TO ELECTRIFICATION

We simulated $NO_X$ emissions (which include $NO_2$) under the assumption that all the energy generated by gas appliance usage is replaced by clean electricity. We modeled and evaluated an all-electrification scenario, as compared to the "business as usual" scenario with no replacement of appliances (though this scenario is unrealistic, as normal replacement rates are a part of "business as usual"). This simulates 100% replacement of gas appliances with electric appliances.

Our modeled scenario, which is for the year 2018, was based on the assumption of adoption of entirely clean electric technologies. We discussed the limitations associated with this, with consideration to the emissions from electricity generation at power plants,[298] and the reduction in EFs over time.

### A.2.3. REDUCED AMBIENT $PM_{2.5}$ CONCENTRATIONS AND RESULTING MORTALITY AND MORBIDITY IMPACTS DUE TO ELECTRIFICATION

Using the same scenario listed in Section A.2.2 and the previously calculated reduction in total outdoor emissions, we estimated the potential mortality and morbidity impacts (only for acute and chronic bronchitis) in California due to residential building electrification and the resulting reduction in ambient outdoor concentrations of $PM_{2.5}$. This analysis was entirely separate from the indoor air exposure analysis in Section 2. Again, we operated under the conservative assumption that all indoor emissions are transported outdoors.

Approximately 40% of $NO_X$ converts into nitrate-$PM_{2.5}$ after emission. Thus, we first estimated reductions in secondary $PM_{2.5}$ levels by county (later aggregated to air basin) due to reduction in $NO_X$ and resulting nitrate $PM_{2.5}$, using methods described in a recently published paper.[299]

$$\Delta PM_{2.5\ ij} = PM_{2.5\ (ambient)ij} * \ k \ * \frac{\Delta NOx_{ij}}{NOx_{ij}} \qquad (4)$$

In Equation 4, i represents the year, $j$ represents the area of analysis (county), and $k$ is the conversion rate for $NO_x$ to nitrate (0.4).[300] $\Delta NO_x ij$ is the reduction in $NO_x$ emissions in that particular county and year, and $NO_x ij$ is the total $NO_X$ emissions in county $j$ in that year, extracted from CARB's State Implementation Plan Standard Emission Tool database.[73] $PM_{2.5} ij$ is the nitrate $PM_{2.5}$ level in each county, and was calculated by averaging nitrate $PM_{2.5}$ data from the US EPA Air Data portal.[294]

We did not develop EFs for primary $PM_{2.5}$ in our study, due to data paucity and uncertainty regarding how much gas combustion contributes to $PM_{2.5}$ (our literature review provided sufficient evidence of a relationship between UFPs and gas combustion, but not solely $PM_{2.5}$ in papers published in recent decades, particularly for kitchen appliances). Since we were not able to calculate EFs for $PM_{2.5}$ for reasons stated previously, we extracted CARB estimates of emissions for $PM_{2.5}$ from residential gas appliances. We used Equation 5 shown here to calculate changes in ambient $PM_{2.5}$ levels; the baseline $PM_{2.5}$ levels were extracted from the US EPA Air Data portal:

$$\Delta PM_{2.5\ ij} = PM_{2.5\ (ambient)ij} * \frac{\Delta PM2.5_{ij}}{PM2.5} \qquad (5)$$

We then summed the two changes in $PM_{2.5}$, with the final $PM_{2.5}$ increase representing both changes in primary and secondary (nitrate) $PM_{2.5}$ from gas use.

We used the EPA's BenMAP tool to estimate the mortality and morbidity (acute and chronic bronchitis) impacts for the scenario as compared to "business as usual," using the standard U.S. EPA preloaded selections when possible. BenMAP uses established concentration-response functions to quantify mortality from increased $PM_{2.5}$ pollution. Inputs included: the change in $PM_{2.5}$ calculated in Equation 5, population rates,[301] incidence rates,[302] and a $\beta$ value,[156,303] which represents the health impact per unit change of pollution and is drawn from epidemiologic literature. We used California-specific input values.

Additionally, we used our BenMAP mortality and morbidity outputs to monetize the benefits of electrification. This process was done using the BenMAP software (it has VSL calculations and other specific morbidity valuation functions as well), except for chronic bronchitis, which has a functionality issue within BenMAP that we confirmed with U.S. EPA staff. We calculated valuation for chronic bronchitis manually. As stated in Section 3.2.3, we used VSL estimates for monetizing mortality benefits, which is standard in health impact assessment literature. For acute and chronic bronchitis, we used the WTP metric for valuing illness. The U.S. EPA BenMAP manual defines WTP as "the willingness of individuals to pay for a good or service, such as a reduction in the risk of illness," and it is considered conservative.[304] We adjusted all of the monetary outputs for inflation by converting them to 2019 dollars, using the Bureau of Labor Statistics' CPI Inflation Calculator.

The mortality and valuation results are presented in Tables 3-1 and 3-2 in the report, and Tables B-3 to B-6 in Appendix B.

# Appendix B: Supplemental Figures and Tables

**Figure B-1:** Relationship between predicted and measured EFs of (a) CO, (b) $NO_2$, and (c) $NO_X$. (Blue line = correlation between predicted EFs and measured EFs.)

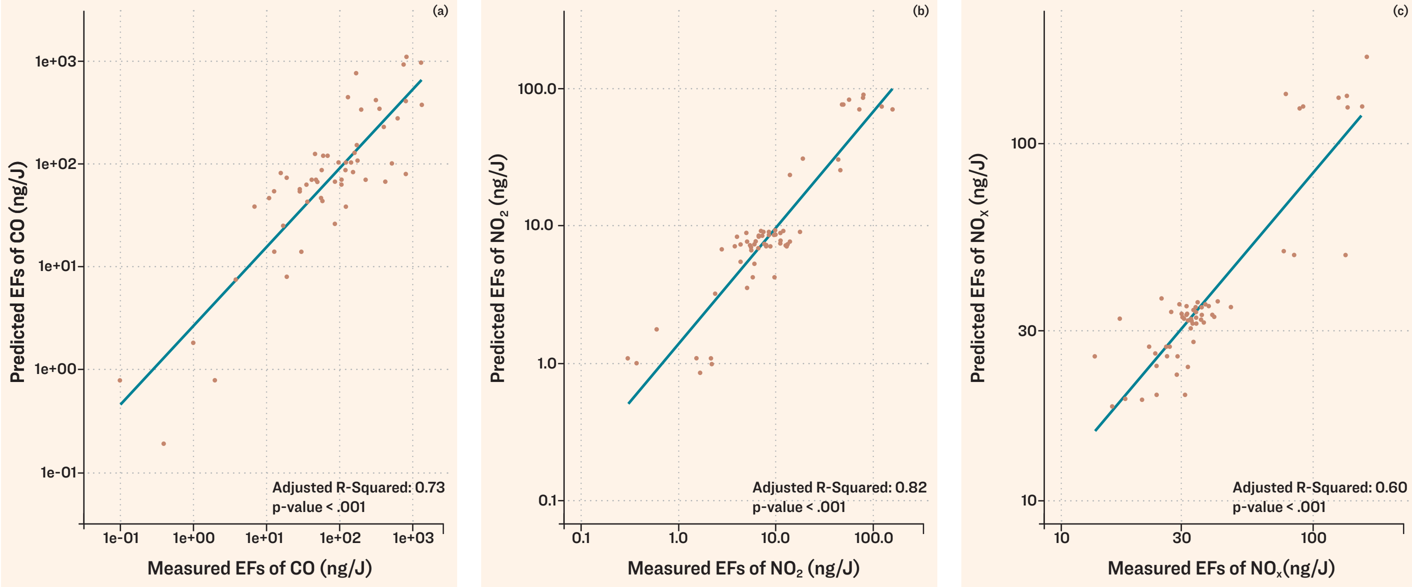



**Figure B-2:** Energy use histogram of (a) heating devices, (b) water heaters, (c) ovens, and (d) stoves, gathered from online resources.

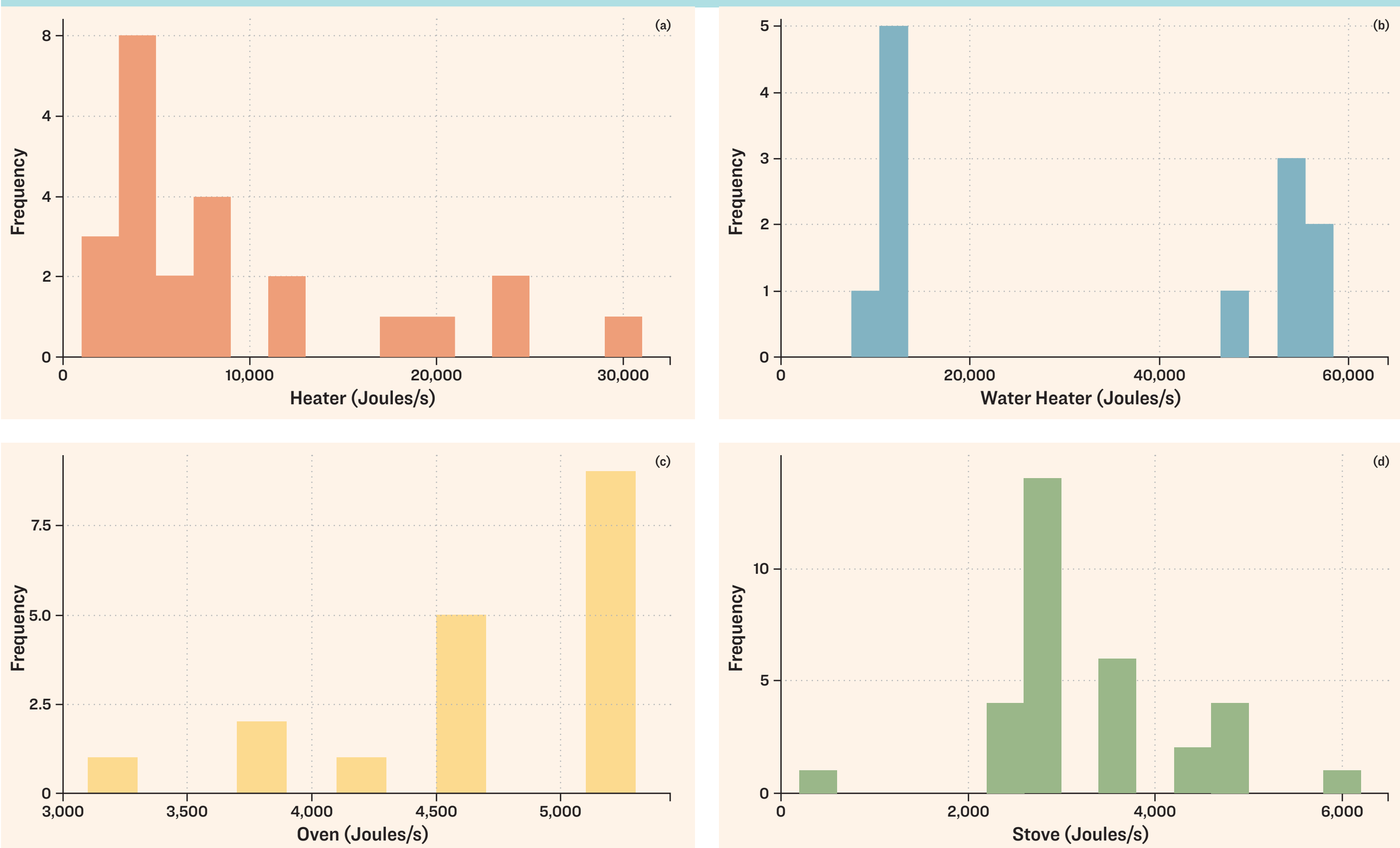


**Figure B-3:** Predicted emission rates of (a) CO, (b) $NO_2$, and (c) $NO_X$ in μg/h for various gas appliances.

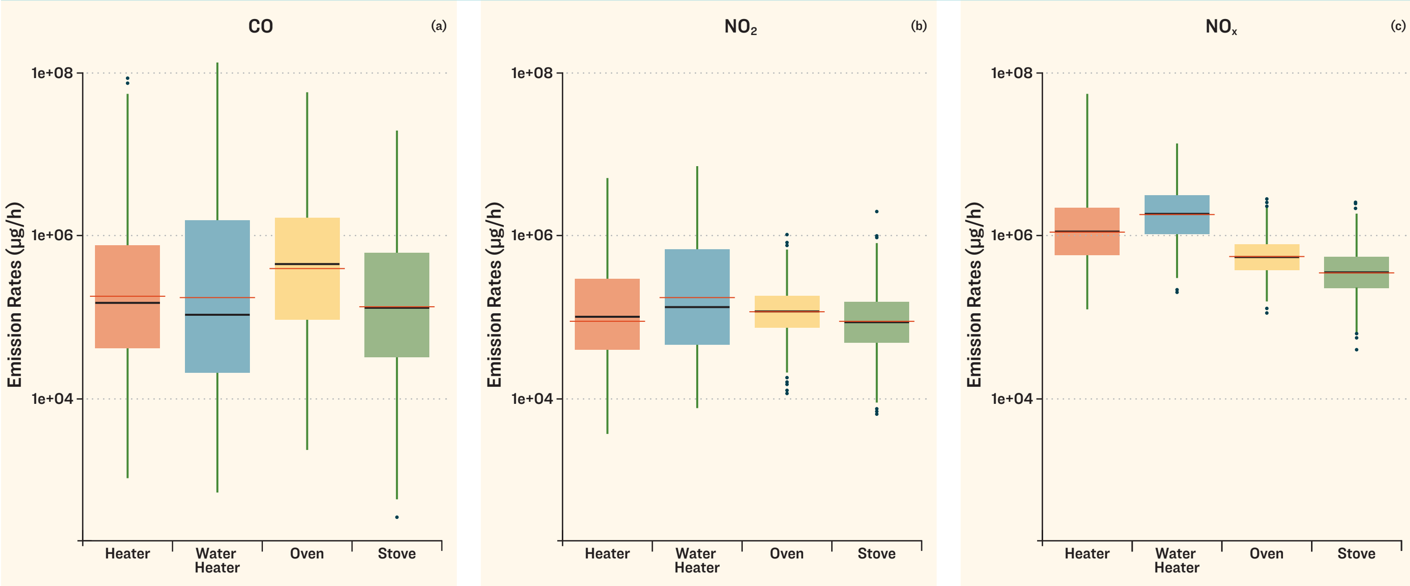


**Figure B-4:** Total emissions of air pollutants (a) CO, (b) $NO_2$, and (c) $NO_x$ by county (gray outlines) in 2018.

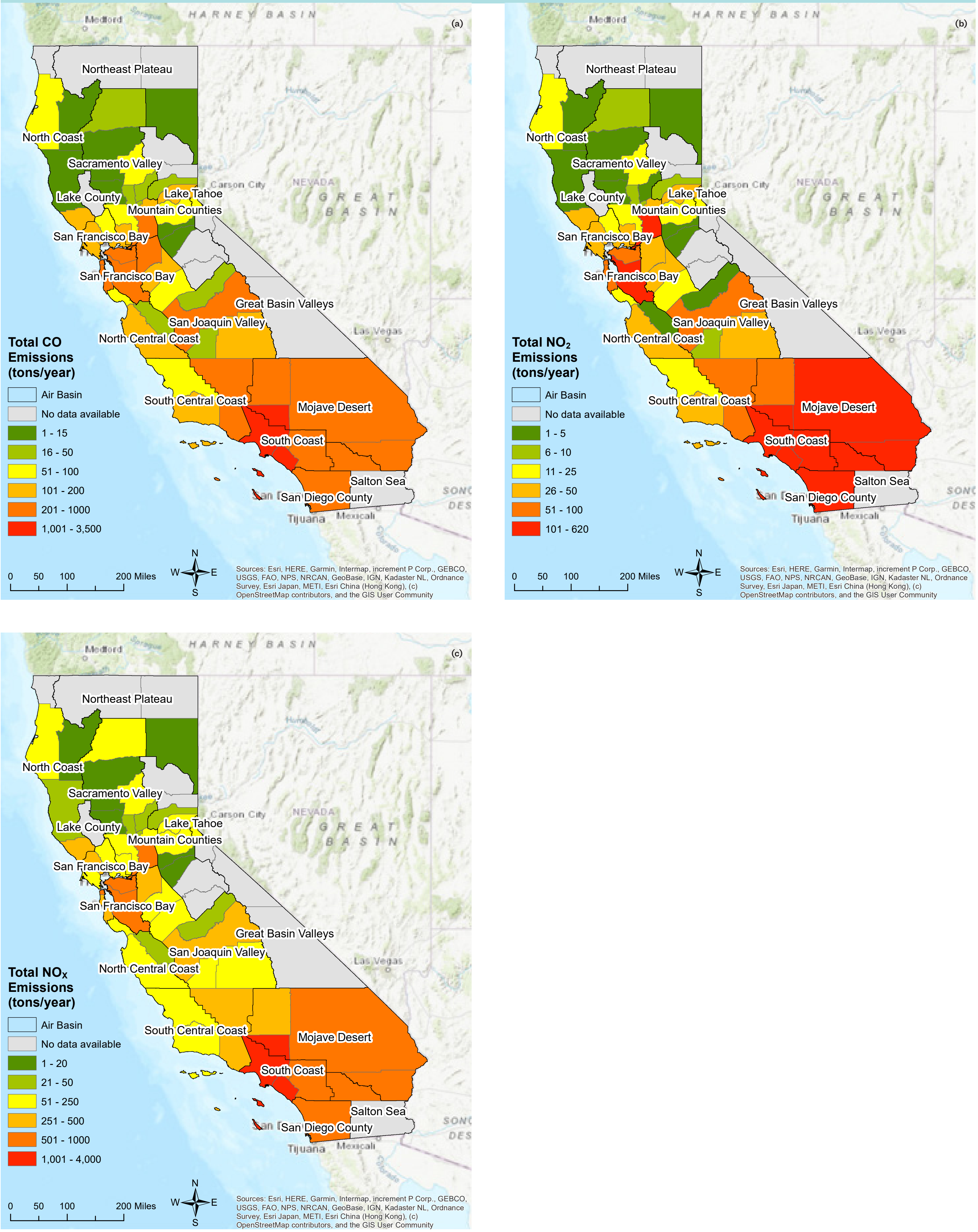


**Figure B-5:** $NO_X$ emissions from residential gas appliances as compared to $NO_X$ emissions from all sources (all California counties).

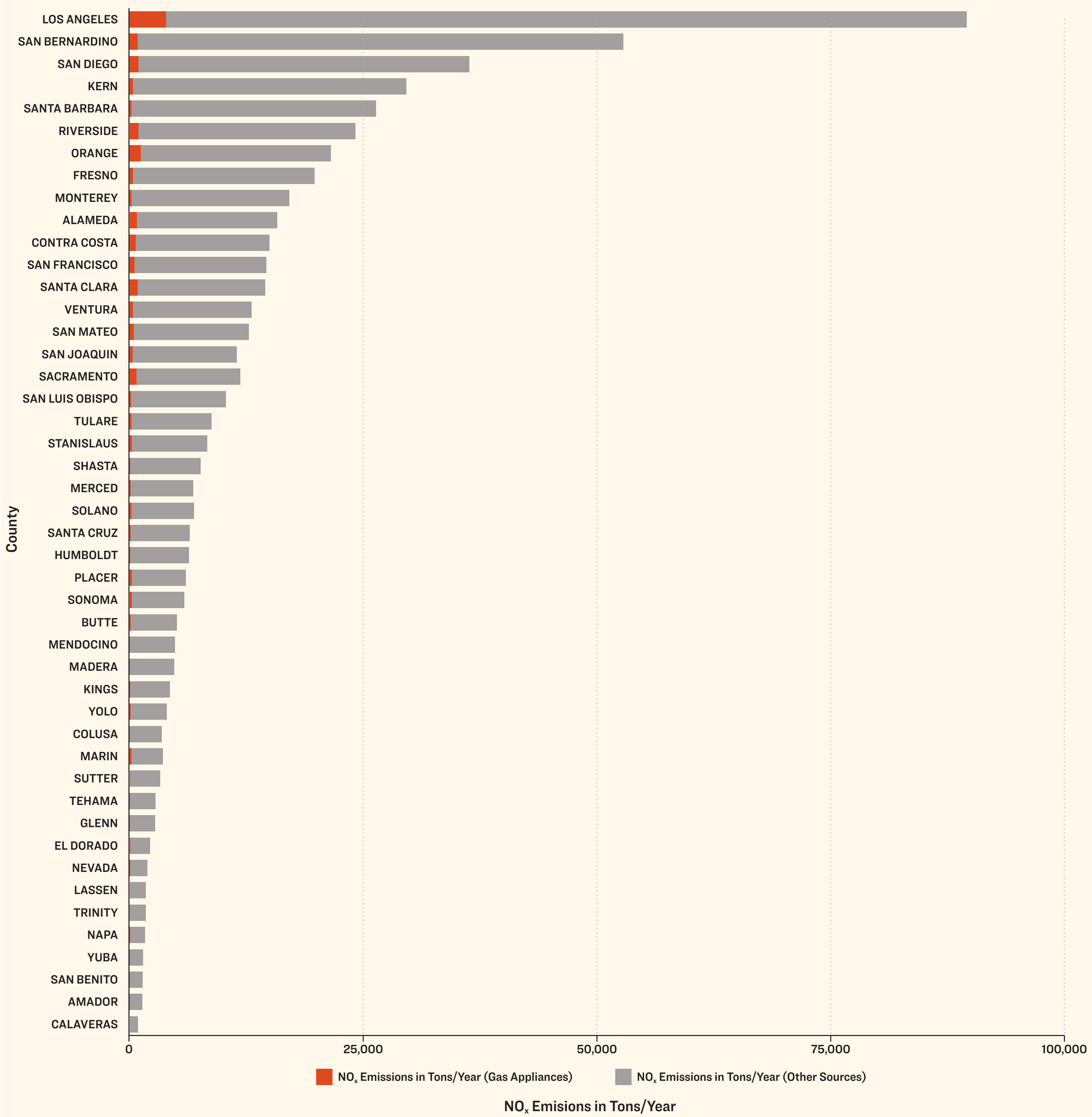


**Table B-1:** Pollutants other than CO, $NO_2$, and $NO_x$ emitted during gas appliance use as identified in various studies and reports, and a summary of associated key findings and health outcomes.

| Pollutant | Source | Description of Study | Key Findings Related to Pollutant | Example of Health Impacts of Exposure to Pollutant |
|---|---|---|---|---|
| Acetaldehyde | Fortmann, R., Kariher, P. & Clayton, R. 2001[27] | Assessed emissions of multiple pollutants during typical stove and oven use activities in a California home (both gas and electric ranges). | Acetaldehyde is present in air samples collected during fish broiling, oven self-cleaning, and pork roast tests. These findings, though limited, provide evidence that cooking may have a substantial effect on aldehyde levels more broadly. | Carcinogenic, sensory irritant, and affects the respiratory system.[305,306] |
| | Mullen, N. A., Li, J. & Singer, B. C. 2012[45] | Concentrations of CO, $NO_x$, $NO_2$, formaldehyde, and acetaldehyde were measured over 6-day periods in 155 California homes to assess associations of pollutant concentrations with natural gas appliances. | The geometric mean of acetaldehyde concentrations in both kitchens and bedrooms was 9 ppb, which was much higher than outdoor concentrations. However, acetaldehyde concentrations were not significantly affected by gas appliance use. | Carcinogenic, sensory irritant, and affects the respiratory system.[305,306] |
| | Mullen, N. A. et al. 2016[46] | To assess the indoor air quality impacts of gas appliance use, collected indoor and outdoor measurements of pollutants at 352 California homes with natural gas appliances, and conducted interviews with residents. | This study did not find statistically significant changes in acetaldehyde levels due to gas appliance use. | Carcinogenic, sensory irritant, and affects the respiratory system.[305,306] |
| | Singer, B. C. et al. 2009[44] | Measured emission rates of multiple pollutants from natural gas combustion in various types of stoves, ovens, broilers, water heaters, and furnaces. This was a large study with multiple objectives. | Acetaldehyde emission rates were low for all burners. | Carcinogenic, sensory irritant, and affects the respiratory system.[305,306] |
| Formaldehyde | Fortmann, R., Kariher, P. & Clayton, R. 2001[27] | Assessed emissions of multiple pollutants during typical stove and oven use activities in a California home (both gas and electric ranges). | Formaldehyde was present in air samples collected during fish broiling, oven self-cleaning, and pork roast tests. This study reported formaldehyde concentrations far above the acute Reference Exposure Level set by OEHHA during gas cooking, both with and without food. These findings, though limited, provide evidence that cooking may have a substantial effect on aldehyde levels more broadly. | Carcinogenic, sensory and respiratory irritant, causes nausea and headache.[103,200,201,206,307,308] |
| | Mullen, N. A., Li, J. & Singer, B. C. 2012[45] | Concentrations of CO, $NO_x$, $NO_2$, formaldehyde, and acetaldehyde were measured over 6-day periods in 155 California homes to assess associations of pollutant concentrations with natural gas appliances. | The geometric mean of formaldehyde concentrations in both kitchens and bedrooms was 15 ppb, which was much higher than outdoor concentrations. About 95% of homes had indoor formaldehyde levels above the Chronic Reference Exposure Level set by OEHHA. However, formaldehyde concentrations were not significantly affected by gas appliance use. | Carcinogenic, sensory and respiratory irritant, causes nausea and headache.[103,200,201,206,307,308] |
| | Mullen, N. A. et al. 2016[46] | To assess the indoor air quality impacts of gas appliance use, collected indoor and outdoor measurements of pollutants at 352 California homes with natural gas appliances, and conducted interviews with residents. | This study did not find statistically significant changes in formaldehyde levels due to gas appliance use. | Carcinogenic, sensory and respiratory irritant, causes nausea and headache.[103,200,201,206,307,308] |
| | Singer, B. C. et al. 2009[44] | Measured emission rates of multiple pollutants from natural gas combustion in various types of stoves, ovens, broilers, water heaters, and furnaces. This was a large study with multiple objectives. | Formaldehyde emission rates showed high variability across all burners, but were particularly low in storage water heaters and high in tankless water heaters. | Carcinogenic, sensory and respiratory irritant, causes nausea and headache.[103,200,201,206,307,308] |

**Table B-1:** Pollutants other than CO, $NO_2$, and $NO_x$ emitted during gas appliance use as identified in various studies and reports, and a summary of associated key findings and health outcomes, cont.

| Pollutant | Source | Description of Study | Key Findings Related to Pollutant | Example of Health Impacts of Exposure to Pollutant |
|---|---|---|---|---|
| Polycyclic Aromatic Hydrocarbons (PAHs) | Fortmann, R., Kariher, P. & Clayton, R. 200[127] | Assessed emissions of multiple pollutants during typical stove and oven use activities in a California home (both gas and electric ranges). | PAHs were found in cooking oils used, though PAH air concentrations were low; the study concluded that additional research on PAHs is necessary in order to fully assess the impact of cooking on PAH concentrations. | Varies by PAH. Examples are carcinogenic and teratogenic effects, and various impacts from oxidative stress.[311,312] |
| | Ruiz, P. A. et al. 2010[66] | Conducted indoor and outdoor sampling of 16 homes with unvented space heaters using different energy sources (electric/central heating, compressed natural gas, liquified petroleum gas, and kerosene) in Chile. | This study found high levels of PAHs in homes with kerosene space heaters. The impacts of gas space heaters were less significant. | Varies by PAH. Examples are carcinogenic and teratogenic effects, and various impacts from oxidative stress.[311,312] |
| | Yu, K.-P. et al. 2015[241] | Sampled particle concentrations and PAHs in five gas cooking kitchens of non-smoking families in Taiwan and conducted a health risk assessment. | This study found that PAH concentrations were correlated with PM concentrations, and PAH cooking exposures could result in cancer risks exceeding the well-established threshold of $10^{-6}$. | Varies by PAH. Examples are carcinogenic and teratogenic effects, and various impacts from oxidative stress.[311,312] |
| | Dutton, S. J., Hannigan, M. P. & Miller, S. L. 2001[90] | Monitored emissions of $NO_2$, CO, and PAHs from unvented natural gas fireplaces in two Colorado residences. | The concentrations measured here were more than an order of magnitude larger than ambient measurements in urban areas. This study highlights the need for research to assess the effects of PAH exposure further. | Varies by PAH. Examples are carcinogenic and teratogenic effects, and various impacts from oxidative stress.[312,313] |
| Sulfur Dioxide ($SO_2$) | Jones, A. P. 1999[23] | This review paper assessed indoor air quality and health. | This review identifies that indoor $SO_2$ concentrations can be high in homes with poorly vented gas appliances and kerosene space heaters, citing studies (before 2000) that have sampled homes with both types of appliances. | Respiratory symptoms and disease, premature mortality.[313,314] |
| | Triche, E. W. et al. 2005[315] | Assessed respiratory symptoms and exposures of almost 900 women who used secondary heating devices, including gas space heaters, during winter. | A 10-ppb increase in $SO_2$ was associated with an increase in respiratory symptoms (wheezing and chest tightness), though kerosene heaters evaluated in this study were the primary source of $SO_2$. | Respiratory symptoms and disease, premature mortality.[313,314] |

**Table B-1:** Pollutants other than CO, $NO_2$, and $NO_x$ emitted during gas appliance use as identified in various studies and reports, and a summary of associated key findings and health outcomes, cont.

| Pollutant | Source | Description of Study | Key Findings Related to Pollutant | Example of Health Impacts of Exposure to Pollutant |
|---|---|---|---|---|
| Ultrafine Particles (UFP)/Particle Number (PN) | Dennekamp, M. et al. 2001[28] | Measured UFP and nitrogen oxide emissions from gas and electric stoves and ovens in a laboratory chamber with no ventilation. | Gas combustion alone and with boiling water produced UFP in a peak size range of 15-40 nm. Electric stove coils also generate UFP. The authors suggested that cooking in kitchens with inadequate ventilation could produce toxic particle number concentrations. | Respiratory impacts, cardiovascular disease, various impacts from oxidative stress, neurological impacts.[199,316–318] |
| | Minutolo, P. et al. 2008[319] | Measured UFP emissions from 3 heater burners and 1 stove burner in an experimental chamber. | UFP in the size range of 1 nm-10 nm formed under all examined conditions, but at very low mass concentrations. Larger UFP (soot particles) are not formed under the conditions studied. A larger amount of particles were ultimately emitted from the stove top burner than heater burners. | Respiratory impacts, cardiovascular disease, various impacts from oxidative stress, neurological impacts.[199,316–318] |
| | Ruiz, P. A. et al. 2010[66] | Conducted indoor and outdoor sampling of 16 homes with unvented space heaters using different energy sources (electric/central heating, compressed natural gas, liquified petroleum gas, and kerosene) in Chile. | Found higher levels of UFP in homes with combustion heaters (including gas heaters) than in homes with electric heaters or central heating. | Respiratory impacts, cardiovascular disease, various impacts from oxidative stress, neurological impacts.[199,316–318] |
| | Wallace, L., Wang, F., Howard-Reed, C. & Persily, A. 2008[320] | Measured UFP emissions of a gas stove, electric stove, and electric toaster oven in a test house. 150 tests were conducted. | Found larger particle number concentrations than reported in previous studies assessing larger particles >10 nm, with the highest concentrations occurring at a 5 nm particle size. The study concludes that gas and electric stoves produce these small particles in significant quantities. | Respiratory impacts, cardiovascular disease, various impacts from oxidative stress, neurological impacts.[199,316–318] |
| | Zhang, Q., Gangupomu, R. H., Ramirez, D. & Zhu, Y. 2010[57] | Measured UFP, $PM_{2.5}$, and black carbon concentrations from cooking in residences. | Cooking increased UFP concentrations in the kitchen significantly. This study found that the highest UFP concentrations occurred when gas stoves were turned on high and range hoods were not on. | Respiratory impacts, cardiovascular disease, various impacts from oxidative stress, neurological impacts.[199,316–318] |
| Other Volatile Organic Compounds (VOCs) | Stocco, C. et al. 2008[323] | Measured personal, indoor and outdoor 24-hour levels of 188 VOCs (though analysis focused on 18) in 48 homes for 8 weeks during winter and summer in Canada. Created an exposure model using predictions based on indoor concentrations. | Indoor concentrations of VOCs are predictive of personal exposures. Having a gas stove in the home was a significant predictor of acrolein exposure. | Varies by VOC. Examples are headaches, fatigue, respiratory issues.[321,322] |

**Table B-2:** Emissions of $PM_{2.5}$ and $NO_X$ from residential gas appliance use and estimated primary, nitrate, and total $PM_{2.5}$ reductions under an electrification scenario in which all residential gas appliances are replaced with electric appliances.

| County | $PM_{2.5}$ Emissions from Gas Appliances (tons/year)[73] | Primary $PM_{2.5}$ Reduction (μg/m³) | $NO_x$ Emissions from Gas Appliances (tons/year) | Nitrate $PM_{2.5}$ Reduction (μg/m³) | Total $PM_{2.5}$ Reduction (μg/m³) |
|---|---|---|---|---|---|
| Alameda | 101 | 0.47 | 793 | 0.021 | 0.49 |
| Alpine | 0 | 0 | 0 | 0 | 0 |
| Amador | 0.77 | 0.0072 | 7.5 | 0.00081 | 0.0081 |
| Butte | 12 | 0.029 | 91 | 0.0031 | 0.032 |
| Calaveras | 0.22 | 0.0017 | 1.4 | 0.00023 | 0.0020 |
| Colusa | 0.80 | 0.0036 | 7.5 | 0.00051 | 0.0042 |
| Contra Costa | 77 | 0.29 | 634 | 0.018 | 0.30 |
| Del Norte | 0 | 0 | 0 | 0 | 0 |
| El Dorado | 1.2 | 0.0027 | 84 | 0.0025 | 0.0052 |
| Fresno | 44 | 0.045 | 368 | 0.018 | 0.062 |
| Glenn | 1.0 | 0.013 | 8.2 | 0.00070 | 0.014 |
| Humboldt | 7.0 | 0.0091 | 71 | 0.00071 | 0.0098 |
| Imperial | 1.9 | 0.0015 | 0 | 0 | 0.0015 |
| Inyo | 0 | 0 | 0 | 0 | 0 |
| Kern | 39 | 0.030 | 343 | 0.0133 | 0.043 |
| Kings | 6.7 | 0.067 | 53 | 0.0114 | 0.078 |
| Lake | 0 | 0 | 0 | 0 | 0 |
| Lassen | 0 | 0 | 3.9 | 0.00018 | 0.00018 |
| Los Angeles | 368 | 0.22 | 3883 | 0.041 | 0.26 |
| Madera | 3.1 | 0.018 | 29 | 0.0055 | 0.024 |
| Marin | 23 | 0.41 | 192 | 0.0086 | 0.41 |
| Mariposa | 0 | 0 | 0 | 0 | 0 |
| Mendocino | 2.7 | 0.0036 | 21 | 0.00027 | 0.0038 |
| Merced | 10 | 0.072 | 90 | 0.012 | 0.085 |
| Modoc | 0 | 0 | 0 | 0 | 0 |
| Mono | 0 | 0 | 0 | 0 | 0 |
| Monterey | 22 | 0.0086 | 181 | 0.0017 | 0.010 |
| Napa | 9.2 | 0.19 | 75 | 0.019 | 0.20 |
| Nevada | 2.1 | 0.013 | 48 | 0.0037 | 0.017 |
| Orange | 120 | 0.33 | 1178 | 0.041 | 0.37 |
| Placer | 18 | 0.025 | 229 | 0.0057 | 0.031 |
| Plumas | 0 | 0 | 0 | 0 | 0 |
| Riverside | 72 | 0.12 | 960 | 0.043 | 0.16 |

**Table B-2:** Emissions of $PM_{2.5}$ and $NO_X$ from residential gas appliance use and estimated primary, nitrate, and total $PM_{2.5}$ reductions under an electrification scenario in which all residential gas appliances are replaced with electric appliances, cont.

| County | $PM_{2.5}$ Emissions from Gas Appliances (tons/year)[73] | Primary $PM_{2.5}$ Reduction (µg/m³) | $NO_x$ Emissions from Gas Appliances (tons/year) | Nitrate $PM_{2.5}$ Reduction (µg/m³) | Total $PM_{2.5}$ Reduction (µg/m³) |
|---|---|---|---|---|---|
| Sacramento | 73 | 0.21 | 716 | 0.028 | 0.24 |
| San Benito | 2.2 | 0.032 | 23 | 0.0027 | 0.035 |
| San Bernardino | 80 | 0.062 | 857 | 0.0039 | 0.066 |
| San Diego | 62 | 0.057 | 942 | 0.016 | 0.073 |
| San Francisco | 65 | 0.84 | 512 | 0.015 | 0.85 |
| San Joaquin | 35 | 0.22 | 316 | 0.026 | 0.24 |
| San Luis Obispo | 14 | 0.051 | 132 | 0.0025 | 0.053 |
| San Mateo | 56 | 0.53 | 446 | 0.015 | 0.55 |
| Santa Barbara | 25 | 0.028 | 194 | 0.0014 | 0.029 |
| Santa Clara | 112 | 0.37 | 851 | 0.040 | 0.41 |
| Santa Cruz | 14 | 0.062 | 122 | 0.0031 | 0.065 |
| Shasta | 6.6 | 0.0064 | 61 | 0.00057 | 0.0070 |
| Sierra | 0 | 0 | 0 | 0 | 0 |
| Siskiyou | 0 | 0 | 0 | 0 | 0 |
| Solano | 27 | 0.22 | 205 | 0.0098 | 0.23 |
| Sonoma | 30 | 0.15 | 252 | 0.011 | 0.16 |
| Stanislaus | 26 | 0.16 | 220 | 0.022 | 0.19 |
| Sutter | 5.4 | 0.051 | 43 | 0.0031 | 0.054 |
| Tehama | 1.3 | 0.0023 | 13 | 0.0011 | 0.0034 |
| Trinity | 0 | 0 | 0.080 | 0.0000032 | 0.0000032 |
| Tulare | 21 | 0.043 | 181 | 0.016 | 0.060 |
| Tuolumne | 0 | 0 | 0 | 0 | 0 |
| Ventura | 41 | 0.12 | 344 | 0.0050 | 0.13 |
| Yolo | 10 | 0.064 | 91 | 0.0053 | 0.070 |
| Yuba | 2.6 | 0.044 | 26 | 0.0042 | 0.048 |

**Table B-3:** Estimated mortality and morbidity reductions from the electrification scenario by air basin, in 2018.

| Air Basin | Nitrate $PM_{2.5}$: Reduced Mortality | All $PM_{2.5}$: Reduced Mortality | Acute Bronchitis (Cases Avoided) | Chronic Bronchitis (Cases Avoided) |
|---|---|---|---|---|
| Great Basin Valleys | 0 | 0 | 0 | 0 |
| Lake County | 0 | 0 | 0 | 0 |
| Lake Tahoe | 0.061 | 0.29 | 0.35 | 0.19 |
| Mojave Desert | 9.7 | 61 | 109 | 52 |
| Mountain Counties | 0.088 | 0.43 | 0.45 | 0.27 |
| North Central Coast | 0.063 | 0.91 | 1.5 | 0.76 |
| North Coast | 0.14 | 2.2 | 2.5 | 1.5 |
| Northeast Plateau | 0.0021 | 0.0024 | 0.0031 | 0.0017 |
| Sacramento Valley | 2.1 | 20 | 31 | 14 |
| Salton Sea | 1.4 | 5.4 | 9.9 | 4.1 |
| San Diego | 2.1 | 9.5 | 15 | 7.8 |
| San Francisco Bay Area | 6.0 | 125 | 196 | 115 |
| San Joaquin Valley | 2.8 | 18 | 35 | 13 |
| South Central Coast | 0.24 | 5.7 | 9.3 | 4.4 |
| South Coast | 14 | 105 | 185 | 91 |
| **Total** | **39** | **354** | **596** | **304** |

**Table B-4:** Estimated monetization of the health benefits from the electrification scenario by air basin, in 2018.

| Air Basin | Nitrate $PM_{2.5}$: Mortality Valuation | All $PM_{2.5}$: Mortality Valuation | Acute Bronchitis Valuation | Chronic Bronchitis Valuation |
|---|---|---|---|---|
| Great Basin Valleys | $0 | $0 | $0 | $0 |
| Lake County | $0 | $0 | $0 | $0 |
| Lake Tahoe | $565,809 | $2,707,573 | $185 | $95,561 |
| Mojave Desert | $90,280,523 | $570,844,734 | $57,048 | $26,344,390 |
| Mountain Counties | $818,583 | $3,972,574 | $239 | $134,691 |
| North Central Coast | $592,105 | $8,528,831 | $808 | $380,196 |
| North Coast | $1,332,509 | $20,137,675 | $1,331 | $729,931 |
| Northeast Plateau | $20,055 | $22,050 | $2 | $831 |
| Sacramento Valley | $19,919,241 | $187,863,502 | $16,487 | $7,182,284 |
| Salton Sea | $13,247,571 | $50,566,952 | $5,187 | $2,069,330 |
| San Diego | $19,447,260 | $88,384,743 | $8,138 | $3,932,141 |
| San Francisco Bay Area | $56,225,767 | $1,168,481,307 | $103,155 | $57,664,334 |
| San Joaquin Valley | $26,072,110 | $168,161,417 | $18,422 | $6,390,438 |
| South Central Coast | $2,226,449 | $53,269,838 | $4,888 | $2,206,735 |
| South Coast | $134,721,840 | $983,223,898 | $97,343 | $45,569,735 |
| **Total** | **$365 million** | **$3.31 billion** | **$0.31 million** | **$153 million** |

**Table B-5:** BenMAP outputs for estimated mortality reductions from the electrification scenario by county, in 2018.

| County | All $PM_{2.5}$: Reduced Mortality (95% Confidence Interval) |
|---|---|
| Alameda | 29.4 (19.85, 38.93) |
| Alpine | 0 (0, 0) |
| Amador | 0.02 (0.02, 0.03) |
| Butte | 0.43 (0.29, 0.57) |
| Calaveras | 0.01 (0, 0.01) |
| Colusa | 0.003 (0.002, 0.004) |
| Contra Costa | 14.58 (9.84, 19.29) |
| Del Norte | 0 (0, 0) |
| El Dorado | 0.05 (0.03, 0.07) |
| Fresno | 2.5 (1.69, 3.31) |
| Glenn | 0.02 (0.01, 0.03) |
| Humboldt | 0.08 (0.05, 0.1) |
| Imperial | 0.02 (0.02, 0.03) |
| Inyo | 0 (0, 0) |
| Kern | 1.5 (1.02, 1.99) |
| Kings | 0.38 (0.26, 0.51) |
| Lake | 0 (0, 0) |
| Lassen | 0 (0, 0) |
| Los Angeles | 96.88 (65.42, 128.22) |
| Madera | 0.16 (0.11, 0.21) |
| Marin | 5.17 (3.49, 6.85) |
| Mariposa | 0 (0, 0) |
| Mendocino | 0.02 (0.01, 0.03) |
| Merced | 0.85 (0.58, 1.13) |
| Modoc | 0 (0, 0) |
| Mono | 0 (0, 0) |
| Monterey | 0.15 (0.1, 0.2) |
| Napa | 1.55 (1.04, 2.05) |
| Nevada | 0.11 (0.07, 0.14) |
| Orange | 44.89 (30.31, 59.42) |
| Placer | 0.79 (0.53, 1.05) |
| Plumas | 0 (0, 0) |
| Riverside | 16.16 (10.91, 21.39) |
| Sacramento | 16.04 (10.83, 21.23) |
| San Benito | 0.07 (0.05, 0.09) |
| San Bernardino | 13.03 (8.8, 17.25) |
| San Diego | 9.46 (6.39, 12.52) |
| San Francisco | 28.64 (19.33, 37.93) |
| San Joaquin | 7.79 (5.26, 10.31) |
| San Luis Obispo | 0.79 (0.53, 1.05) |
| San Mateo | 16.03 (10.82, 21.23) |
| Santa Barbara | 0.54 (0.36, 0.71) |
| Santa Clara | 25.35 (17.12, 33.56) |
| Santa Cruz | 0.69 (0.47, 0.91) |
| Shasta | 0.1 (0.06, 0.13) |
| Sierra | 0 (0, 0) |
| Siskiyou | 0 (0, 0) |
| Solano | 4.54 (3.07, 6.01) |
| Sonoma | 4.11 (2.78, 5.44) |
| Stanislaus | 4.51 (3.05, 5.97) |
| Sutter | 0.24 (0.16, 0.32) |
| Tehama | 0.01 (0.01, 0.02) |
| Trinity | 0 (0, 0) |
| Tulare | 1.05 (0.71, 1.39) |
| Tuolumne | 0 (0, 0) |
| Ventura | 4.37 (2.95, 5.78) |
| Yolo | 0.56 (0.38, 0.74) |
| Yuba | 0.16 (0.11, 0.21) |

**Table B-6:** BenMAP outputs for estimated monetization of mortality reductions from the electrification scenario by county, in 2018 (shown in 2015 dollars, pre-inflation adjustments).

| County | All $PM_{2.5}$: Mortality Valuation & 95% Confidence Interval | | |
|---|---|---|---|
| | Estimate | Lower Bound | Upper Bound |
| Alameda | $255,972,880 | $23,741,228 | $694,285,440 |
| Alpine | $0 | $0 | $0 |
| Amador | $202,040 | $18,743 | $547,983 |
| Butte | $3,774,240 | $350,120 | $10,236,708 |
| Calaveras | $54,529 | $5,058 | $147,895 |
| Colusa | $28,817 | $2,673 | $78,159 |
| Contra Costa | $126,885,312 | $11,769,362 | $344,151,744 |
| Del Norte | $0 | $0 | $0 |
| El Dorado | $448,354 | $41,592 | $1,216,049 |
| Fresno | $21,789,786 | $2,021,323 | $59,099,608 |
| Glenn | $174,319 | $16,171 | $472,799 |
| Humboldt | $687,856 | $63,810 | $1,865,639 |
| Imperial | $217,224 | $20,151 | $589,166 |
| Inyo | $0 | $0 | $0 |
| Kern | $13,099,088 | $1,215,142 | $35,528,120 |
| Kings | $3,329,755 | $308,882 | $9,031,176 |
| Lake | $0 | $0 | $0 |
| Lassen | $20,543 | $1,906 | $55,719 |
| Los Angeles | $843,326,528 | $78,224,896 | $2,287,352,832 |
| Madera | $1,362,471 | $126,391 | $3,695,370 |
| Marin | $45,046,368 | $4,178,137 | $122,180,416 |
| Mariposa | $0 | $0 | $0 |
| Mendocino | $180,162 | $16,713 | $488,643 |
| Merced | $7,440,247 | $690,186 | $20,179,928 |
| Modoc | $0 | $0 | $0 |
| Mono | $0 | $0 | $0 |
| Monterey | $1,315,226 | $122,009 | $3,567,224 |
| Napa | $13,453,159 | $1,247,909 | $36,488,836 |
| Nevada | $921,981 | $85,529 | $2,500,646 |
| Orange | $390,746,336 | $36,243,116 | $1,059,827,968 |
| Placer | $6,895,075 | $639,627 | $18,701,210 |
| Plumas | $0 | $0 | $0 |
| Riverside | $140,681,808 | $13,049,800 | $381,568,448 |
| Sacramento | $139,650,096 | $12,953,712 | $378,772,032 |
| San Benito | $613,641 | $56,925 | $1,664,352 |
| San Bernardino | $113,451,872 | $10,524,312 | $307,711,232 |
| San Diego | $82,344,448 | $7,638,627 | $223,339,872 |
| San Francisco | $249,324,048 | $23,121,320 | $676,267,520 |
| San Joaquin | $67,795,992 | $6,288,626 | $183,882,720 |
| San Luis Obispo | $6,890,559 | $639,203 | $18,688,990 |
| San Mateo | $139,580,336 | $12,945,681 | $378,590,528 |
| Santa Barbara | $4,701,954 | $436,181 | $12,752,904 |
| Santa Clara | $220,699,600 | $20,470,320 | $598,609,024 |
| Santa Cruz | $6,017,096 | $558,174 | $16,319,943 |
| Shasta | $835,539 | $77,510 | $2,266,194 |
| Sierra | $0 | $0 | $0 |
| Siskiyou | $0 | $0 | $0 |
| Solano | $39,542,172 | $3,667,874 | $107,249,936 |
| Sonoma | $35,786,860 | $3,319,630 | $97,063,968 |
| Stanislaus | $39,271,068 | $3,642,793 | $106,514,288 |
| Sutter | $2,117,348 | $196,416 | $5,742,799 |
| Tehama | $110,212 | $10,224 | $298,922 |
| Trinity | $0 | $0 | $0 |
| Tulare | $9,130,249 | $846,966 | $24,763,624 |
| Tuolumne | $0 | $0 | $0 |
| Ventura | $38,036,816 | $3,528,387 | $103,166,240 |
| Yolo | $4,857,377 | $450,592 | $13,174,484 |
| Yuba | $1,407,345 | $130,553 | $3,817,086 |

**Table B-7:** List of air quality standards

| Air Pollutant | Standard | Averaging Time | Concentration |
|---|---|---|---|
| CO | CAAQS | 1-hour | 23,000 µg/m³ (20 ppm) |
| | | 8-hour | 10,000 µg/m³ (9 ppm) |
| | NAAQS | 1-hour | 40,000 µg/m³ (35 ppm) |
| | | 8-hour | 10,000 µg/m³ (9 ppm) |
| | Health Canada Residential Indoor Air Quality Guideline | 1-hour | 28,600 µg/m³ (25 ppm) |
| | Health Canada Residential Indoor Air Quality Guideline | 24-hour | 11,500 µg/m³ (10 ppm) |
| $NO_2$ | CAAQS | 1-hour | 339 µg/m³ (180 ppb) |
| | | Annual mean | 57 µg/m³ (30 ppb) |
| | NAAQS | 1-hour | 188 µg/m³ (100 ppb) |
| | | Annual mean | 100 µg/m³ (53 ppb) |
| | Health Canada Residential Indoor Air Quality Guideline | 1-hour | 170 µg/m³ (90 ppb) |
| | Health Canada Residential Indoor Air Quality Guideline | 24-hour | 20 µg/m³ (11 ppb) |

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Sierra Club National
2101 Webster Street, Suite 1300
Oakland, CA 94612
(415) 977-5500

Sierra Club Legislative
50 F Street, NW, Eighth Floor
Washington, DC 20001
(202) 547-1141

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