

1 **COVID-19 LOCKDOWNS REVEAL PRONOUNCED**
2 **DISPARITIES IN NITROGEN DIOXIDE POLLUTION LEVELS**

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9 1. ABSTRACT

10 The unequal spatial distribution of ambient nitrogen dioxide (NO₂), an air pol-
11 lutant related to traffic, leads to higher exposure for minority and low socioeco-
12 nomic status communities. We exploit the unprecedented drop in urban activity
13 during the COVID-19 pandemic and use high-resolution, remotely-sensed NO₂
14 observations to investigate disparities in NO₂ levels across different demographic
15 subgroups in the United States. We show that COVID-19 lockdowns reduced,
16 but did not eliminate, the overall racial, ethnic, and socioeconomic NO₂ dispari-
17 ties. Prior to the pandemic, satellite-observed NO₂ levels in the least white census
18 tracts of the United States were double NO₂ levels in the most white tracts. Dur-
19 ing the pandemic, the largest lockdown-related NO₂ reductions occurred in urban
20 neighborhoods that have 30% fewer white residents and 111% more Hispanic resi-
21 dents than neighborhoods with the smallest reductions, likely driven by the greater
22 density of highways and interstates in these racially and ethnically diverse areas.
23 However, the least white tracts still experienced ~50% higher NO₂ levels during
24 the lockdowns than the most white tracts experienced prior to the pandemic. Fu-
25 ture policies aimed at eliminating pollution disparities will need to look beyond
26 reducing emissions from only passenger traffic and also consider other collocated
27 sources of emissions such as heavy-duty trucks, power plants, and industrial facil-
28 ities.

29 2. SIGNIFICANCE STATEMENT

30 We leverage the unparalleled changes in human activity during COVID-19 and
31 the unmatched capabilities of the TROPOspheric Monitoring Instrument to under-
32 stand how lockdowns impact ambient nitrogen dioxide (NO₂) pollution disparities

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33 in the United States. The least white communities experienced the largest NO₂
34 improvements during lockdowns; however, disparities between the least and most
35 white communities are so large that the least white communities still faced higher
36 NO₂ levels during lockdowns than the most white communities experienced prior
37 to lockdowns, despite a $\sim 50\%$ reduction in passenger vehicle traffic. Similar
38 findings hold for ethnic, income, and educational attainment subgroups. Future
39 strategies to reduce NO₂ disparities will need to target emissions from not only
40 passenger vehicles but other collocated on-road and stationary sources.

41

3. INTRODUCTION

42 Adverse air quality is an environmental justice issue as it disproportionately
43 affects lower income, minority, and marginalized populations around the world
44 [Bell and Ebisu, 2012, Landrigan et al., 2018, Schell et al., 2020]. Growing evidence
45 suggests that these populations experience more air pollution than is caused by
46 their consumption [Nguyen and Marshall, 2018, Tessum et al., 2019, Sergi et al.,
47 2020]. Within the United States (U.S.), disparities in exposure are persistent,
48 despite successful regulatory measures that have reduced pollution [Clark et al.,
49 2017, Colmer et al., 2020]. Nitrogen dioxide (NO₂) is a short-lived trace gas
50 formed shortly after fossil fuel combustion and regulated by the National Ambient
51 Air Quality Standards under the Clean Air Act. Exposure to NO₂ is associated
52 with a range of respiratory diseases and premature mortality [Jerrett et al., 2013,
53 Anenberg et al., 2018, Achakulwisut et al., 2019]. NO₂ is also a precursor to
54 other pollutants such as ozone and particulate matter [Stohl et al., 2015]. Major
55 sources of anthropogenic NO₂, such as roadways and industrial facilities, are often
56 located within or nearby minority and disenfranchised communities [Mohai et al.,
57 2009, Rowangould, 2013], and disparities in NO₂ exposure across demographic
58 subgroups have been the focus of several recent studies [Hajat et al., 2013, Clark
59 et al., 2014, Clark et al., 2017, Demetillo et al., 2020].

60 In early 2020, governments around the world imposed lockdowns and shelter-in-
61 place orders in response to the spread of the coronavirus disease 2019 (COVID-19).
62 The earliest government-mandated lockdowns in the U.S. began in California on
63 19 March 2020, and many states followed suit in the following days. Changes in
64 mobility patterns indicate that self-imposed social distancing practices were un-
65 derway days to weeks before the formal announcement of lockdowns [Badr et al.,
66 2020]. Lockdowns led to sharp reductions in surface-level NO₂ [He et al., 2020, Shi
67 and Brasseur, 2020, Venter et al., 2020] and tropospheric column NO₂ measured
68 from satellite instruments [Bauwens et al., 2020, Goldberg et al., 2020b] over the
69 U.S., China, and Europe. According to government-reported inventories, roughly
70 60% of anthropogenic emissions of nitrogen oxides (NO_x \equiv NO + NO₂) in the U.S.
71 in 2010 were emitted by on-road vehicles [US Environmental Protection Agency,
72 2015], and up to 80% of ambient NO₂ in urban areas can be linked to traffic
73 emissions [Levy et al., 2014, Sundvor et al., 2013]. As such, NO₂ is often used

74 as a marker for road traffic in urban areas. Multiple lines of evidence such as
75 seismic quieting and reduced mobility via location-based services point to changes
76 in traffic-related emissions as the main driver of drops in NO₂ pollution during
77 lockdowns due to the large proportion of the population working from home [Dif-
78 fenbaugh et al., 2020, Lecocq et al., 2020, Venter et al., 2020].

79 Here we exploit the unprecedented changes in human activity unique to the
80 COVID-19 lockdowns and remotely-sensed NO₂ columns with unprecedented spa-
81 tial resolution and coverage to understand inequalities in the distribution of NO₂
82 pollution for different racial, ethnic, and socioeconomic subgroups in the U.S.
83 Specifically, we address the following: Which demographic subgroups received the
84 largest NO₂ reductions? Did the lockdowns grow or shrink the perennial dis-
85 parities in NO₂ pollution across different demographic subgroups? Although the
86 lockdowns are economically unsustainable, how can they advance environmental
87 justice and equity by informing long-term policies to reduce NO₂ disparities and
88 the associated public health damages?

89

4. RESULTS

90 Previous studies examining satellite-derived NO₂ found the highest levels in
91 urban areas [Krotkov et al., 2017, Goldberg et al., 2020a], and we find that these
92 areas clearly stand out as NO₂ hotspots during our baseline period (Figure 1a).
93 NO₂ column densities averaged over all urban areas are a factor of two higher
94 than over rural areas during the baseline period. Absolute differences in NO₂
95 between the baseline and lockdown periods (“drops”) show sharp decreases over
96 virtually all major metropolitan regions (Figure 1b). Outside of metropolitan
97 areas, we note smaller NO₂ drops in Appalachia and the South, likely stemming
98 from a combination of lockdown-related changes in traffic emissions as well as
99 favorable weather [Goldberg et al., 2020b]. Parts of the Great Plains and Midwest
100 experience slight increases in NO₂ during lockdowns ($< 0.5 \times 10^{15}$ molecules cm⁻²),
101 which could reflect differences in natural (e.g., soil, lightning, stratospheric NO_x) or
102 anthropogenic sources of NO₂ between the baseline and lockdown periods. Given
103 that the largest lockdown-related changes in NO₂ occur in urban areas and to avoid
104 urban-rural demographic gradients, we primarily focus on urban NO₂ changes and
105 how these changes impact different demographic subgroups in urban areas.

106 The largest urban NO₂ drops occur in census tracts that are more non-white
107 and Hispanic and have a higher proportion of their population without a vehicle or
108 a post-secondary education compared with tracts with the smallest drops (Figure
109 1d-h). The percentage of white residents in tracts with the largest drops in NO₂
110 is 30% less compared with tracts with the smallest drops, which represent a slight
111 increase over baseline levels (Figure 1g). The percentage of Hispanic- or Latinx-
112 identifying residents in tracts with the largest drops is 111% larger than tracts with
113 the smallest drops (Figure 1d). This pattern found in urban tracts also holds in all

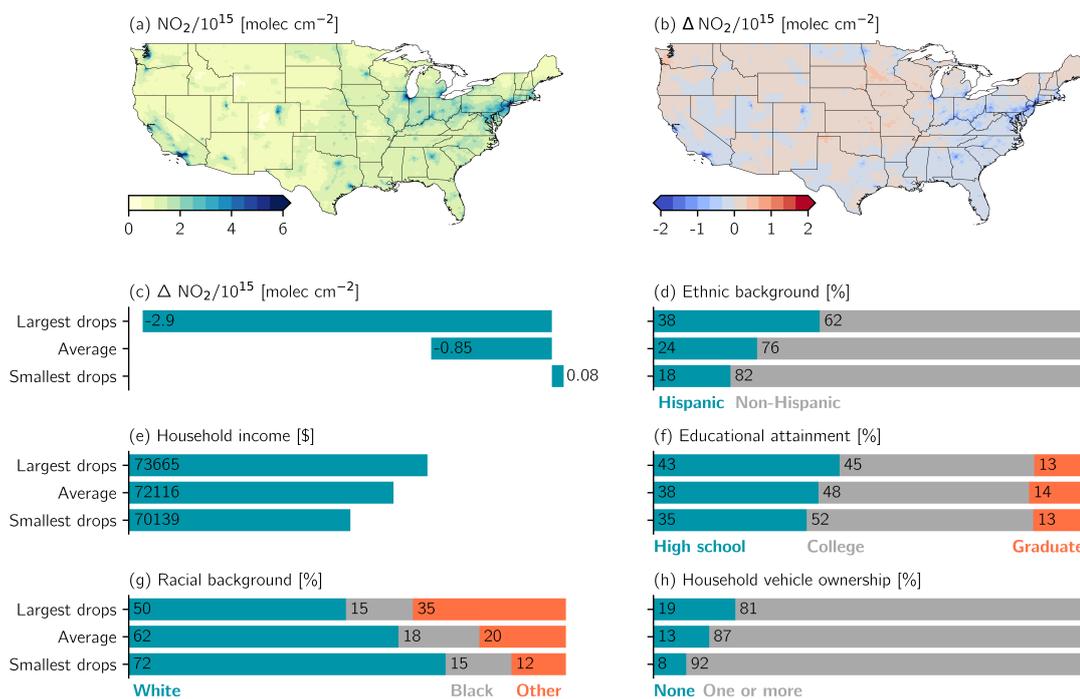


FIGURE 1. Spatial distribution of NO_2 columns during the baseline and COVID-19 lockdown periods and apportionment of drops among different demographic subgroups. (a) Census-tract average baseline NO_2 (13 March-13 June 2019). (b) Absolute difference between lockdown (13 March - 13 June 2020) and baseline NO_2 (ΔNO_2), where $\Delta \text{NO}_2 < 0$ corresponds to NO_2 drops during lockdowns. (c) Demographic data averaged over urban tracts with the largest drops (ΔNO_2 in first decile), all urban tracts, and urban tracts with the smallest drops (ΔNO_2 in the tenth decile).

114 (urban and rural) tracts and rural tracts, despite the different socio-demographic
 115 composition of the population in these areas (compare Figures 1 and S1).

116 Since less educated communities and communities with a large proportion of
 117 racial and ethnic minorities have faced higher levels of NO_2 and other pollutants
 118 for decades [Hajat et al., 2013, Hajat et al., 2015, Clark et al., 2017, Colmer et al.,
 119 2020, Schell et al., 2020], it is surprising that these communities experienced the
 120 largest drops in NO_2 pollution during COVID-19 lockdowns. However, Figure 1
 121 does not indicate how lockdown-related NO_2 drops grew or shrunk disparities, and
 122 we next examine disparities in baseline and lockdown NO_2 in the most advantaged
 123 versus disadvantaged census tracts in the U.S.

124 In the baseline period, low income, less educated neighborhoods and those with
125 a higher proportion of minority residents consistently face higher levels of NO_2
126 among all urban tracts across the U.S. and in nearly all 15 major metropolitan
127 statistical areas (MSAs) explored (Figure 2). An unexpected finding is that tracts
128 with the highest income and educational attainment in rural areas and aggregated
129 over both rural and urban areas have higher NO_2 levels than tracts with the
130 lowest income or educational attainment (Figure 2). When considering all census
131 tracts (both urban and rural), the most pronounced disparities are on the basis
132 of race and ethnicity: the least white tracts and most Hispanic tracts have 2.1
133 and 1.9 times greater baseline NO_2 levels than the most white and least Hispanic
134 tracts, respectively (Figure 2a, S2g). These disparities persist when examining the
135 individual MSAs in the U.S. For example, baseline NO_2 in tracts with the lowest
136 median household income in New York and Los Angeles is 1.6 and 1.7 times higher,
137 respectively, than tracts with the highest income (Figure 2b).

138 The unprecedented change in human activity during COVID-19 lockdowns nar-
139 rowed disparities in NO_2 across demographic subgroups in the U.S. (Figures 2,
140 S2). The ratio of NO_2 in the least white urban tracts to NO_2 in the most white
141 urban tracts in the U.S. decreased from 1.51 prior to the lockdowns to 1.36 during
142 the lockdowns (Figure 2a). Individual MSAs such as New York, Los Angeles, and
143 Atlanta undergo even more striking reductions in their racial, income, and edu-
144 cational attainment disparities. There are some cities or aggregations, however,
145 where disparities remain constant or even grow during lockdowns. As examples:
146 the ratio of NO_2 in all urban tracts with the lowest income to those with the high-
147 est income grows from unity prior to the lockdowns to 1.06 during the lockdowns
148 (Figure 2b), and the magnitude of disparities across demographic subgroups is
149 relatively constant in Phoenix (Figure 2).

150 Although the short-term changes in NO_2 during lockdowns reduced disparities,
151 the most exposed demographic subgroups prior to the lockdowns remained so dur-
152 ing the lockdowns (Figure 2, S2). For example, the racial disparities were so large
153 during the baseline period that even the unprecedented drop in human activity dur-
154 ing lockdowns did not bring NO_2 levels for the least white tracts down to the levels
155 experienced by the most white tracts prior to the lockdowns. The same patterns
156 hold true on the bases of ethnicity, income, and educational attainment (Figures
157 2, S2). These results are neither an artifact of how we defined demographic sub-
158 groups (Figure S2) or the precise time period over which we characterize disparities
159 (Figure S3).

160 Within urban areas, we find that the magnitude of NO_2 drops is tightly coupled
161 to the density of nearby primary roads (highways and interstates). The density
162 of primary roads in urban tracts with the largest NO_2 drops is six times greater
163 than in urban tracts with the smallest NO_2 drops (Figure 3). The racial, ethnic,
164 income, and educational composition of tracts are also closely related to primary
165 road density; urban tracts with lower income and vehicle ownership and a larger

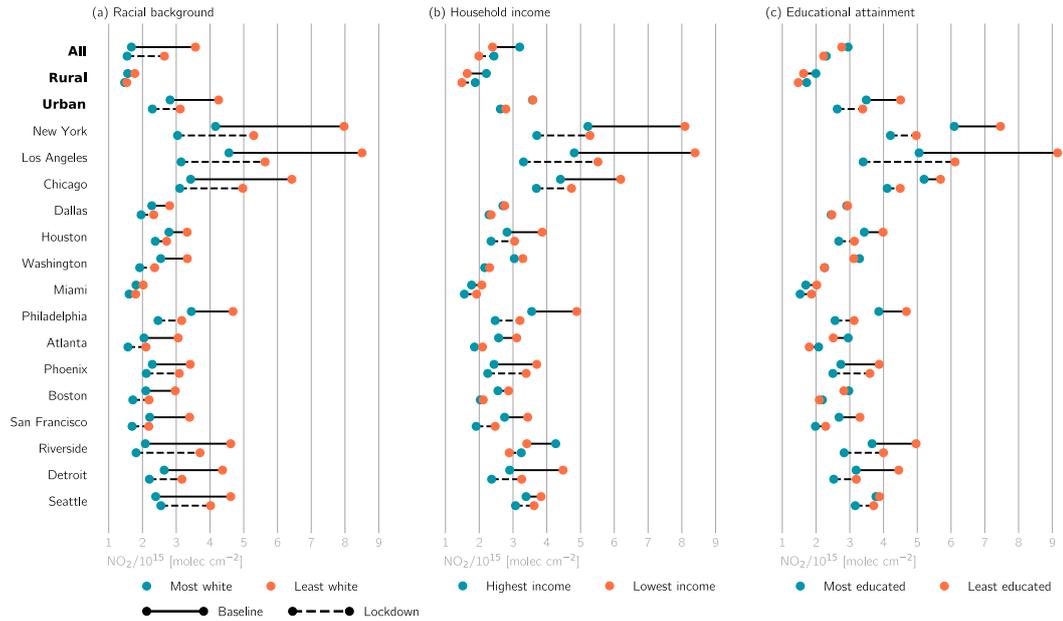


FIGURE 2. Disparities in baseline and lockdown NO_2 columns across different demographic subgroups. Subgroups are determined by identifying census tracts with extreme values for each demographic variable, and NO_2 levels are averaged over all, rural, and urban tracts with these extreme values. Urban tracts are further separated into the fifteen largest MSAs listed on the vertical axis.

166 percentage of racial and ethnic minorities are located near a higher density of
 167 primary roads (Figure 3). The difference in primary road density on the basis
 168 of vehicle ownership is especially stark: tracts with the lowest vehicle ownership
 169 (i.e., tracts in the first decile) have ~ 9.5 times higher primary road density than
 170 tracts with the highest ownership (i.e., tenth decile). Similarly, the least white
 171 tracts have a primary road density ~ 4.5 times higher than the most white tracts.
 172 Educational attainment is the only demographic variable considered in this study
 173 that exhibits a different relationship with primary road density, and we observe a
 174 U-shaped relationship between these variables (Figure 3).

175 To better understand the impact of the lockdowns on NO_2 exposure disparities,
 176 we consider case studies of individual cities: New York, Detroit, and Atlanta (Fig-
 177 ure 4). Among individual neighborhoods in each of these cities, the magnitude of
 178 NO_2 drops vary up to 50% above and below the citywide average (Figure 4a-c).
 179 The portions of New York, Atlanta, and Detroit that received the largest drops
 180 tend to have lower median household income and a high percentage of non-white
 181 residents (Figure 4d-i). In New York the largest drops are concentrated in Harlem

182 and The South Bronx (Figure 4a), where the high concentration of major high-
183 ways and industrial facilities has been linked to disproportionate exposure to air
184 pollution [Patel et al., 2009]. The largest drops in Atlanta occur in the southwest-
185 ern part of the city where median household income generally is $< \$30000$ and
186 the percentage of Black residents in each tract is nearly 100. Although large-scale
187 drops in NO_2 are primarily driven by reductions in on-road emissions [Quéré et al.,
188 2020, Venter et al., 2020], examining drops on smaller spatial scales, such as in
189 Atlanta (Figure 4b), suggests that emissions from other sectors may be at play. In
190 Atlanta, the largest drops occur southwest of downtown, near Hartsfield-Jackson
191 International Airport and several major highways (Figure 4b). The airport re-
192 ported a $\sim 50\%$ decrease in the daily number of flights during lockdowns [Shah,
193 2020]. Therefore, both on-road and aviation emissions may be responsible for the
194 disparities in NO_2 levels in Atlanta. The largest drops in Detroit are concentrated
195 on the west shores of the Detroit River; Interstates 75 and 94 and the Ambassador
196 Bridge, one of the busiest U.S.-Canada border crossing, transect this part of De-
197 troit (Figure 4c) [Martenies et al., 2017]. Although these Detroit neighborhoods
198 are not predominantly non-white (Figure 4f), they are home to a large Hispanic
199 population (not shown) with low median household income (Figure 4i).

200

5. DISCUSSION

201 Our results reveal that neighborhoods with a large population of racial and eth-
202 nic minorities, lower income, and lower educational attainment saw improvements
203 in NO_2 pollution during the COVID-19 lockdowns. In many cases, though, NO_2
204 disparities during the baseline period were so large that disadvantaged communi-
205 ties faced higher NO_2 levels during the lockdowns than advantaged communities
206 experienced prior to the lockdowns. Overall, these findings are consistent with
207 contemporaneous studies that have analyzed long-term trends in NO_2 and other
208 air pollutants and found that, despite widespread decreases in pollution, the most
209 exposed demographic subgroups in the 1980s and 1990s remain the most exposed
210 in the present-day [Clark et al., 2017, Colmer et al., 2020].

211 Disparities for certain spatial aggregations or for particular demographic vari-
212 ables deviate from the overall conclusions of this study. As an example, median
213 household income is $\sim \$3000$ higher in urban tracts with the largest drops com-
214 pared with those with the smallest drops, which may be counterintuitive given
215 the lower educational attainment (Figure 1e-f). Hajat et al. [Hajat et al., 2015]
216 found higher concentrations of particulate matter and NO_x in neighborhoods with
217 higher socioeconomic status in some North American cities. They posited that
218 busy roadways often run along rivers and lakes, and higher socioeconomic status
219 individuals may choose to live near these features for more scenic views and access
220 to urban amenities. We also find higher baseline NO_2 for the most white and most
221 educated tracts when considering all census tracts and only rural tracts (Figure
222 2b-c). A possible explanation for this may be that white, educated subpopulations

223 choose to live in suburban areas outside the census-designed urban boundaries but
 224 within the polluted airshed of the city.

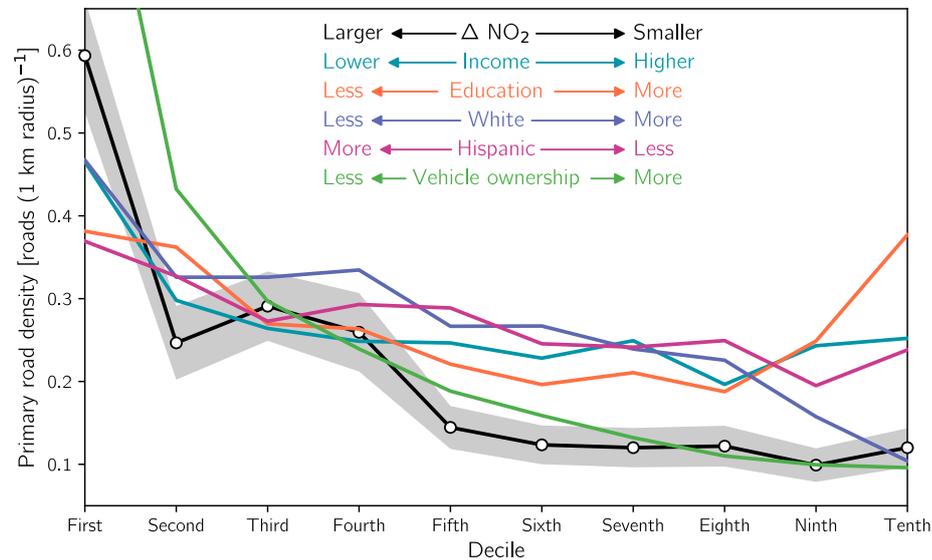


FIGURE 3. The relationship of road density with urban lockdown-related drops in NO_2 columns and demographic variables. Road density is calculated as the number of primary road segments within a 1 km radius of tracts’ centroids for each decile of demographic variables. The colored legend indicates the directionality of each demographic variable. As an example, the density corresponding to the lowest decile of the “White” curve represents the road density in urban tracts that are the least white (i.e. in the first decile of the percentage of their population that is white). Shading for the ΔNO_2 curve indicates the 90% confidence interval.

225 Tracts’ proximities to roadways may be responsible for both the lockdown-
 226 related drops and the persistent disparities of NO_2 pollution among demographic
 227 subgroups (Figures 1-3). The collocation of primary roads with poor, minority
 228 communities is not happenstance but a consequence of the Eisenhower-era federal
 229 highway program, which often deliberately routed highways through these poor,
 230 minority neighborhoods [Rose and Mohl, 2012, Boehmer et al., 2013, Rowangould,
 231 2013, Clark et al., 2017]. Additionally, other potent sources of pollution such as
 232 power plants, manufacturing facilities, and heavy-duty trucking operations are also
 233 collocated with primary roads due to these industries’ needs for highway access
 234 [Mohai et al., 2009, Demetillo et al., 2020].

235 Interestingly, urban tracts with the lowest vehicle ownership have both the high-
236 est density of nearby primary roads and the largest drops in NO_2 (Figures 1h, 3).
237 This result suggests that these communities may breathe more traffic-related NO_2
238 pollution than they produce. This is indeed the case for particulate matter pol-
239 lution: recent work found that particulate matter exposure is disproportionately
240 caused by rich, non-Hispanic white communities, while poor, Black and Hispanic
241 communities face higher exposure than is caused by their own consumption [Tes-
242 sum et al., 2019, Sergi et al., 2020].

243 Preliminary research suggests that high levels of NO_2 pollution contribute to
244 underlying health conditions that lead to increased COVID-19 fatality rates [Liang
245 et al., 2020]. Therefore, the decrease in NO_2 in low income or ethnicity and
246 racially diverse communities (Figure 2) could decrease population susceptibility
247 to COVID-19. This is especially important as these communities have increased
248 risk to COVID-19 and higher hospitalization rates [Raifman and Raifman, 2020].
249 Since short-term NO_2 exposure is associated with respiratory disease [Chauhan
250 et al., 2003, Hansel et al., 2015], the temporary NO_2 drops may have reduced
251 acute respiratory health outcomes, but the actual health effects of NO_2 drops
252 during the pandemic are difficult to tease out since the degree to which people
253 sought health care was also affected by the pandemic. These findings are especially
254 relevant for disadvantaged neighborhoods in cities (e.g., New York, Atlanta, and
255 Detroit; Figure 4) that have been long-plagued by high rates of asthma and other
256 respiratory diseases due, in part, to their proximity to on-road and point source
257 NO_x emissions [Patel et al., 2009, Martenies et al., 2017].

258 We have considered singular demographic variables and their relationship with
259 baseline and lockdown NO_2 . The case studies in Figure 4 hint that the intersec-
260 tionality between race and poverty may be associated with even more pronounced
261 lockdown-related drops in NO_2 pollution. Although the vast majority of tracts in
262 the southern half of Atlanta have a majority non-white population (Figure 4h),
263 the largest drops occur in tracts that are both majority non-white and low income
264 (Figure 4b, e, h). Recent work by Demetillo et al. [Demetillo et al., 2020] ex-
265 amined NO_2 exposure for Houston neighborhoods where poverty and racial and
266 ethnic identities intersect and found a disproportionate share of NO_2 pollution for
267 neighborhoods with these intersecting identities. Assessing other forms of intersec-
268 tionality and their relationship with air pollution exposure is a key area for future
269 research.

270 We relied on TROPOMI tropospheric column abundances rather than surface-
271 level concentrations to understand the impact of lockdowns on disparities in NO_2 .
272 Surface-level NO_2 concentrations inferred from satellites exist [Geddes et al., 2016,
273 Cooper et al., 2020], but not for 2020. Surface-level observations are sparse and
274 unevenly distributed in the U.S. [Lamsal et al., 2015]. TROPOMI provides sig-
275 nificant advances over predecessor instruments on account of its unprecedented

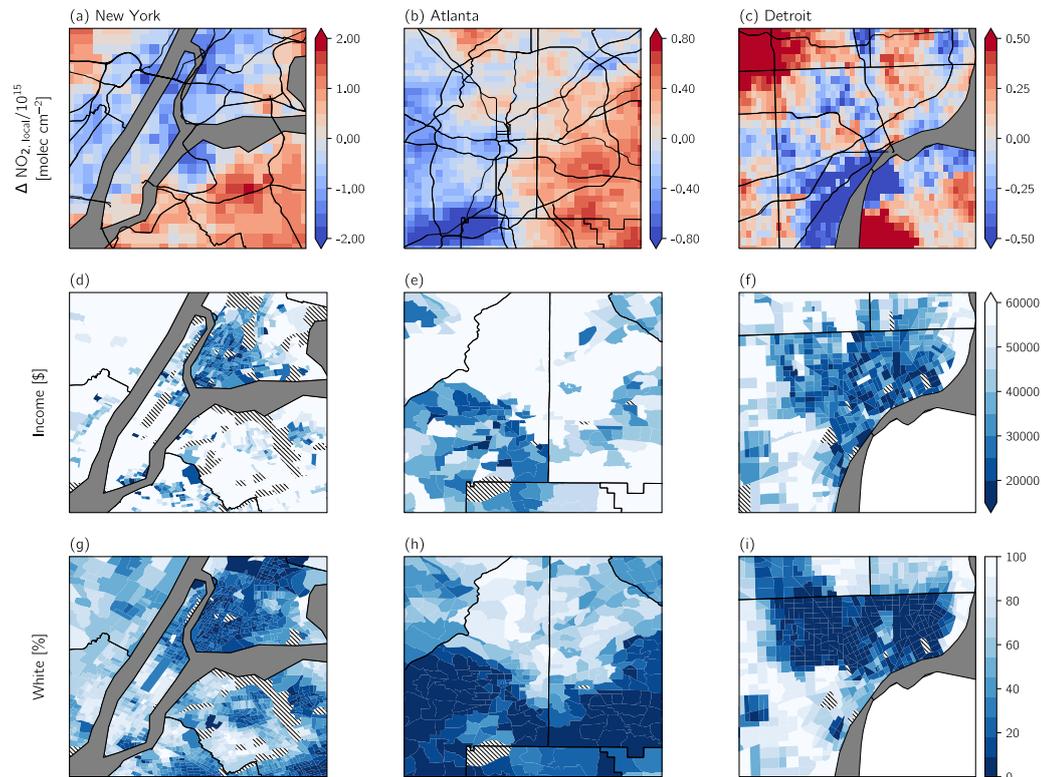


FIGURE 4. Case studies of lockdown NO_2 drops, income, and race for (left column) New York, (middle) Atlanta, and (right) Detroit. (a-c) $\Delta \text{NO}_{2, \text{local}}$ is calculated from oversampled TROPOMI data as the difference between ΔNO_2 and the city average ΔNO_2 to highlight neighborhoods with larger drops (i.e., negative values) and smaller drops (i.e., positive values) compared with the city-averaged drops. Primary roads are shown in thick black lines. (d-f) Median household income and (g-i) percentage of the population that is white. Tracts in (d-i) that are employment centers, airports, parks, or forests and therefore report no demographic data are denoted with hatching.

276 spatial resolution [Goldberg et al., 2019] and has been used for understanding eth-
 277 nic, racial, and socioeconomic status NO_2 disparities [Demetillo et al., 2020]. We
 278 tested whether TROPOMI has consistent spatial patterns with surface-level ob-
 279 servations and found good agreement (Figure S4a, Supporting Information Text).
 280 The ratios of 24-hour average NO_2 to NO_2 near the time of satellite overpass are
 281 also similar between least- and most-polluted sites (Figure S4b). These results sug-
 282 gest that column-to-surface or time-to-day biases do not underscore TROPOMI's

283 ability to capture disparities. Future work may infer surface concentrations of NO₂
284 from satellite-derived column abundances during lockdowns using these satellite
285 data within land-use regression models [Novotny et al., 2011] or chemical transport
286 models [Cooper et al., 2020]. We encourage the use of these ground-level estimates
287 to better understand exposure across demographic subgroups.

288 Lockdown-related changes in other air pollutants, particularly secondary pol-
289 lutants such as ozone and particulate matter, do not exhibit the same spatial
290 patterns as NO₂ [Chang et al., 2020, Shi and Brasseur, 2020, Venter et al., 2020].
291 Future research should investigate how changes in these species impact pollutant
292 disparities and environmental justice during lockdowns.

293

6. CONCLUSIONS

294 This study provides a unique look at air pollution disparities in the U.S., leverag-
295 ing the extraordinary confluence of unparalleled changes in human activity during
296 COVID-19 lockdowns and unmatched spatial coverage and resolution of air qual-
297 ity surveillance from the TROPOMI satellite instrument. Lockdowns decreased
298 tropospheric column abundances of NO₂ across the vast majority of urban areas.
299 However, drops in NO₂ pollution were uneven within these urban areas and largely
300 benefitted communities with a high proportion of racial and ethnic minorities and
301 lower educational attainment. Our results reveal that, despite the improvements
302 in NO₂ pollution during lockdowns, minority communities and communities with
303 lower income and educational attainment continued to face higher levels of NO₂
304 during the lockdowns than majority white communities and those with higher in-
305 come and educational attainment experienced prior to the pandemic. As traffic
306 emissions represent a major source of NO₂ variability, the proximity of disad-
307 vantaged neighborhoods to a high density of major roadways is likely the key
308 determinant in explaining lockdown-related drops in NO₂ pollution.

309 Our finding that even the $\sim 50\%$ drop in passenger vehicle emissions [Qu  r  
310 et al., 2020] did not reduce NO₂ levels among the most disadvantaged urban census
311 tracts to the levels experienced by the most advantaged tracts before the pandemic
312 indicates that profound changes are needed to address disparities in NO₂ pollution
313 in the U.S. In particular, this unintended natural experiment shows that policies
314 aimed at reducing emissions from passenger vehicle traffic (e.g., mode shifting to
315 public transportation and active transportation, widespread use of electric vehi-
316 cles) would not be enough. Policy strategies such as traffic rerouting and low
317 emissions zones [Nguyen and Marshall, 2018] and the widespread electrification of
318 heavy-, medium- and light-duty vehicles [Peters et al., 2020] are urgently needed.
319 Moreover, as stationary sources (e.g., power plants, industrial facilities) are often
320 collocated with major highways and interstates, emission control strategies that
321 reduce inequality in exposure while maximizing health benefits [Levy et al., 2007]
322 from these stationary sources should also be a key priority.

323

7. MATERIALS AND METHODS

324 **7.1. Remotely-sensed NO₂.** We obtain retrievals of the tropospheric NO₂ col-
325 umn from the Tropospheric Monitoring Instrument (TROPOMI) aboard the Sentinel-
326 5 Precursor (S5P) satellite. S5P is a nadir-viewing satellite in a sun-synchronous,
327 low-earth orbit that achieves near-global daily coverage with a local overpass time
328 of ~ 1330 hours [Veefkind et al., 2012]. TROPOMI provides NO₂ measurements at
329 an unprecedented spatial resolution of 5×3.5 km² (7×3.5 km² prior to 6 August
330 2019) [van Geffen et al., 2020]. Specifically, we use Level 2 data and only consider
331 pixels with a quality assurance value > 0.75 . Data are thereafter oversampled by
332 regriding to a standard grid with a resolution of 0.01° latitude \times 0.01° longitude
333 (~ 1 km \times 1 km) and averaged over two time periods: a baseline period (13 March-
334 13 June 2019) and a lockdown period (13 March-13 June 2020). Regrided data are
335 publicly available at Figshare (www.figshare.com/s/75a00608f3faedc4bca7).

336 Comparing the same time period across different years is commonplace in satel-
337 lite studies investigating changes in NO_x and other trace gases, and averaging over
338 three month timeframes smooths natural NO₂ variations that arise from differ-
339 ences in meteorology and sun angle, which are especially relevant during boreal
340 spring [Goldberg et al., 2020b]. This temporal averaging also removes part of the
341 random error in the TROPOMI single-pixel uncertainties, which can be 40-60% of
342 the tropospheric column abundances [Bauwens et al., 2020].

343 **7.2. Socio-demographic Data.** Demographic information is derived from the
344 American Community Survey (ACS) conducted by the U.S. Census Bureau and
345 maintained by the National Historical Geographic Information System [Manson
346 et al., 2019]. Data are publicly available at www.nhgis.org. We extract 2014-
347 2018 5-year estimates on race, Hispanic or Latino origin (henceforth “ethnicity”),
348 educational attainment, median household income, and vehicle availability for the
349 72,538 census tracts in the contiguous U.S. To minimize the number of differ-
350 ent categorical variables presented in this study, we combine racial groups into
351 three categories: white, Black (includes Black and African American), and Other
352 (includes American Indian/Alaska Native, Asian, Native Hawaiian/Other Pacific
353 Islander, and some other race). Similarly, we form three different levels for edu-
354 cational attainment: high school (includes no high school diploma, regular high
355 school diploma, and GED or alternative credentials), college (includes some col-
356 lege without a degree, Associate’s degree, and Bachelor’s degree), and graduate
357 (includes Master’s degree, Professional school degree, and Doctorate degree).

358 **7.3. Methods.** We harmonize the regrided TROPOMI NO₂ measurements with
359 tract-level ACS demographics by determining the geographic boundaries of each
360 tract and thereafter calculating a simple arithmetic average over all TROPOMI
361 grid cells within the tract for the baseline and lockdown periods. Approximately
362 8% of tracts lack a co-located TROPOMI grid cell due to their small size or irreg-
363 ular geometry, and we exclude these tracts from our analysis. Tracts are classified

364 as either rural or urban based on the census-designed rurality level from the last
365 decadal census in 2010. We further stratify the tracts into metropolitan-level sub-
366 sets for the 15 largest metropolitan statistical areas (MSAs) in the U.S.: New
367 York City-Newark-Jersey City, NY-NJ-PA; Los Angeles-Long Beach-Anaheim,
368 CA; Chicago-Naperville-Elgin, IL-IN-WI; Dallas-Fort Worth-Arlington, TX; Houston-
369 The Woodlands-Sugar Land, TX; Washington-Arlington-Alexandria, DC-VA-MD-
370 WV; Miami-Fort Lauderdale-Pompano Beach, FL; Philadelphia-Camden-Wilmington,
371 PA-NJ-DE-MD; Atlanta-Sandy Springs-Alpharetta, GA; Phoenix-Mesa-Chandler,
372 AZ; Boston-Cambridge-Newton, MA-NH; San Francisco-Oakland-Berkeley, CA;
373 Riverside-San Bernardino-Ontario, CA; Detroit-Warren-Dearborn, MI; and Seattle-
374 Tacoma-Bellevue, WA. For brevity we refer to these MSAs by their colloquial
375 names (e.g., Los Angeles, rather than Los Angeles-Long Beach-Anaheim, CA)
376 when discussing them.

377 We calculate the density of nearby primary roadways for each census tract as a
378 proxy for exposure to traffic-related NO_2 pollution. Primary roads are generally
379 divided, limited-access highways within the Interstate Highway System or under
380 state management, and their locations are determined from the U.S. Census Bu-
381 reau’s TIGER/Line geospatial database. Specifically, we determine density as the
382 number of primary road segments within 1 km of a tract’s centroid. We choose
383 1 km as our threshold for what constitutes as “nearby,” as NO_2 concentrations
384 decrease up to $\sim 50\%$ within 0.5 – 2 km from major roadways [Novotny et al.,
385 2011, Demetillo et al., 2020], and we note that our findings are robust when con-
386 sidering all primary roads within 2 km (not shown). Other means of quantifying
387 traffic exist (e.g., length of roadway within a specified distance, traffic within buffer
388 zones, sum of distance traveled) [Pratt et al., 2013], but our approach allows for
389 consistent use of geospatial data from the U.S. Census Bureau.

390 We partition census tracts by extreme values of their change in NO_2 (ΔNO_2)
391 or demographic variables using the first decile (0-10th percentile) and tenth decile
392 (90-100th percentile). As examples, tracts classified as “Most white” or “Highest
393 income” have a white population fraction or median household income which falls
394 in the tenth decile. Likewise, ΔNO_2 in tracts with the “Largest drops” (i.e., the
395 largest decrease in NO_2 during lockdowns) falls in the first decile. Our results are
396 not sensitive to the use of the first and tenth deciles, and we have tested the upper
397 and lower vigintiles, quintiles, and quartiles and obtain similar results (Figure S2).
398 The use of percentiles rather than absolute thresholds yields a consistent sample
399 size for the upper and lower extrema and also avoids defining absolute thresholds
400 for different variables. This is especially important as thresholds may change along
401 the urban-rural gradient or among different metropolitan areas.

402 The start date of the baseline and lockdowns periods used in this study (13
403 March) corresponds to the date of national emergency declaration in the U.S. and
404 the beginning of a pronounced decrease in mobility patterns in 2020 [Badr et al.,
405 2020]. Our results could be an artifact of the start date or length of the baseline

406 and lockdown periods. We test whether the overall racial, ethnic, income, and
407 educational disparities hold for other periods and find that the disparities among
408 different demographic subgroups persist regardless of the start date or length of the
409 baseline period (Figure S3). While the absolute NO₂ levels experienced by these
410 subgroups slightly change based on the baseline period, our overall results do not
411 hinge on the precise definition of the baseline period. We are inherently limited
412 by the short TROPOMI data record, and interannual variability could play a role
413 in modulating the magnitude of disparities in NO₂ levels. Testing this possibility
414 is important as more TROPOMI data become available.

415

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422

REFERENCES

- 423 [Achakulwisut et al., 2019] Achakulwisut, P., Brauer, M., Hystad, P., and Anenberg, S. C.
424 (2019). Global, national, and urban burdens of paediatric asthma incidence attributable to
425 ambient NO₂ pollution: estimates from global datasets. *The Lancet Planetary Health*, 3(4):e166–
426 e178.
- 427 [Anenberg et al., 2018] Anenberg, S. C., Henze, D. K., Tinney, V., Kinney, P. L., Raich, W.,
428 Fann, N., Malley, C. S., Roman, H., Lamsal, L., Duncan, B., Martin, R. V., van Donkelaar, A.,
429 Brauer, M., Doherty, R., Jonson, J. E., Davila, Y., Sudo, K., and Kuylenstierna, J. C. (2018).
430 Estimates of the global burden of ambient PM_{2.5}, ozone, and NO₂ on asthma incidence and
431 emergency room visits. *Environmental Health Perspectives*, 126(10):107004.
- 432 [Badr et al., 2020] Badr, H. S., Du, H., Marshall, M., Dong, E., Squire, M. M., and Gardner,
433 L. M. (2020). Association between mobility patterns and COVID-19 transmission in the USA:
434 a mathematical modelling study. *The Lancet Infectious Diseases*.
- 435 [Bauwens et al., 2020] Bauwens, M., Compernelle, S., Stavrou, T., Müller, J.-F., Gent, J.,
436 Eskes, H., Levelt, P. F., A, R., Veeffkind, J. P., Vlietinck, J., Yu, H., and Zehner, C. (2020).
437 Impact of coronavirus outbreak on NO₂ pollution assessed using TROPOMI and OMI obser-
438 vations. *Geophysical Research Letters*, 47(11).
- 439 [Bell and Ebisu, 2012] Bell, M. L. and Ebisu, K. (2012). Environmental inequality in exposures
440 to airborne particulate matter components in the United States. *Environmental Health Per-
441 spectives*, 120(12):1699–1704.
- 442 [Boehmer et al., 2013] Boehmer, T. K., Foster, S. L., Henry, J. R., Woghiren-Akinnifesi, E. L.,
443 and Yip, F. Y. (2013). Residential proximity to major highways - United States, 2010. *Morbidity
444 and Mortality Weekly Report*, 62(3):46–50.
- 445 [Chang et al., 2020] Chang, Y., Huang, R.-J., Ge, X., Huang, X., Hu, J., Duan, Y., Zou, Z.,
446 Liu, X., and Lehmann, M. F. (2020). Puzzling haze events in china during the coronavirus
447 (COVID-19) shutdown. *Geophysical Research Letters*, 47(12).
- 448 [Chauhan et al., 2003] Chauhan, A., Inskip, H. M., Linaker, C. H., Smith, S., Schreiber, J.,
449 Johnston, S. L., and Holgate, S. T. (2003). Personal exposure to nitrogen dioxide (NO₂) and
450 the severity of virus-induced asthma in children. *The Lancet*, 361(9373):1939–1944.
- 451 [Clark et al., 2014] Clark, L. P., Millet, D. B., and Marshall, J. D. (2014). National patterns in
452 environmental injustice and inequality: Outdoor NO₂ air pollution in the united states. *PLoS
453 ONE*, 9(4):e94431.
- 454 [Clark et al., 2017] Clark, L. P., Millet, D. B., and Marshall, J. D. (2017). Changes in
455 transportation-related air pollution exposures by race-ethnicity and socioeconomic status: out-
456 door nitrogen dioxide in the United States in 2000 and 2010. *Environmental Health Perspectives*,
457 125(9):097012.
- 458 [Colmer et al., 2020] Colmer, J., Hardman, I., Shimshack, J., and Voorheis, J. (2020). Disparities
459 in PM_{2.5} air pollution in the United States. *Science*, 369(6503):575–578.
- 460 [Cooper et al., 2020] Cooper, M. J., Martin, R. V., McLinden, C. A., and Brook, J. R. (2020).
461 Inferring ground-level nitrogen dioxide concentrations at fine spatial resolution applied to the
462 TROPOMI satellite instrument. *Environmental Research Letters*, 15(10):104013.
- 463 [Demetillo et al., 2020] Demetillo, M. A. G., Navarro, A., Knowles, K. K., Fields, K. P., Geddes,
464 J. A., Nowlan, C. R., Janz, S. J., Judd, L. M., Al-Saadi, J., Sun, K., McDonald, B. C., Diskin,
465 G. S., and Pusede, S. E. (2020). Observing nitrogen dioxide air pollution inequality using high-
466 spatial-resolution remote sensing measurements in houston, texas. *Environmental Science &
467 Technology*, 54(16):9882–9895.
- 468 [Diffenbaugh et al., 2020] Diffenbaugh, N. S., Field, C. B., Appel, E. A., Azevedo, I. L., Bal-
469 docchi, D. D., Burke, M., Burney, J. A., Ciais, P., Davis, S. J., Fiore, A. M., Fletcher, S. M.,

- 470 Hertel, T. W., Horton, D. E., Hsiang, S. M., Jackson, R. B., Jin, X., Levi, M., Lobell, D. B.,
471 McKinley, G. A., Moore, F. C., Montgomery, A., Nadeau, K. C., Pataki, D. E., Randerson,
472 J. T., Reichstein, M., Schnell, J. L., Seneviratne, S. I., Singh, D., Steiner, A. L., and Wong-
473 Parodi, G. (2020). The COVID-19 lockdowns: a window into the earth system. *Nature Reviews*
474 *Earth & Environment*, 1(9):470–481.
- 475 [Geddes et al., 2016] Geddes, J. A., Martin, R. V., Boys, B. L., and van Donkelaar, A. (2016).
476 Long-term trends worldwide in ambient NO₂ concentrations inferred from satellite observations.
477 *Environmental Health Perspectives*, 124(3):281–289.
- 478 [Goldberg et al., 2020a] Goldberg, D. L., Anenberg, S., Moheg, A., Lu, Z., and Streets, D. G.
479 (2020a). TROPOMI NO₂ in the United States: a detailed look at the annual averages, weekly
480 cycles, effects of temperature, and correlation with PM_{2.5}. *Unpublished results*.
- 481 [Goldberg et al., 2020b] Goldberg, D. L., Anenberg, S. C., Griffin, D., McLinden, C. A., Lu, Z.,
482 and Streets, D. G. (2020b). Disentangling the impact of the COVID-19 lockdowns on urban
483 NO₂ from natural variability. *Geophysical Research Letters*, 47(17).
- 484 [Goldberg et al., 2019] Goldberg, D. L., Lu, Z., Streets, D. G., de Foy, B., Griffin, D., McLinden,
485 C. A., Lamsal, L. N., Krotkov, N. A., and Eskes, H. (2019). Enhanced capabilities of TROPOMI
486 NO₂: Estimating NO_x from North American cities and power plants. *Environmental Science*
487 *& Technology*, 53(21):12594–12601.
- 488 [Hajat et al., 2013] Hajat, A., Diez-Roux, A. V., Adar, S. D., Auchincloss, A. H., Lovasi, G. S.,
489 O’Neill, M. S., Sheppard, L., and Kaufman, J. D. (2013). Air pollution and individual and
490 neighborhood socioeconomic status: Evidence from the multi-ethnic study of atherosclerosis
491 (MESA). *Environmental Health Perspectives*, 121(11-12):1325–1333.
- 492 [Hajat et al., 2015] Hajat, A., Hsia, C., and O’Neill, M. S. (2015). Socioeconomic disparities and
493 air pollution exposure: a global review. *Current Environmental Health Reports*, 2(4):440–450.
- 494 [Hansel et al., 2015] Hansel, N. N., McCormack, M. C., and Kim, V. (2015). The effects of
495 air pollution and temperature on COPD. *COPD: Journal of Chronic Obstructive Pulmonary*
496 *Disease*, 13(3):372–379.
- 497 [He et al., 2020] He, G., Pan, Y., and Tanaka, T. (2020). The short-term impacts of COVID-19
498 lockdown on urban air pollution in China. *Nature Sustainability*.
- 499 [Jerrett et al., 2013] Jerrett, M., Burnett, R. T., Beckerman, B. S., Turner, M. C., Krewski, D.,
500 Thurston, G., Martin, R. V., van Donkelaar, A., Hughes, E., Shi, Y., Gapstur, S. M., Thun,
501 M. J., and Pope, C. A. (2013). Spatial analysis of air pollution and mortality in California.
502 *American Journal of Respiratory and Critical Care Medicine*, 188(5):593–599.
- 503 [Krotkov et al., 2017] Krotkov, N. A., Lamsal, L. N., Celarier, E. A., Swartz, W. H., Marchenko,
504 S. V., Bucsela, E. J., Chan, K. L., Wenig, M., and Zara, M. (2017). The version 3 OMI no₂
505 standard product. *Atmospheric Measurement Techniques*, 10(9):3133–3149.
- 506 [Lamsal et al., 2015] Lamsal, L. N., Duncan, B. N., Yoshida, Y., Krotkov, N. A., Pickering, K. E.,
507 Streets, D. G., and Lu, Z. (2015). U.S. NO₂ trends (2005-2013): EPA Air Quality System (AQS)
508 data versus improved observations from the Ozone Monitoring Instrument (OMI). *Atmospheric*
509 *Environment*, 110:130–143.
- 510 [Landrigan et al., 2018] Landrigan, P. J., Fuller, R., Acosta, N. J. R., Adeyi, O., Arnold, R.,
511 Basu, N. N., Baldé, A. B., Bertollini, R., Bose-O’Reilly, S., Boufford, J. I., Breyse, P. N.,
512 Chiles, T., Mahidol, C., Coll-Seck, A. M., Cropper, M. L., Fobil, J., Fuster, V., Greenstone,
513 M., Haines, A., Hanrahan, D., Hunter, D., Khare, M., Krupnick, A., Lanphear, B., Lohani,
514 B., Martin, K., Mathiasen, K. V., McTeer, M. A., Murray, C. J. L., Ndahimananjara, J. D.,
515 Perera, F., Potočnik, J., Preker, A. S., Ramesh, J., Rockström, J., Salinas, C., Samson, L. D.,
516 Sandilya, K., Sly, P. D., Smith, K. R., Steiner, A., Stewart, R. B., Suk, W. A., van Schayck,
517 O. C. P., Yadama, G. N., Yumkella, K., and Zhong, M. (2018). The Lancet Commission on
518 pollution and health. *The Lancet*, 391(10119):462–512.

- 519 [Lecocq et al., 2020] Lecocq, T., Hicks, S. P., Noten, K. V., van Wijk, K., Koelemeijer, P., Plaen,
520 R. S. M. D., Massin, F., Hillers, G., Anthony, R. E., Apoloner, M.-T., Arroyo-Solórzano, M.,
521 Assink, J. D., Büyükakpınar, P., Cannata, A., Cannavo, F., Carrasco, S., Caudron, C., Chaves,
522 E. J., Cornwell, D. G., Craig, D., den Ouden, O. F. C., Diaz, J., Donner, S., Evangelidis,
523 C. P., Evers, L., Fauville, B., Fernandez, G. A., Giannopoulos, D., Gibbons, S. J., Girona, T.,
524 Grecu, B., Grunberg, M., Hetényi, G., Horleston, A., Inza, A., Irving, J. C. E., Jamalreyhani,
525 M., Kafka, A., Koymans, M. R., Labedz, C. R., Larose, E., Lindsey, N. J., McKinnon, M.,
526 Megies, T., Miller, M. S., Minarik, W., Moresi, L., Márquez-Ramírez, V. H., Möllhoff, M.,
527 Nesbitt, I. M., Niyogi, S., Ojeda, J., Oth, A., Proud, S., Pulli, J., Retailleau, L., Rintamäki,
528 A. E., Satriano, C., Savage, M. K., Shani-Kadmiel, S., Sleeman, R., Sokos, E., Stammer, K.,
529 Stott, A. E., Subedi, S., Sorensen, M. B., Taira, T., Tapia, M., Turhan, F., van der Pluijm,
530 B., Vanstone, M., Vergne, J., Vuorinen, T. A. T., Warren, T., Wassermann, J., and Xiao, H.
531 (2020). Global quieting of high-frequency seismic noise due to COVID-19 pandemic lockdown
532 measures. *Science*, 369(6509):1338–1343.
- 533 [Levy et al., 2014] Levy, I., Mihele, C., Lu, G., Narayan, J., and Brook, J. R. (2014). Evalu-
534 ating multipollutant exposure and urban air quality: pollutant interrelationships, neighbor-
535 hood variability, and nitrogen dioxide as a proxy pollutant. *Environmental Health Perspectives*,
536 122(1):65–72.
- 537 [Levy et al., 2007] Levy, J. I., Wilson, A. M., and Zwack, L. M. (2007). Quantifying the efficiency
538 and equity implications of power plant air pollution control strategies in the United States.
539 *Environmental Health Perspectives*, 115(5):743–750.
- 540 [Liang et al., 2020] Liang, D., Shi, L., Zhao, J., Liu, P., Sarnat, J. A., Gao, S., Schwartz, J., Liu,
541 Y., Ebelt, S. T., Scovronick, N., and Chang, H. H. (2020). Urban air pollution may enhance
542 COVID-19 case-fatality and mortality rates in the united states. *The Innovation*, 1(3):100047.
- 543 [Manson et al., 2019] Manson, S., Schroeder, J., Ripper, D. V., and Ruggles, S. (2019). National
544 historical geographic information system: Version 14.0.
- 545 [Martenies et al., 2017] Martenies, S., Milano, C., Williams, G., and Batterman, S. (2017).
546 Disease and health inequalities attributable to air pollutant exposure in detroit, michigan.
547 *International Journal of Environmental Research and Public Health*, 14(10):1243.
- 548 [Mohai et al., 2009] Mohai, P., Lantz, P. M., Morenoff, J., House, J. S., and Mero, R. P. (2009).
549 Racial and socioeconomic disparities in residential proximity to polluting industrial facili-
550 ties: Evidence from the americans changing lives study. *American Journal of Public Health*,
551 99(S3):S649–S656.
- 552 [Nguyen and Marshall, 2018] Nguyen, N. P. and Marshall, J. D. (2018). Impact, efficiency, in-
553 equality, and injustice of urban air pollution: variability by emission location. *Environmental*
554 *Research Letters*, 13(2):024002.
- 555 [Novotny et al., 2011] Novotny, E. V., Bechle, M. J., Millet, D. B., and Marshall, J. D. (2011).
556 National satellite-based land-use regression: NO₂ in the united states. *Environmental Science*
557 *& Technology*, 45(10):4407–4414.
- 558 [Patel et al., 2009] Patel, M. M., Chillrud, S. N., Correa, J. C., Feinberg, M., Hazi, Y., Deepthi,
559 K., Prakash, S., Ross, J. M., Levy, D., and Kinney, P. L. (2009). Spatial and temporal variations
560 in traffic-related particulate matter at new york city high schools. *Atmospheric Environment*,
561 43(32):4975–4981.
- 562 [Peters et al., 2020] Peters, D. R., Schnell, J. L., Kinney, P. L., Naik, V., and Horton, D. E.
563 (2020). Public health and climate benefits and trade-offs of U.S. vehicle electrification. *Geo-*
564 *Health*, 4(10).
- 565 [Pratt et al., 2013] Pratt, G. C., Parson, K., Shinoda, N., Lindgren, P., Dunlap, S., Yawn, B.,
566 Wollan, P., and Johnson, J. (2013). Quantifying traffic exposure. *Journal of Exposure Science*
567 *& Environmental Epidemiology*, 24(3):290–296.

- 568 [Quéré et al., 2020] Quéré, C. L., Jackson, R. B., Jones, M. W., Smith, A. J. P., Abernethy,
569 S., Andrew, R. M., De-Gol, A. J., Willis, D. R., Shan, Y., Canadell, J. G., Friedlingstein, P.,
570 Creutzig, F., and Peters, G. P. (2020). Temporary reduction in daily global CO₂ emissions
571 during the COVID-19 forced confinement. *Nature Climate Change*, 10(7):647–653.
- 572 [Raifman and Raifman, 2020] Raifman, M. A. and Raifman, J. R. (2020). Disparities in the
573 population at risk of severe illness from COVID-19 by race/ethnicity and income. *American*
574 *Journal of Preventive Medicine*, 59(1):137–139.
- 575 [Rose and Mohl, 2012] Rose, M. H. and Mohl, R. A. (2012). *Interstate: Highway Politics and*
576 *Policy since 1939*. The University of Tennessee Press, Knoxville, 3rd edition.
- 577 [Rowangould, 2013] Rowangould, G. M. (2013). A census of the US near-roadway population:
578 Public health and environmental justice considerations. *Transportation Research Part D: Trans-*
579 *port and Environment*, 25:59–67.
- 580 [Schell et al., 2020] Schell, C. J., Dyson, K., Fuentes, T. L., Roches, S. D., Harris, N. C., Miller,
581 D. S., Woelfle-Erskine, C. A., and Lambert, M. R. (2020). The ecological and evolutionary
582 consequences of systemic racism in urban environments. *Science*, 369(6510):eaay4497.
- 583 [Sergi et al., 2020] Sergi, B., Azevedo, I., Davis, S. J., and Muller, N. Z. (2020). Regional
584 and county flows of particulate matter damage in the US. *Environmental Research Letters*,
585 15(10):104073.
- 586 [Shah, 2020] Shah, K. (2020). 'Mostly empty': Covid-19 has nearly shut down world's busiest
587 airport. Accessed October 15, 2020.
- 588 [Shi and Brasseur, 2020] Shi, X. and Brasseur, G. P. (2020). The response in air quality to the
589 reduction of Chinese economic activities during the COVID-19 outbreak. *Geophysical Research*
590 *Letters*, 47(11).
- 591 [Stohl et al., 2015] Stohl, A., Aamaas, B., Amann, M., Baker, L. H., Bellouin, N., Berntsen,
592 T. K., Boucher, O., Cherian, R., Collins, W., Daskalakis, N., Dusinska, M., Eckhardt, S.,
593 Fuglestvedt, J. S., Harju, M., Heyes, C., Hodnebrog, Ø., Hao, J., Im, U., Kanakidou, M.,
594 Klimont, Z., Kupiainen, K., Law, K. S., Lund, M. T., Maas, R., MacIntosh, C. R., Myhre, G.,
595 Myriokefalitakis, S., Olivíe, D., Quaas, J., Quennehen, B., Raut, J.-C., Rumbold, S. T., Samset,
596 B. H., Schulz, M., Seland, Ø., Shine, K. P., Skeie, R. B., Wang, S., Yttri, K. E., and Zhu, T.
597 (2015). Evaluating the climate and air quality impacts of short-lived pollutants. *Atmospheric*
598 *Chemistry and Physics*, 15(18):10529–10566.
- 599 [Sundvor et al., 2013] Sundvor, I., Balaguer, N. C., Viana, M., Querol, X., Reche, C., Amato,
600 F., Mellios, G., and Guerreiro, C. (2013). Road traffic's contribution to air quality in Euro-
601 pean cities. Technical report, European Topic Centre for Air Pollution and Climate Change
602 Mitigation.
- 603 [Tessum et al., 2019] Tessum, C. W., Apte, J. S., Goodkind, A. L., Muller, N. Z., Mullins, K. A.,
604 Paoletta, D. A., Polasky, S., Springer, N. P., Thakrar, S. K., Marshall, J. D., and Hill, J. D.
605 (2019). Inequity in consumption of goods and services adds to racial-ethnic disparities in air
606 pollution exposure. *Proceedings of the National Academy of Sciences*, 116(13):6001–6006.
- 607 [US Environmental Protection Agency, 2015] US Environmental Protection Agency
608 (2015). 2014 National Emissions Inventory (NEI) data. [https://www.epa.gov/
609 air-emissions-inventories/2014-national-emissions-inventory-nei-data](https://www.epa.gov/air-emissions-inventories/2014-national-emissions-inventory-nei-data). Accessed
610 October 15, 2020.
- 611 [van Geffen et al., 2020] van Geffen, J., Boersma, K. F., Eskes, H., Sneep, M., ter Linden, M.,
612 Zara, M., and Veefkind, J. P. (2020). S5P TROPOMI NO₂ slant column retrieval: method,
613 stability, uncertainties and comparisons with OMI. *Atmospheric Measurement Techniques*,
614 13(3):1315–1335.
- 615 [Veefkind et al., 2012] Veefkind, J., Aben, I., McMullan, K., Förster, H., de Vries, J., Otter,
616 G., Claas, J., Eskes, H., de Haan, J., Kleipool, Q., van Weele, M., Hasekamp, O., Hoogeveen,

- 617 R., Landgraf, J., Snel, R., Tol, P., Ingmann, P., Voors, R., Kruizinga, B., Vink, R., Visser,
618 H., and Levelt, P. (2012). TROPOMI on the ESA Sentinel-5 Precursor: a GMES mission for
619 global observations of the atmospheric composition for climate, air quality and ozone layer
620 applications. *Remote Sensing of Environment*, 120:70–83.
- 621 [Venter et al., 2020] Venter, Z. S., Aunan, K., Chowdhury, S., and Lelieveld, J. (2020). COVID-
622 19 lockdowns cause global air pollution declines. *Proceedings of the National Academy of Sci-*
623 *ences*, 117(32):18984–18990.

1 SUPPORTING INFORMATION FOR “COVID-19 LOCKDOWNS
2 REVEAL PRONOUNCED DISPARITIES IN NITROGEN
3 DIOXIDE POLLUTION LEVELS”

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10 1. REMOTELY-SENSED VERSUS SURFACE-LEVEL NO₂

11 We compare tropospheric column NO₂ from TROPOMI with ground-based ob-
12 servations from the Environmental Protection Agency’s Air Quality System (AQS)
13 [US Environmental Protection Agency, nda] to test whether TROPOMI can pro-
14 vide an accurate characterization of differences in surface-level NO₂. There are
15 439 AQS monitors in the contiguous U.S. with observations during the baseline
16 period, and we average hourly observations over the entire baseline period at each
17 of these sites and compare it with TROPOMI retrievals at the collocated grid cell
18 to each site.

19 TROPOMI struggles to capture large, localized sources of NO₂ on account of
20 the difference in scale between the footprint of the satellite and point-based obser-
21 vations [Judd et al., 2019]. We find that 71 of the 439 monitors are located near
22 (< 20 meters) roads [US Environmental Protection Agency, ndb]. These sites gen-
23 erally have observed surface-level NO₂ > 10 ppbv despite relatively low columnar
24 amounts from TROPOMI (Figure 4). When we consider only AQS monitors that
25 are not located near roads, we find good agreement between TROPOMI and AQS
26 NO₂ levels (Figure 4a). We also find a similar ratio of NO₂ averaged over the
27 24-hour diurnal cycle to NO₂ near the time of satellite overpass at sites that are
28 classified as the most and least polluted (Figure 4b). These findings lend credi-
29 bility to our reliance on TROPOMI to characterize disparities in NO₂ at earth’s
30 surface.

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REFERENCES

- 32 [Judd et al., 2019] Judd, L. M., Al-Saadi, J. A., Janz, S. J., Kowalewski, M. G., Pierce, R. B.,
33 Szykman, J. J., Valin, L. C., Swap, R., Cede, A., Mueller, M., Tiefengraber, M., Abuhas-
34 san, N., and Williams, D. (2019). Evaluating the impact of spatial resolution on tropospheric
35 NO₂ column comparisons within urban areas using high-resolution airborne data. *Atmospheric*
36 *Measurement Techniques*, 12(11):6091–6111.
- 37 [US Environmental Protection Agency, nda] US Environmental Protection Agency (n.d.a). Air
38 Quality System Data Mart. <https://www.epa.gov/airdata>. Accessed October 21, 2020.
- 39 [US Environmental Protection Agency, ndb] US Environmental Protection Agency (n.d.b).
40 Near-road NO₂ monitoring. <https://www3.epa.gov/ttnamti1/nearroad.html>. Accessed Oc-
41 tober 22, 2020.

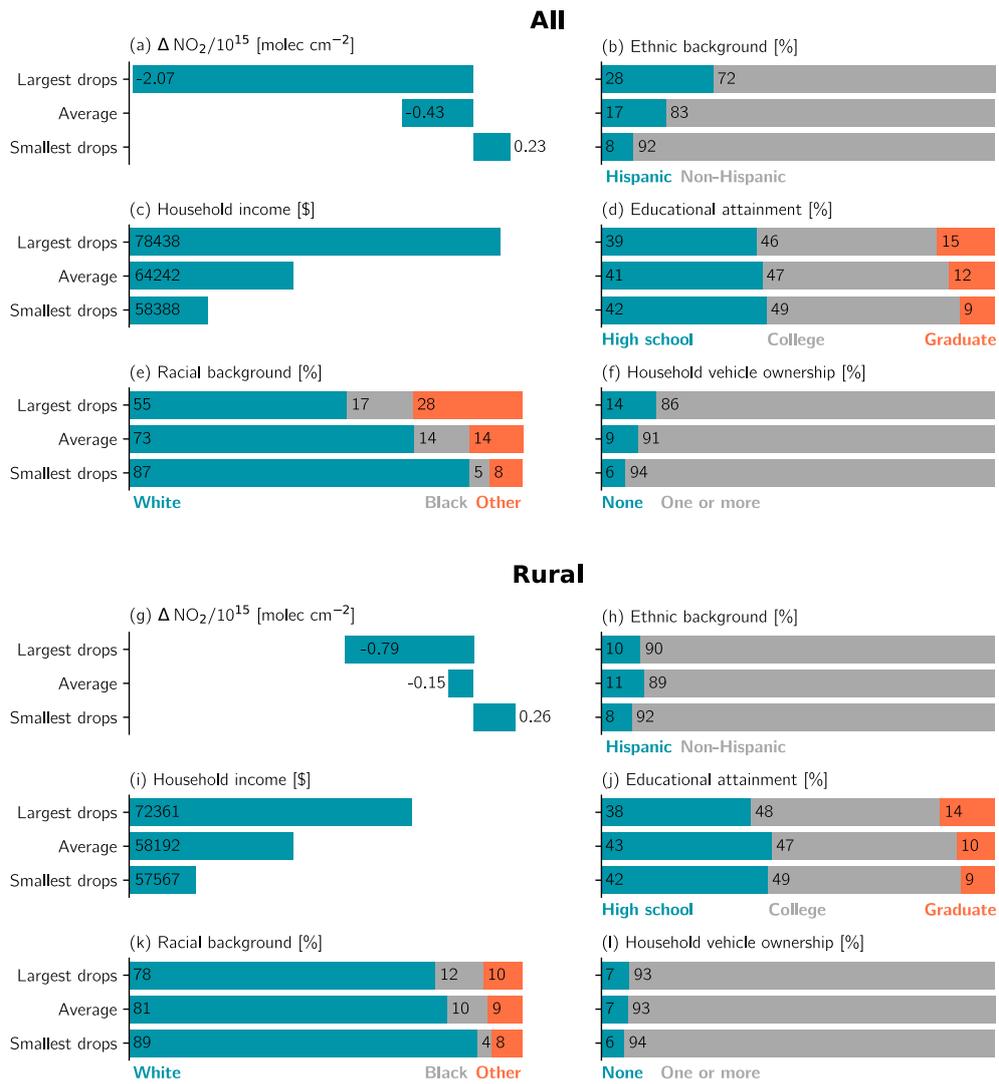


FIGURE 1. Same as Figure 1c-h in the main text but drops and averages are derived from (a-f) all tracts and (g-l) rural tracts.

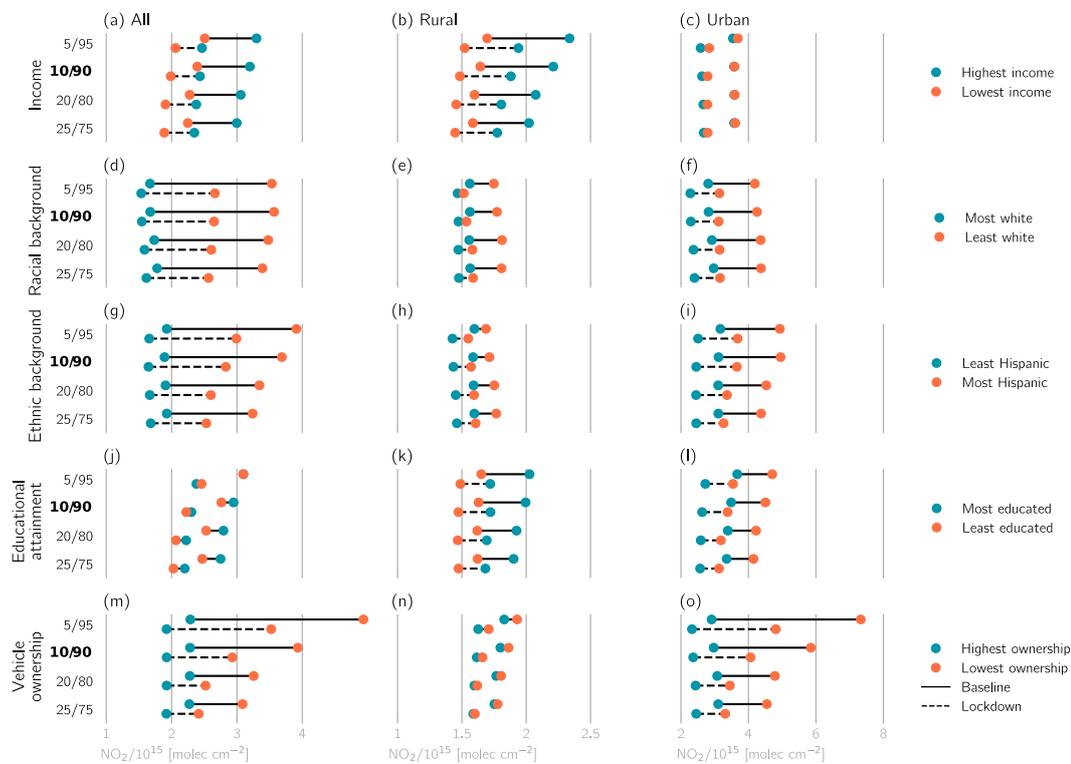


FIGURE 2. **Sensitivity of NO_2 disparities to percentiles chosen to constitute extreme values for each demographic variable.** Interpretation follows Figure 2 in the main text, but each pair of bars in individual subplots represents different percentile thresholds, indicated in the subplots' vertical axes. The boldface 10/90 row corresponds to the first and tenth deciles used in the main text.

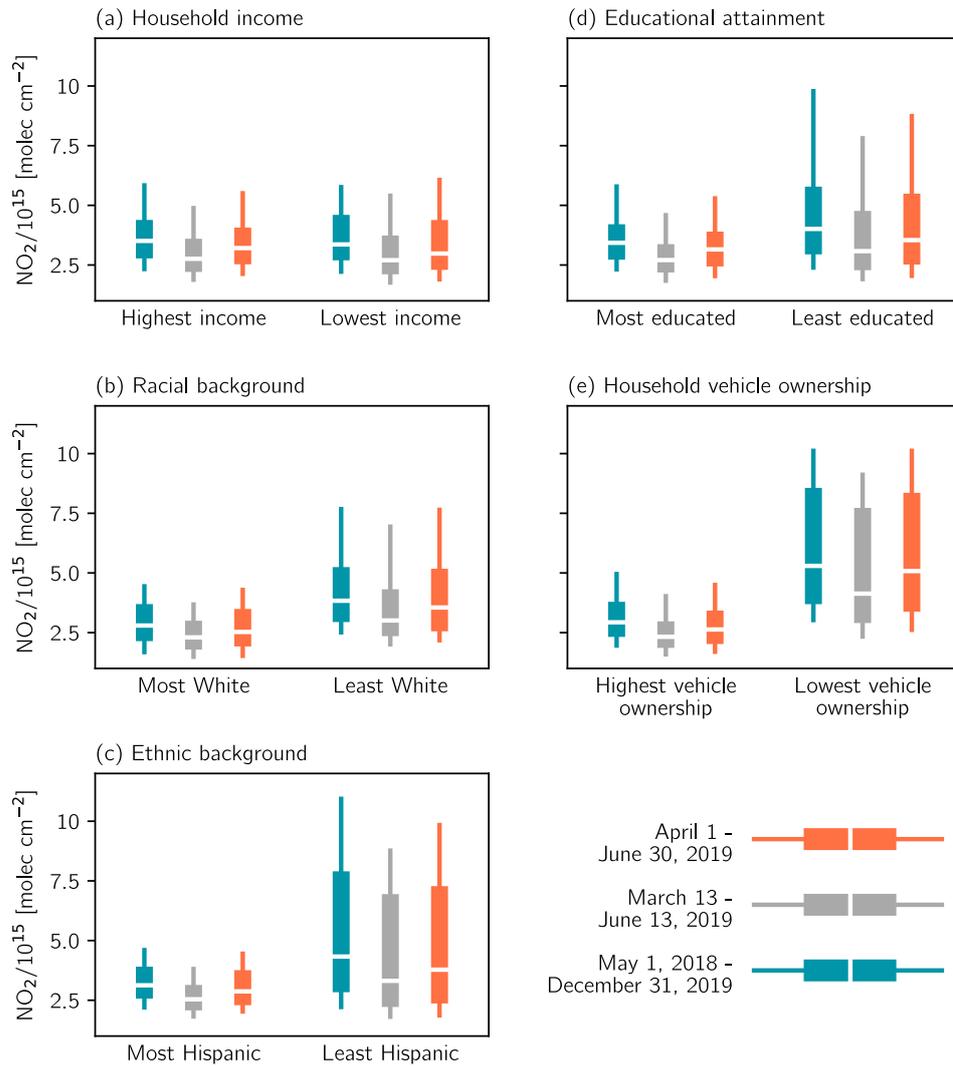


FIGURE 3. Sensitivity of urban NO_2 disparities to the baseline period. Extreme values of each demographic variable (using the first and tenth deciles) for three different baseline periods: 1 April - 30 June 2019, 13 March - 13 June 2019 (the period used in the main text), and 1 May 2018 - 31 December 2019 (the entire TROPOMI data record). Boxes extend to the lower and upper quartiles of the data, and the median value is indicated with the horizontal white lines. The lower and upper whiskers extend to the 10th and 90th percentiles, respectively.

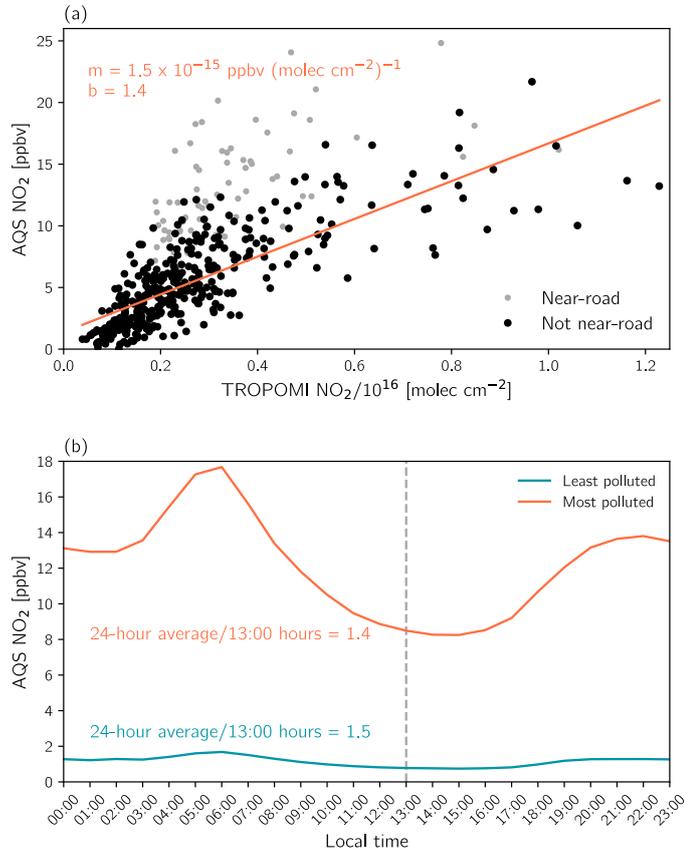


FIGURE 4. (a) Observed NO₂ from AQS monitors versus TROPOMI tropospheric NO₂ columns for the baseline period (13 March - 13 June 2019). TROPOMI data correspond to the nearest 0.01° latitude × 0.01° longitude grid cell to each AQS monitor. The orange line represents the linear regression fitted only through AQS data not flagged as “near-road” (< 20 meters). The orange text gives the slope (m) and intercept (b) of this linear fit. (b) Observed diurnal cycles of NO₂ averaged over the most polluted (AQS monitors where the collocated TROPOMI grid cell > 90th percentile) and least polluted sites (AQS monitors where the collocated TROPOMI grid cell < 10th percentile) during the baseline period. Only sites that are not near-road are considered for these averages. The ratios of 24-hour average NO₂ to NO₂ at the approximate time of satellite overpass (dashed grey line; ~ 13:00 hours local time) are indicated in the colored text.