EMISIONES DE DIÉSEL Y DE TRÁFICO

Impactos en la salud de los niños

Contaminantes de los Gases de Escape de Vehículos de Motor

- **Materia Particulada - Fina (PM2.5) y Ultrafina (UFP)**
- **Sustancias cancerígenas: HAP, Carbono Negro, Benceno, Formaldehído**
- **Óxidos de Nitrógeno**
- **Metales**
- **Monóxido de Carbono**
- **Dióxido de Carbono**

Los Niños Corren un Mayor Riesgo que los Adultos.

- **Sistemas inmunológico, neurológico y pulmonar en rápido desarrollo.**
- **Los niños respiran más aire que los adultos en relación con su tamaño.**
- **Es más probable que estén activos cuando están al aire libre.**

Efectos de Salud en Niños que Viven Cerca de Zonas con Contaminación Relacionada con el Tráfico.

Alta Evidencia de:

- **Prevalencia del asma**
- **Nueva aparición del asma**
- **Síntomas respiratorios crónicos y agudos relacionados con el asma.**
- **Desarrollo tardío de la función pulmonar**
- **Prevalencia de la leucemia infantil**

Evidencia Emergente de:

- **Trastorno del espectro autista**
- **Resultados adversos del parto**
- **Trastorno por déficit de atención con hiperactividad (ADHD)**
- **Retraso en el desarrollo cognitivo**
- **Obesidad infantil, diabetes tipo II y resistencia a la insulina.**

Exposición Prenatal y en la Infancia

- **Cáncer en Adultos**
- **Enfermedad de Parkinson**
- **Enfermedad de Alzheimer**

Fuente de información: Boothe, V. L., Boothe, R. W., & Baldauf, R. (2020).*Transport and children's wellbeing*. Capítulo siete: Impacto de las emisiones del tráfico en la salud y el bienestar infantil. (pp. 119–142). ScienceDirect.

3 Razones para Reportar Camiones Diésel en Ralentí

1 Proteger la Salud de los Niños

La exposición a los gases de escape del diésel daña los pulmones en desarrollo de los niños, aumentando el riesgo de asma, infecciones respiratorias e incluso cáncer.

2 Proteger la Salud de la Comunidad

Los gases de escape del diésel están clasificados como cancerígenos para los seres humanos (IARC 2012). El 70% del riesgo de cáncer conocido en California, derivado de los contaminantes tóxicos del aire, está relacionado con las emisiones de diésel.

3 Reducir la Contaminación

En California, los camiones y autobuses diésel no pueden permanecer en ralentí durante más de 5 minutos, especialmente cerca de escuelas, viviendas y hospitales.

1-800-Cut-Smog

Su informe puede ser anónimo.

California Climate Investments

California Safe Schools Coalition

CALIFORNIA
AIR RESOURCES BOARD

Gavin Newsom, Governor
Yana Garcia, CalEPA Secretary
Liane M. Randolph, Chair

California, ¡respira tranquilo! Apaga tu motor.

¿Qué es el motor en marcha?

Tener el motor en marcha se refiere a tener el motor encendido de un vehículo sin moverse. Tener el motor de camiones de diésel, autobuses y autos de pasajeros en marcha sin alguna necesidad cerca de escuelas y centros comerciales causa preocupación entre muchas comunidades de California.

El Consejo de Recursos del Aire de California (CARB por sus siglas en inglés) hace cumplir las leyes de aire limpio que incluyen motores en marcha de camiones y autobuses de diésel en todo el estado. Aunque las reglas de motor en marcha de CARB no aplican a automóviles de pasajeros, es importante reducir dicha práctica para ayudar a proteger la salud de la comunidad y el medio ambiente.

¿Por qué reducir el motor en marcha? Al apagar los motores, podemos…

Salud comunitaria

Proteger la salud de nuestros miembros de la comunidad que están en mayor riesgo, como niños, ancianos y los que tienen condiciones de salud como asma y problemas cardíacos. Estudios recientes muestran que la mortalidad relacionada a la contaminación del aire es tres veces mayor que la de los accidentes de autos anualmente.

Medio ambiente

Mejora el medio ambiente. Eliminar tener el motor en marcha de forma innecesaria en EE.UU. equivale a eliminar la contaminación de 5 millones de autos, o plantar 30 millones de acres de árboles, cada año.

Ahorros

Ahorre dinero reduciendo el desgaste y deterioro innecesario (y costoso). Investigadores calculan que tener el motor en marcha en camiones, autobuses y autos en los EE.UU. desperdicia alrededor de 6 mil millones de galones de combustible anualmente.

Cumplimiento de leyes de motores en marcha en camiones y autobuses de diésel

CARB tiene reglas que limitan tener el motor de los camiones y autobuses en marcha a 5 minutos en todos lados. Con calcomanías certificadas Clean Idle (motor en marcha limpia) pueden tener el motor en marcha por más de 5 minutos en áreas sin restricciones, pero aún no se les permite tener el motor en marcha dentro de los 100 pies de áreas restringidas, como escuelas, hogares, hospitales e instalaciones para personas mayores y cuidado infantil durante más de 5 minutos.

Esta regla no aplica a autos o camionetas tipo pick-up. Existen otras excepciones a estas reglas, como que los camiones y autobuses pueden tener el motor en marcha mientras usan dispositivos de toma de energía y en el tráfico. Los autobuses pueden tener el motor en marcha hasta 10 minutos antes de que los pasajeros aborden y en cualquier momento que los pasajeros estén a bordo para el control del clima. Los autobuses escolares no pueden tener el motor en marcha durante más de 30 segundos cerca de las escuelas.

Muchos de los camiones y autobuses de California ya tienen calcomanías certificadas Clean Idle (motor en marcha limpia), lo que significa que pueden tener el motor en marcha durante más de 5 minutos en áreas sin restricciones. En 2020, el personal de aplicación de CARB descubrió que más del 97% de los camiones y autobuses observados seguían las reglas de motores en marcha en comunidades de todo California.

¿Qué puedo hacer para ayudar?

1. **Reduzca el tiempo de uso del motor en marcha.** Aunque las reglas de motor en marcha de CARB no aplican a autos de pasajeros (coches y camionetas pick-up), el motor en marcha de todos los vehículos causa daños a nuestra salud y al medio ambiente. De ser posible, apaga el motor si va a esperar más de 10 segundos.
2. **Mantenga su auto en buen estado.** Un vehículo en mal estado contamina el aire más que un vehículo en buen estado, y puede también consumir más combustible. Estos vehículos se averían con más frecuencia lo que conlleva a reparaciones costosas.
3. **Reporte camiones y autobuses con motor en marcha en su comunidad** llamando al 1-800-END-SMOG (1-800-363-7664) o yendo a *calepa.ca.gov/enforcement/complaints*. Recuerda, los camiones y autobuses con calcomanías Clean Idle pueden tener el motor en marcha por más de 5 minutos en áreas sin restricciones.

Incluya la siguiente información al reportar un vehículo con motor en marcha:

- ¿En dónde estuvo el motor en marcha?
- ¿Cuándo ocurrió? ¿Todavía ocurre?
- ¿Quién o que fue la causa?
- Fotos, videos y números de placas frontales o traseras, de ser posible.

Contáctenos *COES@arb.ca.gov* | *bitly.com/CARB-COES*

PRE ENCUESTA SOBRE EL RALENTÍ DIÉSEL (Parte A)

California Safe Schools – Proyecto de Prevención del Ralentí Diésel

Fecha: _____

Instrucciones: Para las preguntas 1–9, marque su respuesta con una X:

1 = Totalmente en desacuerdo | 2 = En desacuerdo | 3 = Neutral | 4 = De acuerdo | 5 = Totalmente de acuerdo

ESCALA 1: Conocimiento (Salud + Exposición)

1. Entiendo cómo las emisiones de escape diésel (PM2.5, NOx) afectan la salud respiratoria y cardiovascular.

1 ☐ 2 ☐ 3 ☐ 4 ☐ 5 ☐

2. Puedo identificar fuentes comunes de contaminación diésel en mi comunidad (por ejemplo, camiones, rutas de carga y vehículos en ralentí cerca de las escuelas).

1 ☐ 2 ☐ 3 ☐ 4 ☐ 5 ☐

3. Entiendo que los niños y las comunidades cercanas a corredores de carga o carreteras principales pueden experimentar una mayor exposición a la contaminación diésel.

1 ☐ 2 ☐ 3 ☐ 4 ☐ 5 ☐

ESCALA 2: Conciencia y Percepción (Medio Ambiente + Sistemas)

4. Sé que algunos vecindarios experimentan mayores cargas de contaminación del aire que otros en el Condado de Los Ángeles.

1 ☐ 2 ☐ 3 ☐ 4 ☐ 5 ☐

5. Puedo identificar al menos un lugar en mi comunidad donde el ralentí de vehículos diésel o el tráfico de camiones contribuye a la contaminación del aire.

1 ☐ 2 ☐ 3 ☐ 4 ☐ 5 ☐

6. Creo que las herramientas visibles, como letreros de “No Dejar el Motor en Ralentí”, pueden influir en el comportamiento de los conductores.
1 ☐ 2 ☐ 3 ☐ 4 ☐ 5 ☐

ESCALA 3: Intención de Comportamiento y Participación

7. Ahora es más probable que evite dejar innecesariamente un vehículo con el motor en ralentí (por ejemplo, al esperar en vehículos estacionados).
1 ☐ 2 ☐ 3 ☐ 4 ☐ 5 ☐

8. Me sentiría cómodo/a reportando infracciones reiteradas relacionadas con vehículos diésel en ralentí si las observara cerca de escuelas o espacios comunitarios.
1 ☐ 2 ☐ 3 ☐ 4 ☐ 5 ☐

9. Es probable que comparta información sobre la contaminación diésel o las prácticas de “No Dejar el Motor en Ralentí” con otras personas de mi hogar o comunidad.
1 ☐ 2 ☐ 3 ☐ 4 ☐ 5 ☐

Gracias por su participación!

ENCUESTA POSTERIOR SOBRE EL RALENTÍ DIÉSEL (Parte B)

California Safe Schools – Proyecto de Prevención del Ralentí Diésel

Fecha: _____

Instrucciones: Para las preguntas 1–9, marque su respuesta con una X:
1 = Totalmente en desacuerdo | 2 = En desacuerdo | 3 = Neutral | 4 = De acuerdo | 5 = Totalmente de acuerdo

ESCALA 1: Conocimiento (Salud + Exposición)

1. Entiendo cómo las emisiones de escape diésel (PM2.5, NOx) afectan la salud respiratoria y cardiovascular.
1 ☐ 2 ☐ 3 ☐ 4 ☐ 5 ☐

2. Puedo identificar fuentes comunes de contaminación diésel en mi comunidad (por ejemplo, camiones, rutas de carga y vehículos en ralentí cerca de las escuelas).
1 ☐ 2 ☐ 3 ☐ 4 ☐ 5 ☐

3. Entiendo que los niños y las comunidades cercanas a corredores de carga o carreteras principales pueden experimentar una mayor exposición a la contaminación diésel.
1 ☐ 2 ☐ 3 ☐ 4 ☐ 5 ☐

ESCALA 2: Conciencia y Percepción (Medio Ambiente + Sistemas)

4. Sé que algunos vecindarios experimentan mayores cargas de contaminación del aire que otros en el Condado de Los Ángeles.
1 ☐ 2 ☐ 3 ☐ 4 ☐ 5 ☐

5. Puedo identificar al menos un lugar en mi comunidad donde el ralentí de vehículos diésel o el tráfico de camiones contribuye a la contaminación del aire.
1 ☐ 2 ☐ 3 ☐ 4 ☐ 5 ☐

6. Creo que las herramientas visibles, como los letreros de “No Dejar el Motor en Ralentí”, pueden influir en el comportamiento de los conductores.
1 ☐ 2 ☐ 3 ☐ 4 ☐ 5 ☐

ESCALA 3: Intención de Comportamiento y Participación

7. Ahora es más probable que evite dejar innecesariamente un vehículo con el motor en ralentí (por ejemplo, al esperar en vehículos estacionados).
1 ☐ 2 ☐ 3 ☐ 4 ☐ 5 ☐

8. Me sentiría cómodo/a reportando infracciones reiteradas relacionadas con vehículos diésel en ralentí si las observara cerca de escuelas o espacios comunitarios.
1 ☐ 2 ☐ 3 ☐ 4 ☐ 5 ☐

9. Es probable que comparta información sobre la contaminación diésel o las prácticas de “No Dejar el Motor en Ralentí” con otras personas de mi hogar o comunidad.
1 ☐ 2 ☐ 3 ☐ 4 ☐ 5 ☐

10. Con cuántas personas compartió estos materiales? Qué es principalmente lo que escuchó en comentarios recibidos?
(por ejemplo: “15 compañeros de clase; a la mayoría le sorprendieron los vínculos con el asma”).

_____

_____

_____

11. Qué es lo más importante que aprendió o que cambió su forma de pensar sobre el ralentí diésel o la calidad del aire en su comunidad?

_____

_____

_____

12. Qué más le gustaría saber acerca de este proyecto, o cómo sugeriría que se podría mejorar?

_____

_____

_____

Gracias por su participación!

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Ambient air pollution and Children's health: An umbrella review

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ABSTRACT

The amount of data that is currently accessible on ambient air pollution and children's health has significantly increased. Although numerous systematic reviews and meta-analyses have evaluated the effect of air pollution on children's health, the results are mixed, including positive, negative, or null associations. The objective of this umbrella review is to thoroughly evaluate the validity of existing research on the connection between air pollution and children's health. We identified 33 systematic reviews of observational studies in which a diversity of health outcomes for children were reviewed, including mortality, respiratory diseases, hypertension, etc. According to our review's findings, being exposed to air pollution increases the chance of developing leukemia among children, as well as autism spectrum disorders, hypertension, otitis media, obesity, pneumonia, asthma, respiratory diseases, eczema, and allergic rhinitis. Additionally, there was a sizable positive association between air pollution and child mortality. The findings of this extensive umbrella review provide convincing proof that exposing children to air pollution increases the risk of developing multiple diseases. Our results have implications for strengthening childhood health care, reducing air pollution, and improving intergenerational equity.

1. Introduction

According to statistics, almost all people (99%) were exposed to concentrations higher than the WHO recommendation (5 μg/m^3 annual mean) (WHO, 2022), and 93% of children under 15 years of age breathe polluted air (WHO, 2018). 4.2 million premature deaths were brought on by ambient air pollution in 2016, with nearly 300,000 of those fatalities occurring in children under the age of five (Adair-Rohani, 2018). Due to their developing bodies and underdeveloped immunological and respiratory systems, children are especially vulnerable to the impact of air pollution (Lin et al., 2021a,b). A serious public threat to the health and welfare of people is air pollution. It has become the fourth leading risk factor for death after high blood pressure, tobacco use and poor diet. In the past, 40% of chronic obstructive pulmonary disease (COPD) and 192 million years of life lost (YLL) were associated with air pollution (IHME, 2023).

According to the studies we included, both developing and developed regions or countries are suffering from air pollution (Karimi and Shokrinezhad, 2020). The effects of toxic substances can begin before birth and continue into later life, in which the structural and functional development of certain organs is affected (Miller and Marty, 2010). Previous research has revealed that air pollution exposure can have an impact on the blood system (Gong et al., 2019), mortality (Karimi and Shokrinezhad, 2020), respiratory system (Orellano et al., 2018), nervous system (Liu et al., 2023), and circulatory system (Yan et al., 2021). It is also linked to metabolic disorders (Huang et al., 2022), allergic disease (Yue et al., 2022), and infectious disease (Lee et al., 2020), etc. In China, studies have shown that air pollution may increase the risk of death among children (He et al., 2022). In terms of the respiratory system, exposure to coarse particulate matter in infants (Lu et al., 2023a,b,c,d) and young children (Lu et al., 2022) may increase the risk of asthma, and allergic rhinitis (Lu et al., 2023a,b,c,d) and pneumonia (Lu et al., 2023a, b,c,d) in infants are also closely related to it. In addition, exposure to air pollution has been associated with autism spectrum disorders (Geng et al., 2019), elevated blood pressure (Zhang et al., 2020), allergies (Liu et al., 2020), and otitis media (Lu et al., 2023a,b,c,d) in children.

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Before this study, hundreds of epidemiological studies explored the effects of air pollution on children's health and conducted many systematic reviews and meta-analyses to synthesize these results; however, the conclusions are mixed. In addition, the majority of the previous studies focused only on one aspect of health outcomes. Since air pollution exposure is proposed to be linked to numerous health outcomes, the overall picture of air pollution exposure and health outcomes remains unclear from the available evidence, which may hinder the guide on children's care for clinicians and public health workers.

To provide researchers and public health professionals with more comprehensive and higher-quality evidence relating air pollution exposure to children's health, we conducted this umbrella review, detailing these links and synthesizing the available evidence to help children understand and prepare for air pollution. This work will protect children today and those who will come after them.

2. Methods

During the whole research process, the PRISMA statement (Table S1) (Page et al., 2021), the preferred reporting items for review overview (Table S2) (Gates et al., 2022), and the umbrella review guidelines (Aromataris et al., 2015) were followed. The protocol has been registered on PROSPERO (CRD42022371501).

2.1. Search strategy

PubMed, Scopus, Medline, Embase, the Web of Science, and the Cochrane Library were used as the main search engines for studies published up to October 8, 2022, and confined to studies published in English. During the literature search, we followed the PRISMA standards and the PRIOR (Preferred Reporting Items for Overviews of Reviews) statement. Supplementary materials provide a complete search strategy. (Table S3).

2.2. Eligibility criteria

PECOS(Participants, Exposure, Comparators, Outcomes, and Study design) was used as eligibility criteria. Children as "Participants", air pollution as "Exposure", "Comparator" stands for risk ratio (RR), odds ratio (OR), or hazard ratio (HR), "Outcome" refers to the various health outcomes of children exposed to air pollution, and "Design" means a review with or without meta-analysis (the focus is on reviews for which a meta-analysis has been performed). Exclusion criteria were as follows: narrative reviews, letters, conference abstracts, grey literature, and research protocols; genetic or experimental studies examining animals; or focused on non-general populations, such as occupational groups. The details are shown in Table 1.

Table 1
Eligibility criteria for this comprehensive review of the relationship between ambient air pollution and child health outcomes.

Element	Inclusion criteria	Exclusion criteria
Population	Children	Adult
Exposures	Outdoor air pollution	Indoor air pollution
Outcomes	Adverse health outcome	Adverse health outcomes not caused by exposure to air pollution
Review Design	Reviews of relevant quantitative observational epidemiologic research, including cohort, case-control, and cross-sectional studies, with or without the utilization of meta-analysis (the focus is on reviews for which a meta-analysis has been performed)	Narrative review Grey literature
Language	English	Non-English

2.3. Study selection

All identified records were imported into Endnote X9, and two researchers (KL and HC) worked alone to screen the title, abstract, and full text of the article and remove duplicate records. Disputes were resolved by consensus with the third author (ZY).

2.4. Data extraction

Information was collected separately by two authors (KL and HC), with inconsistencies resolved by a third researcher (ZY). For each eligible article, we extracted information as shown in Table S4. For each meta-analysis, the key information extracted was shown in Table S7.

2.5. Evaluation of the quality of methodology and evidence

After screening studies, the more specific and sensitive AMSTAR 2 was used instead of JBI to analyze study quality (Ferguson et al., 2022). Quality and credibility were also assessed using AMSTAR 2 (Shea et al., 2017). Detailed scoring items for AMSTAR 2 can be found in Table S5. In this review, we used the concept of recommendation grading, assessment, formulation, and evaluation (GRADE) to evaluate the evidence (Brozek et al., 2021). There are five deterioration criteria (limitation of research method, inconsistencies, impreciseness, possibility of publication bias, indirect) and three upgrade factors (large effect, dose effect, and confounding) in the GRADE system (Guyatt et al., 2011).

2.6. Data synthesis

Given associations of interest, they may be assessed in two or more reviews with overlapping results, which could bias findings and estimates. Therefore, we calculated the level of overlap between component studies using a citation matrix with a method name as the corrected covered area (CCA) (Nyadanu et al., 2022). The CCA was calculated using the following formula, where N is the sum of the number of included primary studies in the umbrella review, r again for the number of rows, and c for the number of columns. CCA >15% indicates a very high degree of overlap, 11–15% indicates high, 6–10% indicates moderate, and less than 5% is slight (Pieper et al., 2014).

$$\text{CCA}\ (\%) = \frac{N - r}{rc - r}$$

Systematic reviews without meta-analysis were narratively synthesized. Pooled estimates (e.g., number of primary studies, OR, and I^2 statistics, etc.) for the meta-analysis review were derived from the preliminary measurements provided. The reviews that underwent meta-analysis used forest plots for combined effect estimation.

3. Results

3.1. Literature search

Six databases yielded 15,502 records (Fig. 1). There were 5121 articles excluded due to duplication. After the initial screening, 91 articles proceeded to the full-text review stage. After further reading, 58 articles were eliminated for the following reasons: articles that were not written in English (n = 15); no systematic review or meta-analysis (n = 14); non-child health (n = 10); conference abstract (n = 13); no clear result (n = 6). In summary, 33 articles were identified as eligible for this umbrella review (30 with meta-analyses and 3 without meta-analyses). Three of these articles were on the blood system, two on mortality, two on the nervous system, fifteen on the respiratory system, three on the circulatory system, one on infectious diseases, three on metabolic systems, and four on allergic diseases.

The selected characteristics of the contained reviews were described in Fig. 2. A total of 526 primary studies from 51 countries were all

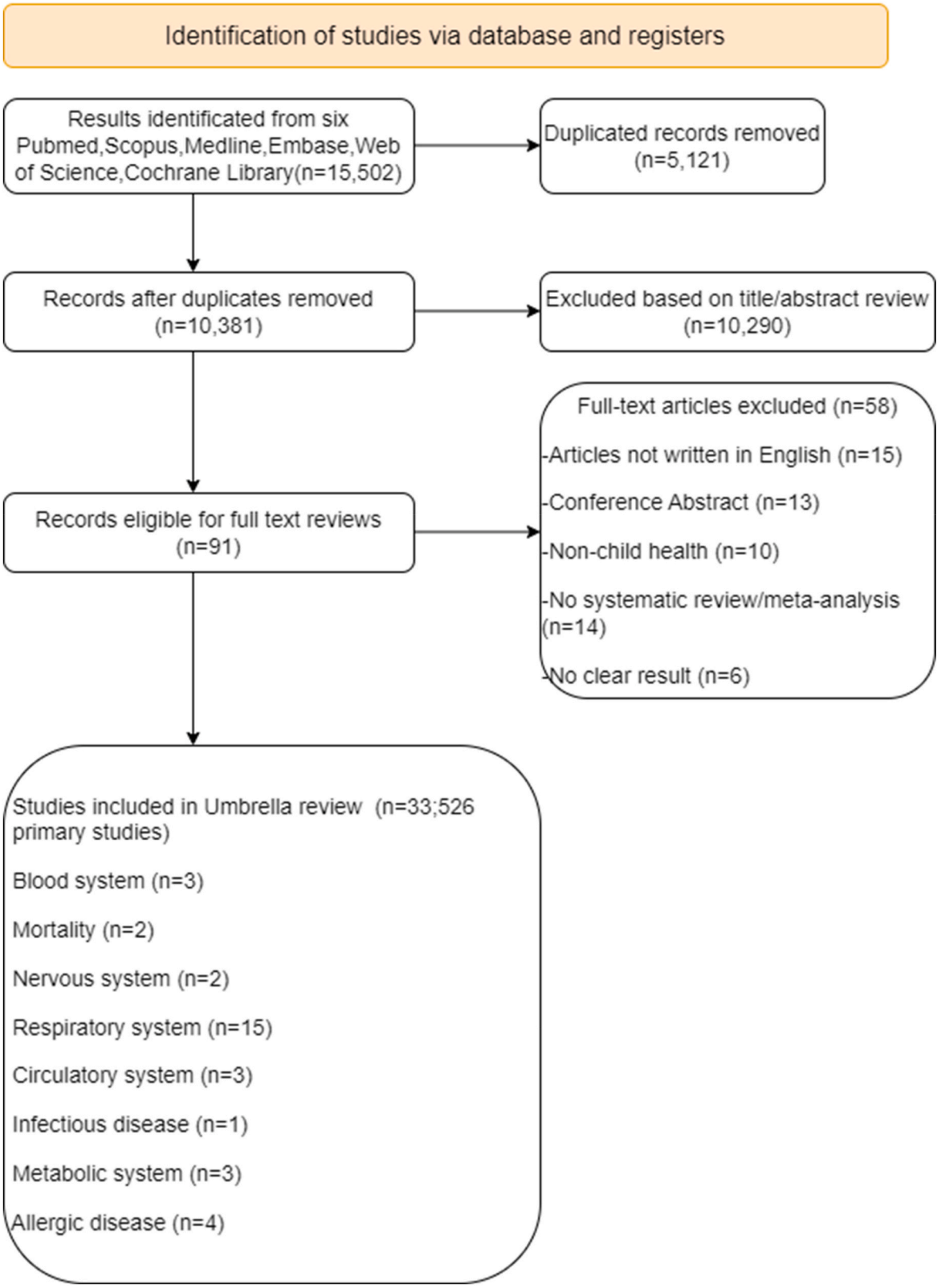

Fig. 1. Flowchart of the systematic search and selection process.

published after 2010, with a focus on the period from 2020 to 2022, including the United States (125), China (107), and Canada (35), among others (Table S8). There were 5–46 primary studies included within every meta-analysis, with an average of 16.

3.2. Risk of bias

The AMSTAR 2 scores of the 33 included articles were as follows: 6 (18.2%) were rated as high quality, 4 (12.1%) as moderate quality, 13 (39.4%) as low quality, and 10 (12.2%) as very low quality. According to

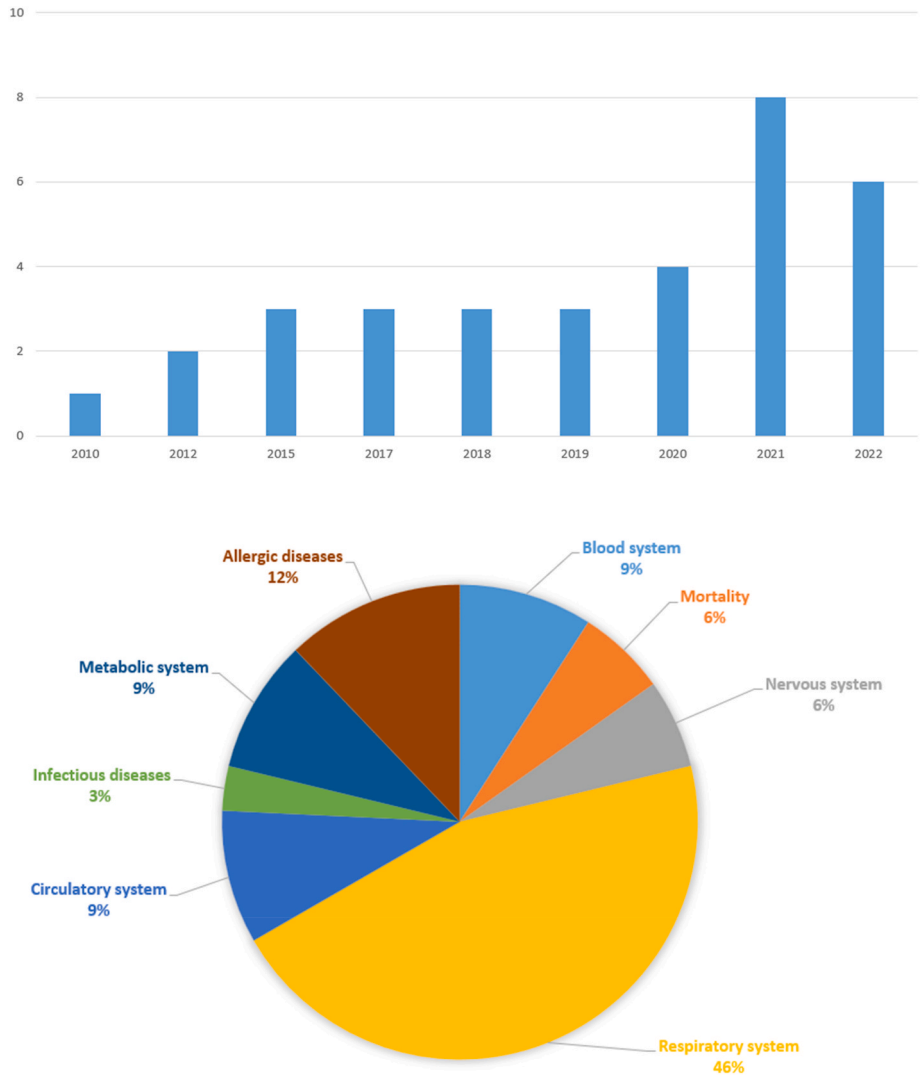

Fig. 2. Essential features of the studies included in this umbrella review. Date of publication of all included studies; Proportion of each health outcome.

GRADE criteria, each estimate was assessed for its quality of evidence. Six studies (18.2%) were evaluated as moderate, ten (30.3%) as low, and seventeen (51.6%) as very low among the 33 studies considered (Table S5).

3.3. Study overlap

The 33 eligible systematic reviews included 771 component studies, including 245 duplicates, of which 526 were unique component studies. The CCA was 1.23%, indicating a slight overlap.

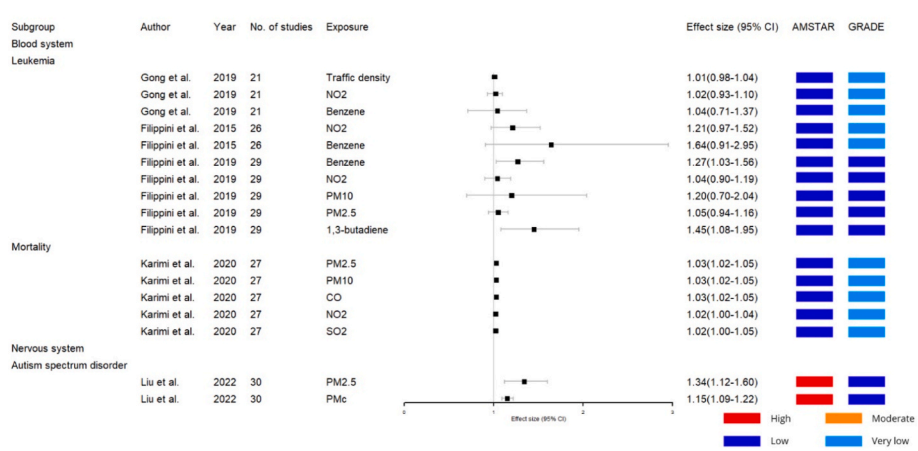

Fig. 3. Association between air pollution and the blood system, mortality, and nervous system.

3.4. Blood system

Three systematic reviews assessed the effect of air pollution on childhood leukemia (Gong et al., 2019; Filippini et al., 2015, 2019). Details of systematic reviews were shown in Table S4. As shown in Fig. 3, traffic density, NO_2, benzene, PM_{10}, $PM_{2.5}$, and 1,3-butadiene were found to increase childhood leukemia risk. Although all three studies found an association between air pollution and leukemia, only benzene (RR = 1.27 (95%CI:1.03, 1.56)) and 1, 3-butadiene (RR = 1.45 (95%CI:1.08, 1.95)) were significantly associated with the increased risk of leukemia (Filippini et al., 2019). There is moderate heterogeneity, as the heterogeneity test showed in most of the studies. Only a few research showed publication bias in their studies. According to the AMSTAR 2, all studies were graded low in terms of quality. On the other hand, when using GRADE, all studies were found to be assessed as low or very low.

3.5. Mortality

Based on the results of a meta-analysis (Karimi and Shokrinezhad, 2020) and a systematic review (Ibrahim et al., 2021), exposure to $PM_{2.5}$ (RR = 1.03 (95%CI:1.02, 1.05)), PM_{10} (RR = 1.03 (95%CI:1.02, 1.05)), CO (RR = 1.03 (95%CI:1.02, 1.05)), NO_2 (RR = 1.02 (95%CI:1.00, 1.04)), and SO_2 (RR = 1.02 (95%CI:1.00, 1.05)) was associated with a higher risk of child mortality (Fig. 3). According to AMSTAR 2, one study was graded as moderate and one study was graded as low. On the other hand, GRADE showed two studies rated as low or very low.

3.6. Nervous system

The relationship between air pollution and child autism spectrum disorder was studied in two meta-analyses (Lin et al., 2021a,b; Liu et al., 2023). Studies with high AMSTAR 2 scores indicate that exposure to $PM_{2.5}$ (RR = 1.34 (95%CI:1.12, 1.60)) and PM_{coarse} (PM_c, PM with an aerodynamic diameter between 2.5 and 10 μm) (RR = 1.15 (95% CI:1.09, 1.22)) was associated with an increased risk of autism spectrum disorder in children (Fig. 3). In addition, one study found that $PM_{2.5}$ exposure–response phenomenon, different concentrations of $PM_{2.5}$ have different effects on autism spectrum disorders. According to AMSTAR 2, one study was graded high and one study was graded low. On the other hand, GRADE showed two studies rated as low or very low.

3.7. Circulatory system

This research included three meta-analyses related to hypertension (Yan et al., 2021; Huang et al., 2021; Qin et al., 2021). Existing evidence shows that long-term exposure to PM_{10} (RR = 1.17 (95%CI:1.13, 1.21)) can raise blood pressure in children and affect the prevalence of hypertension. Three studies had sufficient evidence for long-term exposure, but there was relatively little evidence for short-term exposure. The AMSTAR 2 results showed that one study was rated high and two others were rated low or very low. When GRADE was used to grade the evidence for the included articles, one study was rated moderate, and the other two were rated very low.

3.8. Infectious disease

Included a study on the association of particulate matter with otitis media in children (Lee et al., 2020). Children were more likely to develop otitis media in environments with higher particulate matter concentrations. Compared with the increase of PM_{10} (RR = 1.01 (95% CI:1.01, 1.02)) concentration, the increase of $PM_{2.5}$ (RR = 1.03 (95% CI:1.01, 1.06)) concentration has a greater impact on otitis media (Lee et al., 2020). The level of evidence in this study was low according to AMSTAR 2 and GRADE standards.

3.9. Metabolic system

Three meta-analyses (Parasin et al., 2021; Huang et al., 2022; Bahreynian et al., 2019) showed that air pollution exposure, particularly particulate matter exposure, affects children's weight status and increases the risk of obesity. Among the studies with high AMSTAR 2 and moderate GRADE scores, PM_{10} (RR = 1.12 (95% CI: 1.06, 1.18)), $PM_{2.5}$ (RR = 1.28 (95% CI: 1.13, 1.45)), PM_1 (RR = 1.41 (95% CI: 1.3, 1.53)) and NO_2 (RR = 1.11 (95%CI:1.06, 1.18)) were significantly associated with obesity. The results of the heterogeneity test showed that most of the studies had high inter-study heterogeneity. Two studies were rated high by AMSTAR 2 and moderate by GRADE. The other study had low AMSTAR 2 and GRADE scores.

3.10. Respiratory system

Fifteen systematic reviews have shown that exposure to air pollution has a negative impact on children's respiratory health. (Fig. 4, Fig. 5, Fig. 6). The results of the heterogeneity test showed that most of the studies had high inter-study heterogeneity. Publication bias had only been revealed by a few studies. At least half of the studies in terms of quality were rated low or moderate by AMSTAR 2. On the other hand, GRADE considered the evidence from most of the included studies to be low or very low.

3.10.1. Respiratory diseases

Based on the results of two systematic reviews (King et al., 2018; Rodriguez-Villamizar et al., 2015) and two meta-analyses (Ziou et al., 2022; Zhang et al., 2022), a study with high AMSTAR 2 scores and moderate GRADE scores showed that children and adolescents were exposed to $PM_{2.5}$ (RR = 1.01 (95%CI: 1.00, 1.01)) and PM_{10} (RR = 1.02 (95%CI: 1.01, 1.02)), which may cause upper respiratory tract infection. Boys are more susceptible to short-term exposure to $PM_{2.5}$ than girls (Zhang et al., 2022). In addition, several remaining studies with low or very low AMSTAR 2 and GRADE ratings found an association between air pollutants and bronchiolitis hospitalizations and adverse effects on lung function.

3.10.2. Pneumonia

A meta-analysis found that PM_{10} (RR = 1.02 (95%CI: 1.01, 1.02)), $PM_{2.5}$ (RR = 1.02 (95%CI: 1.01, 1.03)), NO_2 (RR = 1.01 (95%CI: 1.01, 1.03)) 1.00, 1.02)), O_3 (RR = 1.02 (95%CI: 1.01, 1.03)) and SO_2 (RR = 1.03 (95%CI: 1.00, 1.05)) increased the short-term hospitalization for pneumonia in children. Even at low levels, PM2.5 pollution is still associated with pneumonia in children. Both AMSTAR 2 and GRADE scores were very low (Nhung et al., 2017).

3.10.3. Asthma

The results of 10 meta-analyses (Orellano et al., 2018; Orellano et al., 2017; Luong et al., 2019; Takenoue et al., 2012; Gasana et al., 2012; Weinmayr et al., 2010; Zhang et al., 2022; Yan et al., 2020; Bowatte et al., 2015; Han et al., 2021) show that both prenatal maternal exposure and early childhood exposure to air pollution are associated with child asthma incidence and prevalence, as well as asthma development and exacerbation. Studies with high AMSTAR 2 scores and moderate GRADE scores showed that $PM_{2.5}$ (RR = 1.07 (95%CI: 1.00, 1.13)), NO_2 (RR = 1.21 (95%CI: 1.12, 1.31)), benzene (RR = 1.11 (95%CI: 1.06, 1.17) and total volatile organic pollutants (RR = 1.06 (95%CI: 1.03, 1.1)) were significantly associated with the risk of asthma. The effects of $PM_{2.5}$ and NO_2 on asthma in Asia were higher than those in Europe and North America (Han et al., 2021). In addition, the degree of association between $PM_{2.5}$ and asthma increased with age (Bowatte et al., 2015).

3.11. Allergic disease

The effect of air pollution on allergic disease was examined in 4

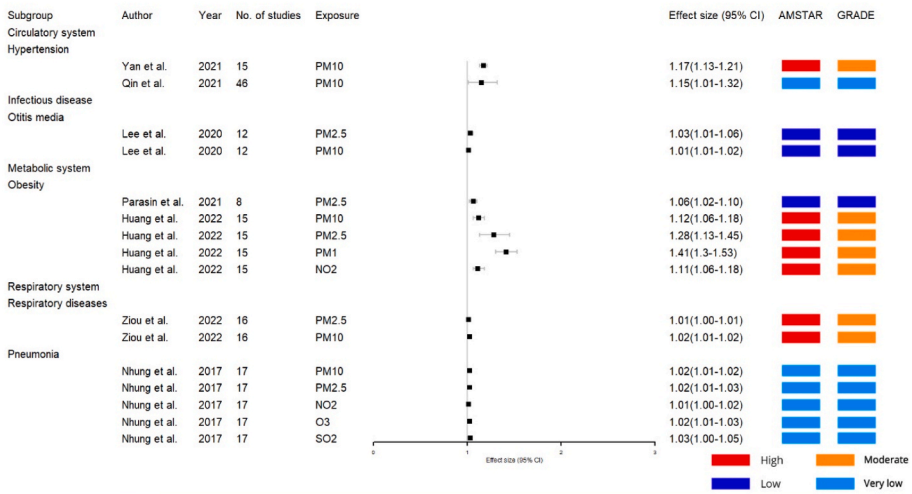

Fig. 4. Association between air pollution and the circulatory system, infectious diseases, metabolic system, and respiratory system.

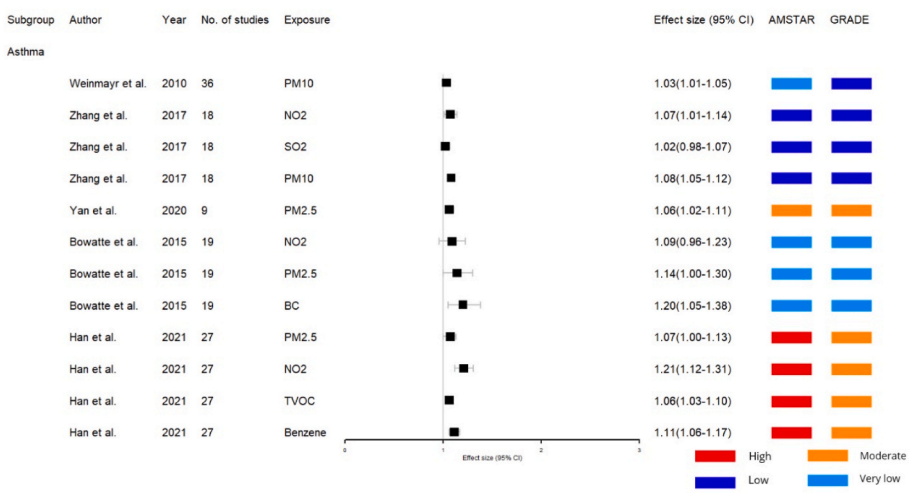

Fig. 5. Association between air pollution and the respiratory system.

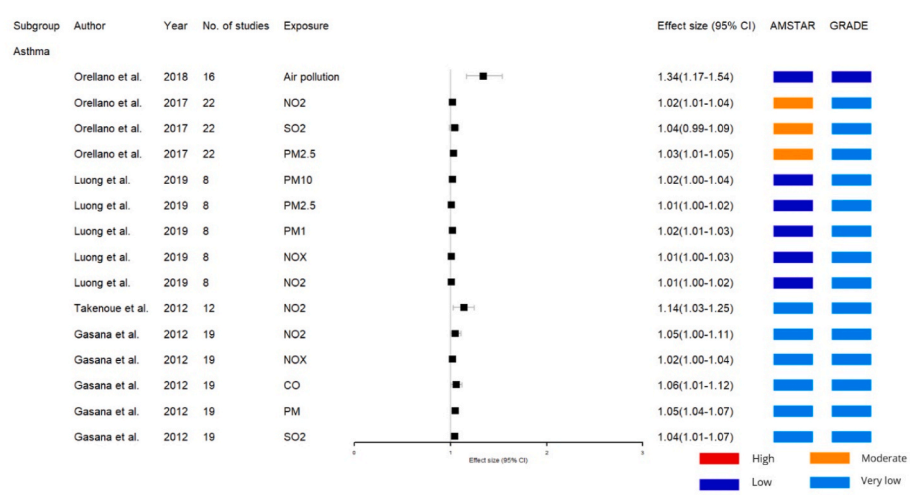

Fig. 6. Association between air pollution and the respiratory system.

systematic reviews (Fig. 7).

3.11.1. Eczema

Children's eczema risk was significantly associated with maternal prenatal exposure to nitrogen dioxide (RR = 1.13 (95%CI: 1.06, 1.19)),

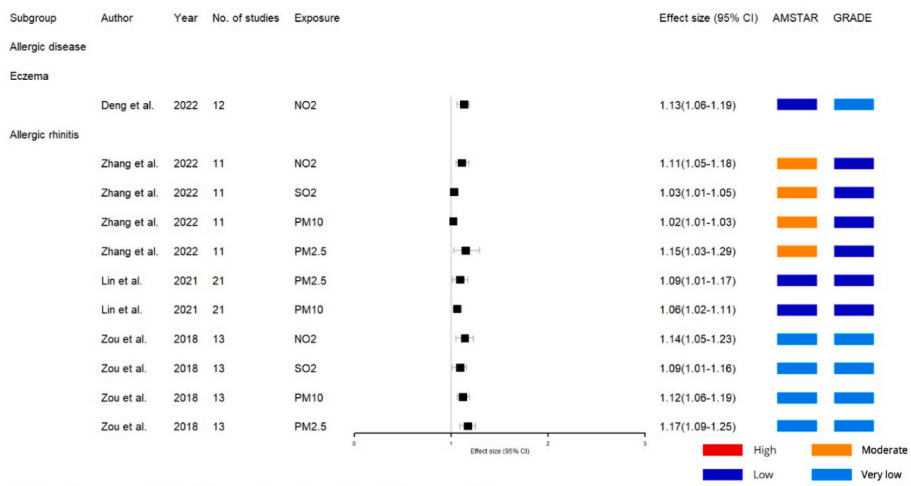

Fig. 7. Association between air pollution and allergic diseases.

particularly during the first and second trimesters (Yue et al., 2022). The study was rated low by AMSTAR 2 and very low by GRADE.

3.11.2. Allergic rhinitis

The results of three meta-analyses (Zhang et al., 2022; Lin et al., 2021a,b; Zou et al., 2018) suggest that exposure to NO_2 (RR = 1.11 (95% CI: 1.05, 1.18)), SO_2 (RR = 1.03 (95%CI: 1.01, 1.05)), PM_{10} (RR = 1.02 (95%CI: 1.01, 1.03)), and $PM_{2.5}$ (RR = 1.15 (95%CI: 1.03, 1.29)) may increase the risk of allergic rhinitis in children. Furthermore, particulate matter poses a greater threat than gaseous pollutants, and $PM_{2.5}$ is more likely to affect developing countries and regions, especially China. PM_{10} is more likely to have an impact on developed countries and regions. The AMSTAR 2 scores for the three studies were moderate, low, and very low, and the GRADE scores were low and very low, respectively.

4. Discussion

4.1. Main findings

A total of 33 systematic reviews were identified in this review (30 meta-analyses were performed and 3 were not performed), including 3 on the blood system, 2 on mortality, 2 on the nervous system, 3 on the circulatory system, 1 on infectious diseases, 3 on the metabolic system, 15 on the respiratory system, and 4 on allergic disease. Overall, we observed that air pollution exposure can increase the risk of leukemia, autism spectrum disorders, hypertension, otitis media, obesity, pneumonia, asthma, other respiratory diseases, allergic rhinitis, eczema, and childhood mortality.

This umbrella review had a slight overlap (1.23% CCA). So far, no other umbrella reviews that have been written and published that may be compared with this have been found. While it is widely recognized that different pollution levels can have differing toxic and health effects, low-level exposure to air pollution is often underestimated. Certain diseases have been linked to even relatively low concentrations of ambient air pollution (Nhung et al., 2017). The association was more significant for particulate matter than gaseous pollutants for increased risk of childhood death (Karimi and Shokrinezhad, 2020). Among particulate matter, $PM_{2.5}$ is more harmful to allergic rhinitis in children than PM_{10}, possibly because its aerodynamic equivalent diameter is smaller and it can enter deeper airways (Lin et al., 2021a,b). Further, air pollution exposure and physical health are more closely related for boys than for girls because boys tend to spend more time outdoors than girls (Zhang et al., 2022). Assessment of ambient air pollution exposure may be an important source of heterogeneity, and there are many assessment methods, such as air station monitoring data, temporal and spatial-temporal models, etc. Some used the spatial-temporal variation, while others didn't (Yan et al., 2021).

Air pollution is composed of a variety of pollutants, but particulate matter is the most representative and harmful to human health (Lin et al., 2021a,b), among which fine particles are more dangerous than coarse particles. Therefore, this paper summarizes the effects of $PM_{2.5}$ on children's health. PM2.5 affects the neurodevelopment of early children and is significantly associated with autism spectrum disorders (Lin et al., 2021a,b). PM2.5 is associated with increased systolic and diastolic blood pressure and can affect the cardiovascular system through various mechanisms (Huang et al., 2021). Short-term increases in PM2.5 concentration may be associated with the development of otitis media (Lee et al., 2020). PM2.5 can not only affect body weight through direct mechanisms, but also lead to obesity through some indirect forms, such as affecting sleep and leading to weight gain (Parasin et al., 2021). There is some association between long-term exposure to PM2.5 and hospitalization for bronchiolitis, but the results are not very consistent among studies (King et al., 2018). However, short-term or long-term exposure to PM2.5 can significantly reduce lung function (Zhang et al., 2022). It is also associated with hospitalization for pneumonia in children (Nhung et al., 2017). PM2.5 is not only associated with wheezing related diseases in children (Orellano et al., 2017), but also can aggravate asthma (Luong et al., 2019). In addition, exposure to PM2.5 is also associated with allergic rhinitis in children (Zhang et al., 2022; Lin et al., 2021a,b; Zou et al., 2018). It is also because of the association of PM2.5 with multi-system diseases that it is further associated with child mortality (Karimi and Shokrinezhad, 2020).

The majority of studies were from the United States and China (44.11%, Table S8). With relatively few studies from developing regions and a lack of data on high pollution levels or vulnerable populations, the call for an investigation in LMICs cannot be overemphasized in the future.

4.2. Biological plausibility

Air pollution and childhood leukemia are associated primarily with benzene, according to biological evidence. Benzene metabolites can cause DNA damage through a variety of mechanisms, and benzene is therefore considered to be a hemotoxic substance and a leukemic progenitor (Whysner et al., 2004). The underlying mechanism by which air pollution contributes to child deaths has not been elucidated, but the process should be related to children's immature immune system and respiratory system (Karimi and Shokrinezhad, 2020). At present, the mechanism of allergic rhinitis caused by particulate matter is not clear. For asthma, common air pollutants may cause or aggravate asthma

through different mechanisms. Exposure to particulate matter may trigger airway inflammation and oxidative stress and may further cause asthma (Orellano et al., 2018). The mechanism by which particulate matter causes autism spectrum disorders in children has not been clarified. Although the specific mechanism of particulate matter causing hypertension is not clear, it is believed to be related to oxidative stress and inflammation caused by exposure to particulate matter. In addition, particulate matter can cause endothelial dysfunction and an imbalance of vascular homeostatic responses, which may result in an increase in blood pressure (Brook and Rajagopalan, 2009). The exact role that particulate matter plays in the development of children's otitis media is unknown. However, particulate matter can affect the occurrence and development of otitis media by promoting cell apoptosis (Song et al., 2012). The mechanisms that link air pollution exposure to childhood obesity are unclear. But it has been linked to oxidative stress, inflammation, and insulin resistance caused by air pollution. In addition, air pollution may reduce people's desire for outdoor activities (Huang et al., 2022; Bahreynian et al., 2019). Air pollution during pregnancy may increase the risk of childhood eczema and allergies through an unknown mechanism. Oxidative stress can also damage the skin barrier (Yue et al., 2022; Ahn, 2014).

4.3. Strengths and limitations

This evaluation of the umbrella provides a number of benefits. First, this umbrella review provides a comprehensive overview of the effects of air pollution on children's health. We conducted a comprehensive literature search and included a large number of studies, including 33 systematic reviews and 526 primary studies covering health outcomes for 7 to 25,704,288 participants in 9 different systems or organs. Second, the PRISMA guidelines and the PRIOR statement were used to achieve this review of methodological rigor. Third, this study achieved visual presentation and comparative analysis of the results of each review, which can assist and guide the work of healthcare decision makers. Fourth, the results of CCA calculations indicate a slight degree of overlap. Several limitations should be considered. First, our study is restricted to English publications, which may limit the general applicability of the research results; however, it appears to have little effect on the conclusions (Dobrescu et al., 2021; Morrison et al., 2012). Second, there were more studies in the United States (125/526) and fewer studies in areas with less developed economies or high pollution levels, which may have led to selection bias. However, this umbrella review included component studies in both highly polluted (China) and less polluted (USA) settings. Third, common child health outcomes have been more intensively studied than others. Based on such studies with small sample size or few cases, the results of observational studies are inevitably biased. In addition, children often face the common harm of multiple pollutants, and it is difficult to distinguish the separate effects of a pollutant and the combined effects of multiple pollutants. Fourth, primary studies with prospective or retrospective designs can lead to bias and confounding. Although the most important confounding factors were controlled in most of the preliminary studies (65.02% for the gender of the child and 42.59% for the age of the child), residual confounding could not be completely excluded. Fifth, the unavoidable high heterogeneity. This might be due to regional variations in exposure levels, illness definitions, and exposure evaluation approaches. Conclusions should be interpreted and applied with full consideration of the strengths and limitations of this umbrella analysis.

4.4. Implications for research and public health

Children of various ages were included in the original studies since the great majority of them did not limit the age of the children. For children, who grow and develop at a faster rate, the effects of air pollution may be different at different ages. Therefore, future researchers should take the age of the children into account and try to include studies where the children are close in age. This will also allow for a more precise implementation of conservation measures.

There is an old saying in China: better safe than sorry. Given the possibility of epidemiological evidence that may link exposure to ambient air pollution in children to poor health, the precautionary principle needs to be considered, which states that "Precautions should be taken when an activity presents hazards to the environment or human health, despite the fact that certain cause-and-effect relationships are not fully established by science" (Hooper and She, 2003). The preventive measures to reduce threats are significant for air pollution since it frequently exhibits persistent positive connections with adverse health outcomes, notwithstanding the difficulties in proving causation with certainty. Healthcare professionals are better placed to inform children and their parents about the potential hazards of exposure to ambient air pollution and the need for targeted preventive action, such as reducing outside activities, wearing particle filters or masks in places with low air quality, and adopting air quality when choosing a residential location. Despite the fact that WHO air quality guidelines have established maximum levels for air pollutants, it does not appear that these levels are sufficient to ensure that infants are protected from air pollutants, so it is of benefit to present and future generations to reduce the maximum levels in the future.

5. Conclusions

This comprehensive umbrella review shows that children who are exposed to air pollution are at greater risk of developing leukemia, allergic rhinitis, pneumonia, asthma, and autism spectrum disorders, as well as significant associations with child mortality, hypertension, otitis media, obesity, and eczema. This comprehensive umbrella review highlights the development and implementation of preventive measures to reduce global air pollution alert levels. Future studies should consider the health effects of exposure to air pollution in children at different ages.

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Data availability

Data will be made available on request.

CRediT authorship contribution statement

Keshuo Liu: Writing – original draft, Conceptualization, Funding acquisition, Visualization, Validation. **Huanhuan Zhang:** Visualization, Methodology, Data curation, Writing – review & editing. **Yacong Bo:** Methodology, Data curation. **Yao Chen:** Investigation, Resources, Methodology, Data curation. **Panpan Zhang:** Writing – review & editing, Funding acquisition. **Cunrui Huang:** Conceptualization, Writing – review & editing, Data curation. **Zengli Yu:** Conceptualization, Validation, Investigation, Data curation, Writing – review & editing. **Zhan Gao:** Validation, Investigation, Writing – review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.apr.2024.102108.

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ENVIRONMENTAL STUDIES

$PM_{2.5}$ exposure disparities persist despite strict vehicle emissions controls in California

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As policymakers increasingly focus on environmental justice, a key question is whether emissions reductions aimed at addressing air quality or climate change can also ameliorate persistent air pollution exposure disparities. We examine evidence from California's aggressive vehicle emissions control policy from 2000 to 2019. We find a 65% reduction in modeled statewide average exposure to $PM_{2.5}$ from on-road vehicles, yet for people of color and overburdened community residents, relative exposure disparities increased. Light-duty vehicle emissions are the main driver of the exposure and exposure disparity, although smaller contributions from heavy-duty vehicles especially affect some overburdened groups. Our findings suggest that a continued trend of emissions reductions will likely reduce concentrations and absolute disparity but may not reduce relative disparities without greater attention to the systemic factors leading to this disparity.

INTRODUCTION

Despite decades of progress in improving ambient air pollution in the United States, people of color still bear a disparate burden of air pollution (*1–12*). Within California, research has quantified and characterized these exposure disparities using both measurements and models (*13–18*). Solutions to this persistent inequality are increasingly a focus for academic research and environmental policy at the federal, state, and local levels (*9*, *17–19*). A growing body of research investigates how air quality policies might contribute to a "triple win" that simultaneously achieves meaningful benefits by reducing population-wide exposures, mitigating greenhouse gas emissions, and reducing exposure disparities and extreme exposures (*20*, *21*). Here, we use a modeling framework to explore how multi-decade emissions reductions shaped by air quality and climate policies have affected environmental justice (EJ) outcomes, using California's aggressive on-road mobile source strategy as a case study. In this work, we focus on exposure disparities, which can be distinct from disparities in health outcomes (*22–27*).

Recent research on how to reduce air pollution exposure disparities in the United States presents two conflicting approaches (*9*, *19–21*, *28*). The first approach starts from the recognition that many major emitting sectors lead to disparate exposures for people of color (*2*). Thus, focusing on emissions reductions for sectors that especially affect people of color could have EJ co-benefits (*21*, *29–32*). This approach mirrors the policy structure in the United States and elsewhere, where environmental regulations are targeted to individual economic sectors (e.g., vehicles, industries, and power plants) and tailored to relevant technology and infrastructure. The second body of research suggests that sector-oriented policies may not be effective in addressing relative disparities in air pollution. For example, optimization modeling found that aggressive nationwide emissions reductions targeting economic sectors with higher-than-average disparity impact would not eliminate racial-ethnic fine particulate matter ($PM_{2.5}$) exposure disparities without nearly eliminating emissions (*9*). In contrast, a location-specific approach—i.e., emissions reductions by location rather than by economic sector—rapidly eliminated disparities. Building upon this finding, two recent studies (*20*, *21*) simulated climate policies with substantial abatement of $PM_{2.5}$ and its precursors across most US economic sectors and found modest potential reductions in disparities. They too reported that "location-specific" policies that target emissions reductions in all sectors within specific overburdened geographies may have a high potential to address relative exposure disparities even with small emissions changes (*9*, *19*). To complement prospective studies, which consider ways to reduce future exposure disparity, we examine the disparity impacts of historical emissions trajectories. We focus on the transportation sector, which is often highlighted as having a high potential to reduce exposure disparities. Historically, racist urban planning and infrastructure decisions (e.g., redlining and freeway siting) have concentrated vehicle emissions in communities of color (*2*, *4*, *7*, *13*, *29*). Furthermore, people who are exposed to the highest levels of traffic-related air pollution often are not the communities who drive the most (*30–32*). Hence, a recent study found that emissions controls for the transportation sector have the greatest potential to mitigate racial-ethnic inequality in US air pollution (*21*). Simultaneously, the transportation sector is a priority area for regulatory agencies and EJ-oriented community groups; emissions reductions from these sources could potentially reduce exposure disparities, human health impacts, and greenhouse gas emissions (*33*).

For nearly 60 years, California led the United States in reducing on-road vehicle emissions. Because California's motor vehicle emission regulation preceded the Clean Air Act of 1970, California is delegated the authority to set vehicle emissions standards more stringently than the federal equivalent (*34–36*). In the present analysis, we model exposure concentrations for the years 2000 through 2019, during which California's regulatory agencies pursued an aggressive and interlinked suite of multi-pollutant policies to reduce emissions across the entire on-road vehicle fleet (*36*). Examples include requiring cleaner fuels and technological advancements (e.g., hybrid drivetrain, alternative fuel and propulsion technologies, and

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advanced emissions controls) specific to light-duty, medium-duty, and heavy-duty vehicle classes (LDV, MDV, and HDV, respectively).

The suite of regulations that comprise California's mobile source strategy has resulted in large aggregate reductions of emissions of multiple pollutants from diverse fleets that make up the state's on- and off-road vehicles (*37*). Here, we examine how changes in on-road vehicle emissions from 2000 to 2019 have affected exposure to $PM_{2.5}$. Over this time period, on-road emissions have been shaped by several aggressive state regulations targeting specific vehicle fleets, including California Air Resources Board's (CARB) Light-Duty Vehicle Emissions Standards, Advanced Clean Cars, and the Truck and Bus Regulation (*38*). Despite statewide and fleetwide on-road vehicle miles traveled increasing ~24%—from 292 billion (2000) to 364 billion (2019)—emissions of the four species that principally drive population-weighted $PM_{2.5}$ exposures from on-road vehicles have decreased. Regulatory emissions data indicate reductions of ~70% for primary $PM_{2.5}$, nitrogen oxides (NO_x), and volatile organic compounds (VOC), while ammonia (NH_3) decreased by ~15% (fig. S1) (*38*). Notably, non-exhaust primary $PM_{2.5}$ emissions (e.g., brake and tire wear) have increased by ~20% over this time period, causing the relative non-exhaust share of primary $PM_{2.5}$ to increase substantially (14 to 50% from 2000 to 2019) (*39*). Diverse measurement and observational datasets (see the Supplementary Materials) corroborate overall declining emissions of PM, NO_x, VOC, NH_3, and other key traffic-related air pollutants (TRAPs) (*40–50*). Considering all species that contribute to total $PM_{2.5}$, California's on-road emissions reductions outpaced the national aggregate, especially for NO_x and VOC (*51*).

On-road vehicle emissions are anticipated to continue to decline in California in response to major new regulations: Advanced Clean Cars II (starting in 2035, requires all new passenger cars, trucks, and SUVs sold in California to be zero-emission vehicles) and Advanced Clean Fleets (starting in 2045, all trucks that drive in California must use zero-emissions technology). A few recent studies have projected the air pollution and equity impacts of vehicle electrification in California and found limited equity benefits. In this paper, we build on a much smaller body of work (*14*, *52*) to focus retrospectively on the equity impacts of past changes in vehicle emissions over two recent decades to inform future policy.

We investigate whether the combined impacts of the ensemble of mobile source strategies have contributed to a reduction in $PM_{2.5}$ exposure disparities. Exposure disparities are multifaceted; we quantify them along several axes described below. Our analysis also considers two specific features (vehicle type and spatial scale) that are central to current regulatory design. We conclude with implications from this California-focused retrospective analysis for future EJ-focused policy for the United States.

We developed and used an open-source analysis method based on atmospheric simulations from the Intervention Model for Air Pollution (InMAP; see Materials and Methods) to model total $PM_{2.5}$ concentrations resulting from emissions of $PM_{2.5}$, NO_x, VOC, NH_3, and sulfur oxides (SO_x) emitted by California's on-road mobile source sector from 2000 to 2019. Estimates of on-road mobile emissions are from CARB's EMission FACtor regulatory model (EMFAC v2021 with MPOv11), which has been approved by the US EPA (*53*). EMFAC represents CARB's best estimate of on-road emissions; it incorporates detailed administrative and observational data pertaining to fleet composition, emissions performance, and spatiotemporal activity patterns (*38*). Variably sized gridded $PM_{2.5}$ concentrations (1 to 48 km, higher resolution in greater population density locations) are combined with tract-level 2010 census population data to estimate exposure disparities among demographic groups (*15*). We disaggregate mobile source impacts into four vehicle types: LDV, MDV, HDV, and all other vehicles (e.g., buses, motorcycles, and motorhomes; table S1).

In the United States and California, air pollution exposure disparities tend to be larger by race-ethnicity than by other socioeconomic and demographic indicators (e.g., income, education, and urbanicity) due in large part to the historical racism and racist practices (e.g., housing discrimination, redlining, and highway relocation) that segregated cities and placed high-pollution sources near communities of color (*2–4*, *10*, *11*, *54*). Accordingly, we focus our analyses on racial-ethnic disparities. In addition, we consider two statutory geographic designations (AB617 and SB535) of cumulative impacts that California uses for prioritizing EJ (fig. S2) (*55*, *56*). Although these geographies have only recently been established (and thus past policy may or may not have explicitly targeted these places), we focus on them here because they are examples of location-specific policies that target emissions reductions in overburdened communities. Through the Community Air Protection Program (AB617), California has designated specific communities (2.7 million people, year 2010; 8.1% of the state's population) for priority in community-based air pollution monitoring and emissions reduction plans (*55*). A second policy, SB535 (10.2 million people, 30.0% of the state's population), focuses on targeting financial investments toward people living in "disadvantaged communities," identified using several environmental, socioeconomic, and public health indicators for each US census tract in California (*56*, *57*).

From here onward, we use the term "overburdened communities" to refer specifically to the areas designated as AB617 or SB535 communities and refer to the people who live in these areas as "residents of overburdened communities." The demographic makeup of residents of overburdened communities has a higher proportion of people of color (all groups except for non-Hispanic white Californians) than the statewide population (people of color: 92.9% in AB617 communities, 82.9% in SB535 communities; table S2). We also specifically consider exposure and disparities experienced by individual racial-ethnic groups (e.g., Hispanic Californians).

RESULTS AND DISCUSSION

Statewide trends in overall exposure and relative exposure disparity

California's mobile source policy has succeeded in its overall goal of reducing $PM_{2.5}$ exposures (Fig. 1A). We find that the modeled statewide population-weighted mean (PWM) $PM_{2.5}$ exposure concentration attributable to on-road vehicles decreased from approximately 3.2 to 1.1 $\mu g/m^3$ from 2000 to 2019, a ~65% decrease (i.e., nearly a factor of 3) in exposure on average for all Californians. This reduction in $PM_{2.5}$ exposure from on-road vehicles outpaced the overall statewide improvement in ambient air quality (fig. S3). For context, multiple independent estimates of total PWM $PM_{2.5}$ from all sources in California show an approximate ~40% decrease from 15 to 9 $\mu g/m^3$ from 2000 to 2019 (*58–60*).

We evaluate PWM $PM_{2.5}$ exposure from on-road mobile sources for racial-ethnic groups and residents of overburdened communities (Fig. 1A). Our modeled estimate of $PM_{2.5}$ declined for all groups,

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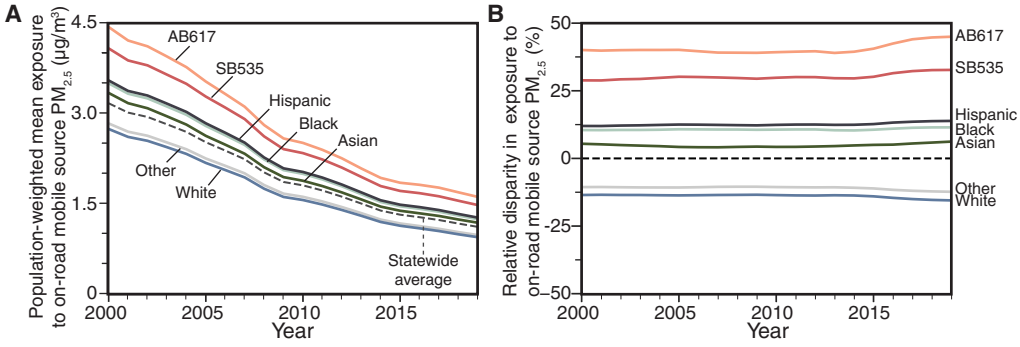

Fig. 1. On-road mobile source $PM_{2.5}$ exposure and relative disparity in exposure for each demographic group. Statewide population-weighted mean $PM_{2.5}$ exposure concentrations (**A**) and relative disparity in exposure (**B**) attributable to on-road mobile sources for the four largest racial-ethnic groups and two policy-relevant environmental justice areas in California. In each year, relative exposure disparities (B) for each racial-ethnic group are computed in reference to the statewide average $PM_{2.5}$ concentration attributable to on-road mobile sources. Concentrations in overburdened communities designated under California's Community Air Protection Program (AB617, ~10% of state population) and as SB535 Disadvantaged Communities (~25% of state population) substantially exceed those experienced on average for the most exposed racial-ethnic group, Hispanic Californians. Crucially, despite greater than 50% reductions in mobile source population-weighted mean $PM_{2.5}$ for all groups (A), relative racial-ethnic disparities increased for Hispanic, Black, and Asian Californians, as well as residents of overburdened communities. Here and elsewhere, the "Hispanic" population reflects Californians of any racial group identifying on the US census as Hispanic, while all other groupings exclude Californians identifying as Hispanic.

and the ordering of exposures by group is generally consistent over time. Among all racial-ethnic groups, Hispanic Californians experienced the highest exposure for all years, with PWM $PM_{2.5}$ exposure concentrations of approximately 3.5 and 1.3 μg/m^3 in 2000 and 2019, respectively. Black Californians experienced the next highest PWM exposure concentration (3.5, 1.2 μg/m^3 in 2000 and 2019, respectively), followed by Asian Californians (3.3, 1.2 μg/m^3). Of the four racial-ethnic groups in Fig. 1A, white Californians were exposed to the lowest PWM $PM_{2.5}$ concentrations: approximately 2.7 and 0.9 μg/m^3 from 2000 and 2019, respectively. Residents of overburdened communities were exposed to substantially higher PWM concentrations of $PM_{2.5}$ from on-road mobile sources (AB617 residents: 4.4, 1.6 μg/m^3 in 2000 and 2019; SB535 residents: 4.1, 1.5 μg/m^3 in 2000 and 2019) than the PWM for any racial-ethnic group shown in Fig. 1A.

For each demographic group, we compute exposure disparity as the absolute (in micrograms per cubic meter) and relative (percent) difference between the average modeled concentration experienced by a group versus the overall state population (Fig. 1B and table S3; see also Materials and Methods). In this work, we discuss exposure disparities in both absolute and relative terms. Both metrics provide useful insights into exposure inequality. Because increases in $PM_{2.5}$ concentration have a causal relationship with increases in adverse health outcomes, absolute differences between groups of people must be minimized to the extent possible. However, systemic inequality in terms of relative exposure disparity can persist even if the most overburdened areas receive the largest reductions in exposure in absolute terms if those reductions are not also the largest in percentage terms. Crucially, our analyses focus exclusively on $PM_{2.5}$ exposure disparities attributable to on-road vehicles. Most other major emitting sectors in California also disparately expose residents of overburdened communities and people of color to $PM_{2.5}$ (*2*, *15*). Likewise, exposures to other air pollutants are also unequally distributed (*4*, *7*, *10*). Accordingly, when we find that disparities persist, they persist in a larger story of environmental inequity in California.

Reflecting the nearly parallel exposure concentration traces over time evident in Fig. 1A, relative disparities in $PM_{2.5}$ exposure from on-road mobile sources (Fig. 1B) were notably persistent, increasing slightly over this time period. The relative disparity in exposure to on-road mobile sources for Hispanic Californians increased slightly from 12.0 (the year 2000) to 13.9% (the year 2019), while the relative disparity in exposure for white Californians decreased slightly from −13.5 to −15.5%. Thus, the overall relative difference between the most and least exposed race-ethnicity increased from 30 to 35%. Given expected model uncertainties, these incremental changes may not necessarily represent evidence of a trend that is distinguishable from approximately constant relative disparity. Likewise, relative disparities for Black and Asian Californians also persisted (respectively 10.5 to 11.5% and 5.4 to 6.2% over this period). Exposure disparities by race-ethnicity are larger than by income (fig. S4). Notably, we find persistent disparities in exposure to both primary and secondary $PM_{2.5}$ from vehicle emissions. Relative disparities in exposure to primary $PM_{2.5}$ components (18.6% for Hispanic Californians) were larger than disparities in exposure to secondary $PM_{2.5}$ (11.1% for Hispanic Californians).

Absolute and relative exposure disparities in overburdened communities are even larger. For example, the relative disparity in exposure (i.e., relative to the overall population average) for on-road mobile source $PM_{2.5}$ is more than three times as large for AB617 communities as for the most-exposed racial-ethnic group, increasing somewhat from 40 (the year 2000) to 45% (the year 2019). Stratifying by CalEnviroScreen score (which is used in part to identify SB535 communities), we find even larger relative disparities (fig. S4).

Disparities in exposure for those living at the extreme ends of the concentration distribution are also relevant for understanding environmental injustice. We estimated population-weighted distributions of exposure by race-ethnicity for each vehicle class (figs. S5 and S6). In general, changes in exposure at the upper percentiles (i.e., 75th and 90th) are consistent with changes in exposure at the PWM and consistent across time. Considering the disparity in exposure at the 75th and 90th percentiles relative to the statewide mean, we find large and increasing relative disparities (e.g., 90th percentile exposure for Hispanic Californians increased from 104 to 118% higher than statewide PWM from 2000 to 2019).

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We also evaluated the degree to which the California populations who experience the highest overall exposure to $PM_{2.5}$ from on-road vehicles are disproportionately composed of people of color, and how this pattern has evolved over time. To do so, we binned the California population by decile of modeled exposure to $PM_{2.5}$ and then compared the racial-ethnic composition of each decile in 2000 and 2019 (Fig. 2 and midpoint result in fig. S7). From 2000 to 2019, Hispanic Californians are overrepresented at the highest exposure deciles. While the California state population is 37.6% Hispanic, the highest decile of exposure for emissions in 2000 and 2019 consists of 47.9 and 50.8% Hispanic people, respectively. Similarly, white Californians, who comprise 40.1% of the population, are overrepresented among the populations with the lowest exposures (62.0% of the lowest-exposure deciles in 2000 and 2019) and are underrepresented in the highest-exposure decile [29.9% (2000) and 27.7% (2019)]. In fig. S8, we examine the racial-ethnic composition of the population across the full distribution of absolute and percentage changes in $PM_{2.5}$ exposure from on-road vehicles. While the grid cells with the largest absolute reduction in concentration consist of more people of color than the statewide average, there are only small demographic differences in the percentage change in exposure. This result arises in large part because the geographies with the largest absolute reductions in $PM_{2.5}$ exposure from on-road mobile sources started with the highest initial levels of exposure in 2000.

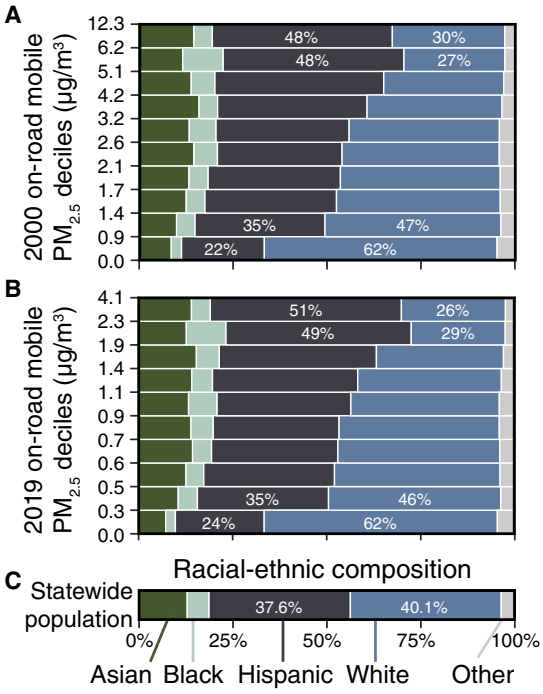

Fig. 2. Racial-ethnic population distribution by exposure decile. Differences in the racial-ethnic composition of the California population exposed to each decile of the distribution of $PM_{2.5}$ attributable to on-road mobile sources in (**A**) 2000 and (**B**) 2019. The statewide population is binned into 10 groups of equal population of $PM_{2.5}$ exposure attributable to the full vehicle fleet. At all years in our assessment, Hispanic Californians are strongly overrepresented among the highest $PM_{2.5}$ exposure deciles (and underrepresented in the lowest exposure deciles). The opposite pattern holds for white Californians. Data are plotted for individual vehicle types and the analysis midpoint year (2010) in the Supplementary Materials.

Differences in contributions to exposure disparity by fleet type

Because California's vehicle emissions control policies generally differentiate by vehicle type, we disaggregate our analyses of emissions, exposures, and disparities by vehicle type based on the official EMFAC2021 documentation (table S1) (*38*). We model the disparities and additive contributions of each vehicle fleet type at the state level for the most exposed racial-ethnic group, Hispanic Californians, to identify which vehicle types have an especially influential role in their exposures and disparities.

At the statewide average, we find that LDVs are the vehicle fleets with the largest aggregate impact on overall PWM $PM_{2.5}$ exposures and absolute disparities. For example, considering Hispanic Californians, LDVs account for 65 to 70% of the 0.2 to 0.4 $\mu g/m^3$ absolute disparity in $PM_{2.5}$ exposure from on-road mobile sources (Fig. 3, A and B). Contributions to the absolute disparity from HDVs (16 to 24%), MDVs (9 to 14%), and all other vehicles (<5%) are substantially smaller. Considering the PWM distribution of $PM_{2.5}$ by vehicle fleet type and race-ethnicity, we find broadly similar racial-ethnic distributions of exposure attributable to LDVs, HDVs, and MDVs, with Hispanic Californians receiving the highest exposures (fig. S5). Between 2000 and 2019, the fractional contributions to absolute disparity from individual vehicle fleet types were stable, likely reflecting the roughly constant distribution of vehicle activity patterns by vehicle fleet.

From here on, we focus our discussion on LDVs and HDVs, which, in combination, account for >80% of exposures and absolute disparities attributable to on-road sources (see fig. S7 for detailed results for other fleets). The dominant influence of LDVs on exposure holds across racial-ethnic groups and for residents of overburdened communities (figs. S9 and S10), but with different overall magnitudes of exposure for different subpopulations. This result likely arises for two reasons. First, LDVs dominate the overall emissions of $PM_{2.5}$ and its precursors. On the basis of the CARB emissions inventories used, LDVs contribute most of the NH_3 and VOC emissions (70 to 95% of NH_3 and >80% of VOC) from vehicles, which account for ~44 to 56% of total $PM_{2.5}$ exposure from vehicles. LDVs and HDVs contribute more similarly to primary $PM_{2.5}$ (23 to 45% LDV and 29 to 56% HDV) and NO_x (35 to 48% LDV and 36 to 43% HDV) emissions, and these species contribute the remaining ~44 to 56% of total $PM_{2.5}$ (figs. S11 to S14). Primary $PM_{2.5}$ emissions are more weighted toward non-exhaust emissions in recent years, especially for LDVs (figs. S11 to S14). Second, LDV emissions are more concentrated near population centers than other vehicle fleets, so LDVs result in a substantially higher-than-average exposure impact [fig. S15, population-weighed exposure per ton of annual emissions (metric: micrograms per cubic meter); this metric is directly related to intake fraction, e.g., (*61–63*)].

While the high activity of LDVs causes a higher aggregate impact on disparity, HDVs stand out as the fleet type whose emissions cause the most disparate impact on Californians of color. As a complement to apportioning the overall absolute exposure disparity to emissions from individual vehicle types (i.e., largest aggregate impact), in Fig. 3C, we also consider which vehicle fleet types have an especially disparate impact on specific racial-ethnic groups (largest relative impact regardless of the magnitude of emissions) relative to the statewide population. For example, the relative disparity caused by HDVs for Hispanic Californians (range: 16 to 17%) was larger than the relative disparity caused by LDVs (range: 11 to 14%). This difference

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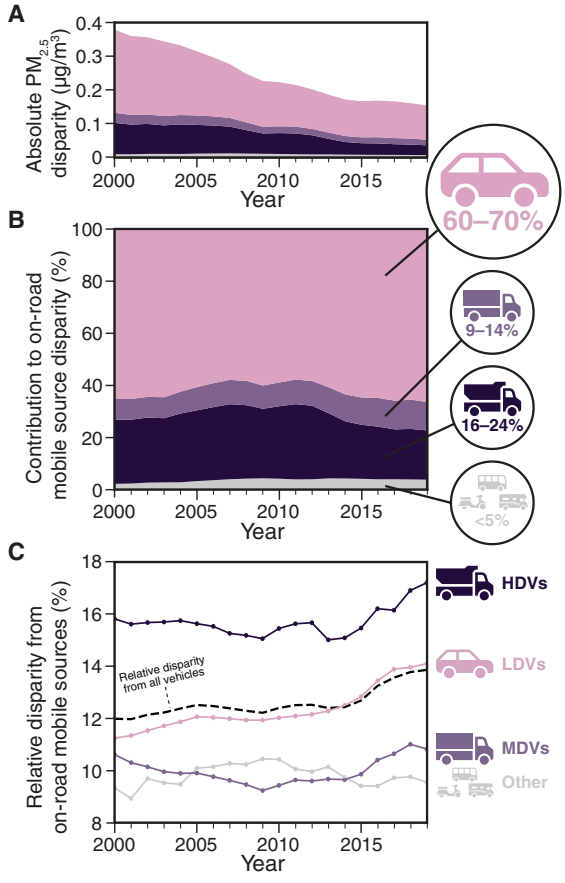

Fig. 3. Contributions to disparity in exposure to mobile-source $PM_{2.5}$ for Hispanic Californians. Two methods of comparing contributions to disparity in $PM_{2.5}$ exposures from on-road vehicle fleet types shown for the most exposed racial-ethnic group, Hispanic Californians. First, we compare the absolute magnitude in contribution from each vehicle group (A and B); then, we compare the relative disparity in exposure to each vehicle group (C). (**A**) Absolute disparities in $PM_{2.5}$ exposure from vehicles for Hispanic Californians relative to the overall statewide population declined between 2000 and 2019, consistent with the overall reduction in emissions (fig. S1) and population-weighted mean $PM_{2.5}$ concentrations (Fig. 1). (**B**) Fractional contributions to the overall disparity that are attributable to each fleet type are estimated by normalizing the absolute contribution to disparity attributable to a single fleet type to the total disparity attributable to all on-road mobile sources. In each year, light-duty vehicle (LDVs) emissions are the dominant contributor to the disparately high exposures experienced by Hispanic Californians. (**C**) Disparities attributable to emissions of individual vehicle fleet types relative to the statewide average $PM_{2.5}$ exposure attributable to emissions of that individual vehicle fleet. Note that heavy-duty vehicles (HDVs) especially disparately impact Hispanic Californians, even though HDVs are not the dominant contributor to overall emissions (fig. S1), PM2.5 concentrations (Fig. 1), or absolute disparities (C).

in impacts by fleet type is consistent with recent traffic equity modeling, which demonstrated that the majority of Californians of all race-ethnicity are exposed to high annual average daily traffic from LDVs, but Californians of color are disproportionately exposed to higher annual average daily traffic from HDVs (*64*). Another useful metric for discussing the especially disparate impact of HDVs is the exposure inequality, defined by Demetillo *et al.* (*65*) as the percent difference in exposure between the most and least exposed racial-ethnic groups. On the basis of our results, the $PM_{2.5}$ inequality for Hispanic Californians, relative to white Californians, increased from 37% in 2000 to 41% in 2019. This finding complements recent work that shows the importance of HDV emissions mitigation for reducing racial-ethnic disparity (*23*, *65*).

Substantial heterogeneity in fleet-wise contributions at the community scale

We find that there is substantial spatial heterogeneity in how different vehicle fleet types contribute to $PM_{2.5}$ exposures. We compare modeled contributions by vehicle type at four spatial scales: statewide (Fig. 4A), regional (Fig. 4B), within overburdened communities (Fig. 4C), and community scale (Fig. 4D). Whereas the previous section and Fig. 4C evaluate aggregate exposure and disparity across all AB617 overburdened communities, in Fig. 4D, we compare contributions to exposure and disparity within individual overburdened communities. The primary goal of this analysis is to highlight the heterogeneity among diverse communities in how vehicle fleets

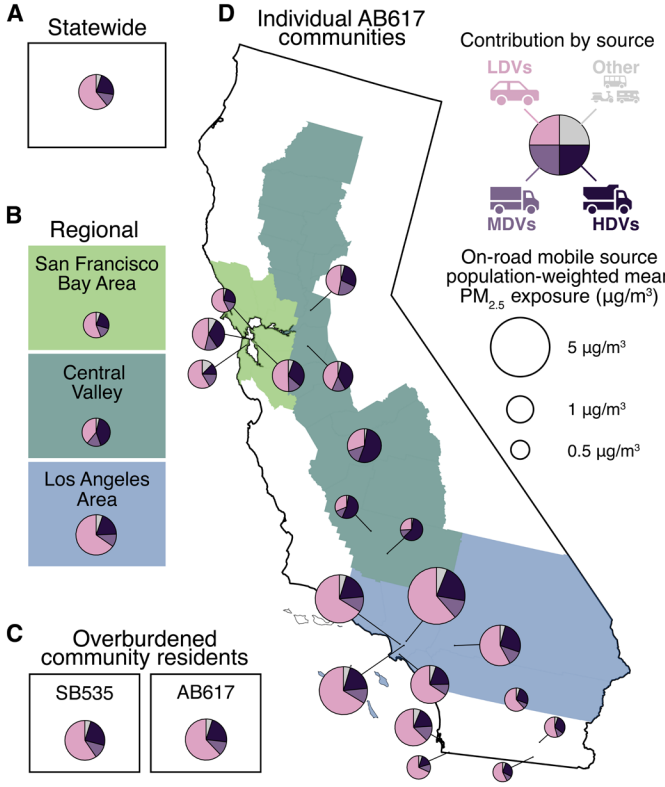

Fig. 4. Spatial heterogeneity in contributions by fleet to mobile source $PM_{2.5}$ exposure. Contribution to $PM_{2.5}$ exposures from distinct vehicle fleets is shown at four spatial scales: (**A**) statewide, (**B**) three major regions, (**C**) residents of overburdened communities, and (**D**) for 19 individual communities designated by the state of California through the Community Air Protection Program (AB617; see fig. S2 for identification of each community). At each spatial scale, pie chart icons indicate the fractional contribution to exposure attributable to each vehicle fleet type, with icons scaled in proportion to the population-weighted mean $PM_{2.5}$ concentration from all vehicle types. Light-duty vehicles contribute especially to mobile source $PM_{2.5}$ exposures in Southern California, while the relative contribution from MDVs and especially HDVs is comparatively higher in the Central Valley and San Francisco Bay Area. There is considerable heterogeneity among AB617 communities in fleet contributions.

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contribute to $PM_{2.5}$; our estimates are not meant to precisely capture community-scale pollution concentrations. As with any emissions inventory, modeled concentrations are much more precise with locally validated, site-specific information that has been observationally verified (*66*). To complement our high-level approach to understanding the heterogeneity in source contributions, future community-specific analyses could use higher spatial resolution modeling tools and local emissions data to better represent the lived experience of individual communities.

On average, the Los Angeles area and its AB617 overburdened communities have high PWM exposures and high contributions from LDVs (>60%). In the Central Valley, while the PWM exposures are lower, the contributions from HDVs are substantially higher (e.g., ~60% in Arvin/Lamont). The diversity in fleet contributions to individual communities showcases the importance of community-specific emissions reduction planning. While a community with a high share of LDVs, for example, might benefit more from policy actions that directly reduce those emissions (e.g., more electric bus routes and street conversion to bicycle paths), different strategies may be more appropriate for a community dominated by HDVs (e.g., additional diesel fuel emissions limits, truck electrification, and low emissions zones). These differences likely arise from differences in spatial distributions of sources relative to residences and the magnitude and mixtures of vehicle activity that occur at the community scale. In sum, our results support the approach of enabling communities to identify and mitigate the largest contributors to local exposures and disparities.

Validation, limitations, and implications for future research

Multiple lines of evidence suggest that our core qualitative results align with available observational evidence. A relative strength of our modeling approach is that it allows us to model temporal changes at a sufficiently high spatial resolution that we can estimate exposure disparities attributable to individual source categories. In contrast, a detailed longitudinal record of in situ observations is not available at sufficient spatial resolution to permit a rigorous assessment of how disparities in exposure to traffic-related $PM_{2.5}$ have evolved. Nonetheless, CARB's analyses of ambient monitoring data from 1990 to 2014 align qualitatively with our results. These analyses of monitoring data indicate declining concentrations of diesel PM, $PM_{2.5}$, and NO_2 but with persistent relative and absolute disparities for the relatively sparse network of sites located in overburdened communities (*60*).

As additional points of comparison for our modeled results, we examined datasets of finely resolved satellite observations and empirical model predictions. These datasets afford the ability to consider changes in exposure and disparity for the entire state (see Supplementary Text) (*58*, *59*). In fig. S3, we compare our analyses with changes in total $PM_{2.5}$ (only moderately influenced by vehicles) and NO_2 (strongly influenced by vehicles). Considering PWM concentration changes from 2000 to 2019, our estimated PWM $PM_{2.5}$ from on-road vehicles declined at a broadly similar rate (~65%) compared to the results from a high-resolution empirical model of PWM NO_2 spatial patterns (~55%). Our estimates of the racial-ethnic ordering of vehicle-emitted $PM_{2.5}$ exposures and disparities closely match those from the total $PM_{2.5}$ and NO_2 datasets. Crucially, our finding of temporally persistent relative disparities in exposure to $PM_{2.5}$ from on-road sources (Fig. 1B) is consistent with highly stable patterns of relative disparity in total $PM_{2.5}$ and NO_2 for Californians of color (fig. S3). Furthermore, we find that the magnitude of our estimate of traffic-related PWM $PM_{2.5}$ is consistent with the on-road vehicle contribution from previous modeling and in situ source apportionment studies in California (*67*–*72*). In combination, these supporting lines of evidence reinforce our key qualitative conclusion that exposures from mobile sources have decreased while relative disparities in exposure have persisted.

It is worthwhile to consider possible uncertainties, biases, and limitations associated with our approach. Our modeling framework is built around the InMAP reduced complexity model (*73*) and its associated InMAP source-receptor matrix [ISRM; (*28*)]. The computational efficiency of this model enabled us to interactively execute thousands of unique model runs representing distinct vehicle fleets for 20 individual years while maintaining a sufficiently fine scale (down to 1 km^2) to capture spatially sharp exposure disparities (*1*, *3*). However, our modeling approaches have notable limitations. First, as with any atmospheric modeling, our results rest on the validity of underlying emissions inventories, including how they represent patterns over time and space [see, e.g., (*74*)], as discussed briefly below. Second, InMAP makes simplifying assumptions that can lead to somewhat higher bias than traditional chemical transport models (CTMs), which model the underlying atmospheric chemistry and dynamics with higher fidelity. One such simplification in our model is a linear approximation of nonlinear secondary aerosol chemistry. Third, the temporal resolution of our results is limited to annual average conditions; we do not quantify exposure disparities that occur on seasonal, diurnal, or shorter-than-annual timescales, which are also relevant (*75*). In addition, our model does not capture sub-grid scale exposure gradients near roads; those gradients can occur and are important for exposure disparities at scales finer than 1 km (*76*). InMAP results are generally considered more robust for spatial aggregations of many grid cells (e.g., air basins and groups of overburdened communities), and less so for individual pixels or neighborhoods (*73*). Last, our core analyses assign exposures based on a fixed residential address [which can misclassify exposures; see, e.g., (*77*)] and were estimated using a temporally static year 2010 US census dataset (selected as the midpoint year of our study).

Considering that our key results emphasize the persistence over time of disparities (especially relative disparities)—rather than absolute concentrations at specific locations—our overarching qualitative insights are likely to be robust. Relative disparities are principally determined by the interaction of fine-scale spatial patterns of demographics, roadways, and fleet activity and are less sensitive to the magnitude of emissions or concentrations. In the Supplementary Materials (the "Model uncertainty and sensitivity" section of Supplementary Text and figs. S16 and S17), we explore how possible biases in emissions estimates and model performance could affect our results. We first review the literature to constrain our understanding of the uncertainty from the state regulatory model of on-road mobile source emissions estimates (*50*, *74*). From previous work validating EMFAC's on-road mobile source emission factors and a similar model's activity-based spatial surrogates, we believe that our qualitative insight is unlikely to be affected by bias in the emissions inventory. We then perform four tests to evaluate the sensitivity of our analyses to potential biases in the emissions and model (fig. S16). A key insight is that within the range of expected model biases for InMAP, we find that the magnitude and ordering of relative disparities are only minimally sensitive. This result arises in part because

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relative disparities involve ratios of modeled concentration estimates. In addition, because we find meaningful disparities for each of the five modeled $PM_{2.5}$ constituents, pollutant-specific model biases (representing, for example, a possible mischaracterization of the nonlinear chemistry in InMAP) are unlikely to strongly skew our results (see fig. S16). Likewise, this analysis implies that inventory biases that affect aggregate levels of emissions are unlikely to affect our core insights. We show that our results are robust against spatial biases in the emissions inventory by repeating our analysis with emissions from two independently derived, peer-reviewed emissions inventories (*78*, *79*) and coarser representations of our emissions inventory. Spatial emissions biases in the inventory could conceivably affect conclusions about relative disparities if they were much larger than what we explored in fig. S16. However, we consider this implausible given how closely our results align with disparity insights from high-resolution NO_2 predictions (fig. S3). Nonetheless, because neither EMFAC nor InMAP is meant to authoritatively describe emissions and concentrations in individual model pixels, we ascribe our greatest confidence to overall patterns in space and time, but caution against overinterpreting results for specific communities or other small regions. Last, in a sensitivity analysis (fig. S17), we repeated our analysis with 2000 decennial census data. These analyses indicate that our core qualitative finding of highly persistent relative disparity was generally not sensitive to the selection of the census dataset. In particular, the choice of which demographic dataset is used minimally affected the PWM concentrations and relative disparities experienced by overburdened communities and individual racial-ethnic groups in California. However, future demographic shifts in the California population that substantially alter patterns of social segregation could meaningfully affect aggregate air pollution disparities at the state level. For example, if suburbs become more racially integrated and California's population becomes increasingly diverse, then relative disparities could decrease as a function of demographic changes and not necessarily emissions mitigation.

Future research beyond the scope of this assessment could further corroborate our findings and build on our results. First, CTM simulations could usefully validate our core results, especially as they concern the behavior of secondary $PM_{2.5}$ from vehicle emissions. Second, it would be helpful to quantify the effect of decades of vehicle emissions controls on other air pollutants that are relevant to the health of overburdened communities, including nitrogen oxides, diesel PM, and air toxics. Third, although in situ observations of TRAPs have historically not been available at a sufficiently high spatial resolution to systematically characterize changes in disparity, careful analysis of data at particular locations may be able to complement our statewide insights. Moreover, as hyperlocal measurements of TRAPs become more widespread, these types of observational studies may be more feasible in the future.

Last, because disparities in terms of health outcomes are also relevant to EJ and distinct from exposure disparities, analyses that quantify the complex interplay of emissions, exposures, and social, demographic, and epidemiological factors could explore the impact of vehicle emissions on environmental health disparities over time (*22*, *23*, *25*–*27*, *80*–*83*). Disparities in health outcomes are strongly influenced by social determinants of health (e.g., age, obesity, access to health care, and criminal justice) that have persisted over time and are independent of air pollution (*80*, *84*). Several recent studies show that Black and Hispanic Americans in the United States have a higher susceptibility to air pollution than non-Hispanic white Americans (*81*–*83*, *85*). Thus, a focus on disparities in exposure may underestimate or mischaracterize the ultimate disparities in health outcomes (*22*–*27*). While this study focused exclusively on exposure disparities, effective policies should address disparities in both exposure and health outcomes.

Policy insights from California's historical mobile vehicle control policies

We have demonstrated that while modeled PWM $PM_{2.5}$ exposures and absolute exposure disparities attributable to on-road mobile sources have decreased over the past two decades across all population groups, relative disparities have remained at both the average and at the extreme ends of the exposure distribution for Californians of color and residents of overburdened communities. Emissions from LDVs and HDVs affect disparities in different ways. LDVs contribute the most to $PM_{2.5}$ concentrations and absolute disparity, while emissions from HDVs most disproportionately expose people of color relative to other fleet types, thereby highlighting the importance of mitigating emissions from both vehicle types. Of the groups considered here, residents of AB617 communities in aggregate experience the highest levels of $PM_{2.5}$ exposure from on-road vehicles, although PWM exposures for these residents have declined by more than 60% since 2000. There is substantial heterogeneity among AB617 communities in terms of the total exposure concentration and the relative contribution from each vehicle type.

Our finding of highly persistent relative disparities for Californians of color is disappointing but consistent with a growing body of literature on sectoral emissions policy. When policies reduce the overall emissions rate without substantially altering the pattern of where emissions occur, relative disparities in exposure can persist (*9*, *14*, *21*). In this vein, the findings from our retrospective analysis resonate with the results of studies that have prospectively modeled the potential future equity impacts associated with specific vehicle policies (e.g., heavy-duty truck electrification and zero-emission vehicle adoption). Consistently with those studies, we have found large absolute concentration changes in regions with the highest share of people of color, yet we nonetheless find a minimal reduction in the relative disparity for $PM_{2.5}$ exposure (*14*, *23*, *25*, *52*, *86*). These results arise because the places with the largest concentration changes over time tend to be the places most affected by vehicles (fig. S8).

While the sustained inequity in $PM_{2.5}$ exposure resulting from on-road mobile sources is problematic, California's mobile source strategy has led to large aggregate reductions in emissions, exposure concentrations, and absolute disparities. Although the relative disparity in exposure to $PM_{2.5}$ from on-road mobile sources is effectively unchanged for Californians of color and residents of overburdened communities, the PWM $PM_{2.5}$ exposures caused by these mobile sources were reduced by approximately 64% for all demographic groups considered during our study. On-road mobile source controls have also reduced emissions from a broad suite of TRAPs (*87*) that are also of health concern. For example, statewide on-road emissions of carbon monoxide, nitrogen oxides, and diesel PM also decreased by ~75% (*38*, *39*). Ambient concentrations of these pollutants have declined substantially in absolute terms, especially at sites in overburdened communities (*60*). These results speak to the value of both aggressive mobile source control and a multi-pollutant mitigation strategy that considers multiple TRAPs at once. Future mitigation efforts should continue this approach to avoid the risk of unintended

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consequences of single-pollutant control strategies (*52*). Despite this success, it is likely that relative disparities for other pollutants with similar spatial patterns of on-road emissions have persisted. For example, NO_x, for which on-road sources contributed 57% of total statewide emissions in 2000, is considered. From 2000 to 2019, our assessment of high-resolution empirical model predictions shows a ~55% decrease in PWM NO_2 but large and moderately increasing relative disparities by race-ethnicity (fig. S3).

It is useful to consider the implications of our retrospective assessment for California's current policy efforts that focus heavily on eliminating exhaust emissions across the on-road fleet through a combination of electrification and—in the case of HDVs—hydrogen. For every year of the study, approximately 90% or more of the PWM $PM_{2.5}$ exposure (and absolute exposure disparity) is attributable to exhaust emissions and approximately 80% or more is attributable to secondary formation from precursor exhaust emissions (figs. S11 to S14). Because California's policies contemplate eliminating exhaust emissions, this result implies that future vehicle electrification has the potential to substantially reduce exposures and absolute disparity. Nonetheless, PWM exposure to non-exhaust primary $PM_{2.5}$ emissions (i.e., brake and tire wear) increased somewhat from 2000 to 2019 (fig. S18), while relative disparities from non-exhaust primary $PM_{2.5}$ remained effectively constant. Non-exhaust emissions would not be fully eliminated through electrification and could conceivably be exacerbated by increases in vehicle mass (*88*). Thus, future low levels of exposure from non-exhaust emissions (e.g., brake and tire wear) might still disparately affect people of color and residents of overburdened communities.

Our results suggest that relative disparities in exposure will persist without a paradigm shift in transportation policy. Some policy approaches have the potential to reduce not only aggregate levels of exposure but also relative disparities. For example, creating low emissions zones or promoting a mode shift away from private automobiles (e.g., dense public transit networks and bike lane infrastructure) could be more likely to reduce exposure disparities from the on-road vehicle fleet than statewide fleet-specific emissions controls, while also improving air pollution throughout the system (*89*). Without systemic changes to transportation infrastructure, it seems possible that these relative disparities could persist even in a future, lower emission scenario. Conversely, by strategically accelerating emissions reductions, such as vehicle electrification efforts, with deployment emphasizing overburdened areas, EJ communities could achieve substantial short-term reductions in relative exposure disparity.

While we have focused on one sector within California, our findings contribute to an emerging body of EJ research indicating that to reduce relative disparities in exposure, the policy must not only continue a trend of emissions reduction but also target the disparate geographical distribution of emissions in overburdened communities. While we focused on California as a case study, it is possible that these general findings would apply across the United States, as most state and national approaches broadly have mirrored California's, with a strong focus on emission rate reductions. Our work provides a compelling illustration of how a highly successful emissions reduction strategy does not necessarily reduce relative disparity in exposures (*20*, *21*). More research is needed to identify the specific suite of strategies that can deliver a "triple win" for climate, health, and equity goals. We hypothesize that particularly effective strategies may go beyond aggregate emission rate reductions by ameliorating the inequitable spatial distribution of where activities and emissions take place. Thus, future work could explore the environmental equity impacts of potential policy actions and public investments that fundamentally change transportation infrastructure.

MATERIALS AND METHODS

Emissions estimates

We obtained estimates of mobile emissions in California from CARB's EMFAC model (version EMFAC 2021 with MPOv11) for calendar years 2000 through 2019 (*38*). The EMFAC model uses detailed California-specific data to estimate emissions by year and fleet and has been approved by the US EPA (*53*). Estimated emissions were spatially allocated to a 1 km–by–1 km grid using surrogates developed by CARB and CARB's Spatial and Temporal Allocator (ESTA) model. The ESTA model uses spatial surrogates that are derived from link-level traffic measurement data combined with population estimates and spatial information about idling locations, rest stops, and distribution centers (*90*). The resulting dataset contained spatially resolved annual total exhaust, evaporative, brake wear, and tire wear emissions for primary $PM_{2.5}$ and four precursor species: NO_x, VOC, NH_3, and SO_x. EMFAC2021 reports results for 54 vehicle categories and five fuel types (gasoline, diesel, natural gas, plug-in hybrid, and electric). Emissions for this analysis were binned into three main vehicle groups: LDVs, MDVs, and HDVs, with all other vehicle types (including motorcycles, motorhomes, and buses) grouped together as "Other" (table S1). Fleet information is derived from detailed data from the California Department of Motor Vehicles, the California Highway Patrol, the International Registration Plan Clearinghouse, and the National Transit Database (*38*). EMFAC is therefore capable of providing a reasonable representation of distinct activity and emissions patterns for specific vehicle fleets.

Estimates of air concentrations

We modeled annual average $PM_{2.5}$ concentrations attributable to vehicle emissions in California using the ISRM (*15*, *28*, *73*). The ISRM was developed from the US InMAP, which used WRF-Chem simulations and US Environmental Protection Agency National Emissions Inventory emissions estimates for 2014. The national version of InMAP was sampled on a population-weighted, variably sized grid (n = 21,705; 1 km to 48 km) for the state of California (*15*). Approximately 74% of grid cells are the finest resolution, with a population-weighted average grid size of 2.4 km (urban: 1.2 km; rural: 7.4 km). The gridding algorithm ensures that no cell larger than 1 km contains more than 20,000 people or a census block group with a population density higher than 2500 people/km.

The ISRM relates, for the n = 21,705 grid cells in California, marginal changes in ground-level concentration in every grid cell to marginal changes in emissions in every cell. Because this work only evaluates impacts from on-road mobile sources, all concentrations were estimated using the ground-level (i.e., 0 to 57 m above ground) layer.

Open-source tool: ECHO-AIR

Air pollution modeling, even with reduced complexity modeling tools such as InMAP, can have major accessibility barriers for non-specialists. For the present analysis, we developed an open-source Python-based pipeline that streamlines exposure concentration and health impact analyses. The resulting system, called Estimating

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Concentrations and Health Outcomes – Automated ISRM Resource (ECHO-AIR), aims to lower barriers to entry for rapid estimation of $PM_{2.5}$ exposure and health assessments.

Executing ECHO-AIR for analyses in California requires only estimates of emissions, which can be input as ArcGIS-compatible shapefiles or comma-separated value files. ECHO-AIR is modular, enabling users to use any ISRM, population data, and health input data, so long as they are formatted correctly ECHO-AIR is managed through a public GitHub repository to ensure transparency, maximize usability, and perform routine model upgrades and maintenance (see Supplementary Text for details).

Population estimates

We obtained population data for the years 2000 and 2010 from the decennial US census for California from the National Historic Geographic Information System (NHGIS) database version 16.0 (*91*). Population estimates were queried at the tract level by age, race, and Hispanic origin. Consistent with prior literature (*4*, *8*, *9*), racial-ethnic categories were estimated as follows: The population count for Hispanic Californians was defined as Californians of any race who were of Hispanic origin; Californians who are not of Hispanic origin and are Black or African American alone, Asian alone, or white alone were defined as Black, Asian, and white Californians, respectively; all other Californians were included in the other category.

Exposure assessment and disparity analysis

We estimated statewide group-level exposures to annual average $PM_{2.5}$ as PWM concentrations, consistent with the air pollution disparity literature (*3*, *8*, *9*, *19*). For the metrics below, we consider only on-road mobile source exposure (i.e., we neglect contributions from other source types unless explicitly stated otherwise). To estimate exposure to $PM_{2.5}$ for each year, we calculate geographic intersections between the 2010 census tract boundaries and the gridded concentration estimates. Population is down-sampled based on area apportionment; concentration estimates are assumed to be constant throughout the grid cell. Exposure concentrations are calculated at the smallest geography possible (e.g., polygon intersection of census tract and ISRM grid cell).

The PWM exposure is estimated by multiplying the annual average $PM_{2.5}$ concentration by the population of the demographic group of interest within that grid cell, summing across all grid cells, and dividing by the total population

$$\mathrm{PWM}_k = \frac{\sum_{i=1}^{n} P_{i,k} \times C_i}{\sum_{i=1}^{n} P_{i,k}}$$

where PWM_k is the PWM exposure concentration for group k across n grid cells, $P_{i,k}$ is the population of group k in grid cell i, and C_i is the concentration of $PM_{2.5}$ in grid cell i. Equity was assessed using the absolute and relative disparities at the PWM. The absolute disparity ($D_{A,k}$) is defined as a demographic group's PWM exposure (PWM_k) subtracted by the statewide PWM exposure (PWM_T)

$$D_{A,k} = \mathrm{PWM}_k - \mathrm{PWM}_T$$

Relative disparities ($D_{R,k}$) are estimated as the absolute disparity divided by the statewide PWM exposure to mobile sources

$$D_{R,k} = \frac{(\mathrm{PWM}_k - \mathrm{PWM}_T)}{\mathrm{PWM}_T} = \frac{D_{A,k}}{\mathrm{PWM}_T}$$

Because the ISRM is a linear model and the absolute disparity is an arithmetic equity metric, absolute disparities can be apportioned to individual source categories to find a relative contribution to the absolute disparity. Thus, the fractional contribution of a source's emissions to a group's exposure is estimated as follows

$$f_{j,k} = \frac{D_{A,j,k}}{D_{A,t,k}}$$

where $f_{j,k}$ is the fractional contribution from source j on the exposure and disparity for group k, $D_{A,j,k}$ is the absolute disparity from source j for group k, and $D_{A,t,k}$ is the absolute disparity for group k from all on-road mobile sources.

Supplementary Materials

This PDF file includes:

Supplementary Text
Figs. S1 to S18
Tables S1 to S6
References

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