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Perspective

Climate mitigation policies for cities must consider air quality impacts

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SUMMARY

Air pollutants are coemitted with carbon dioxide (CO_2) during the combustion of carbon-based fuels used to generate energy. By reducing air pollution in cities, we can improve the quality of life for millions of people, reduce the number of premature deaths, and mitigate the climate impacts of CO_2 emissions. Here, we discuss the links between CO_2 and coemitted air pollutants released from fossil and biofuels in mobile and stationary combustion and the impact of these emissions on air quality in cities. We highlight important examples of how policies to reduce CO_2 emissions can either degrade or improve urban air quality, depending on how they are implemented. Key recommendations are to (1) prioritize the removal of carbon-fuel based combustion, especially coal and wood burning; (2) electrify transportation, especially for colder climates and heavy-duty vehicles; (3) remove sulfur from all fuels; and (4) leverage existing air quality monitoring networks to increase density of CO_2 concentration measurements in cities worldwide.

INTRODUCTION

Covering just three percent of the Earth's land surface, cities are home to over half of the global population.[1] Urban areas are thought to contribute almost three-quarters of global carbon dioxide (CO_2) emissions,[2] but many cities do not have accurate estimates of their CO_2 emissions.[3] CO_2 is the most abundant greenhouse gas (GHG), reaching a record high of 419 ppm (parts per million) in May 2021,[4] and reducing CO_2 emissions is a focus of most climate mitigation policies. In urban areas, the emission of CO_2 and other air pollutants is driven by the combustion of carbon-based fuels (fossil and bio) to generate energy used in transportation, electricity, and residential heating and cooking (Figure 1). Ideally, the combustion of carbon-based fuels would be 100% efficient and lead to the production of only CO_2 and water vapor (H_2O). However, fossil fuels and biofuels are not composed of pure carbon nor are the reactions 100% efficient. Depending on the fuel type, trace gases and particles of various sizes containing impurities such as sulfur, nitrogen, chlorine (Cl), and mercury (Hg) are coemitted with CO_2 when fossil fuels and biofuels are burned, and many of these copollutants are toxic to human health.

Air pollution in cities mostly comprises primary pollutants, such as nitrogen oxides (NO_x = NO + NO_2), sulfur oxides (SO_x = SO + SO_2), carbon monoxide (CO), and particulate matter (PM; including black carbon or "soot"), all of which can be directly emitted from the combustion process alongside CO_2. Secondary pollutants, such as ozone (O_3) and additional "secondary" PM, are formed in the atmosphere when primary pollutants are photolyzed by sunlight and oxidized downwind of the emission source. The atmospheric lifetime of NO_2 is short, and it is destroyed by sunlight (through photolysis and oxidation), which means

The bigger picture

Challenges and opportunities:

- Atmospheric pollution is the leading cause of premature death worldwide. Concentrations of air pollutants that degrade air quality are often highest in and downwind of cities, where emissions from the combustion of fossil and biofuels are most intense.
- Climate policies designed to reduce CO_2 emissions could also improve air quality and save millions of lives. Carbon mitigation policies that do not account for the impact of coemitted pollutants could result in increased deaths from worsening air quality.
- Adding calibrated CO_2 analyzers to existing air quality monitoring networks could provide a framework to measure changes in carbon emissions from combustion in cities worldwide.

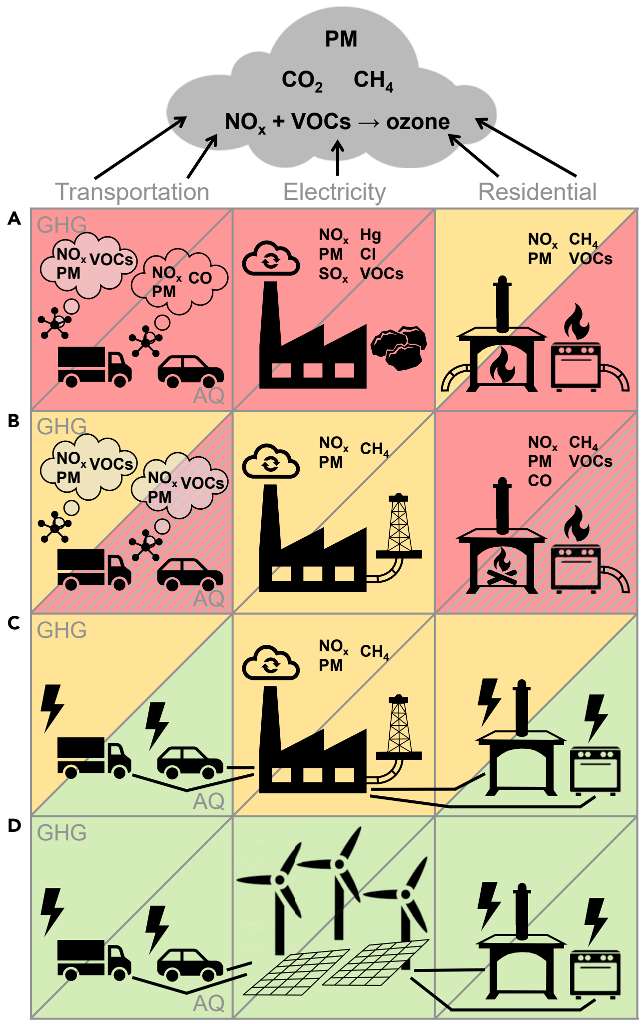

Figure 1. All combustion-based energy sectors can contribute to increased emissions of GHGs, such as CO_2 and CH_4, and impact air quality through the emissions of NO_x, VOCs, and PM

The GHG and air quality (AQ) impacts of various policies are shown here, where moving from red to yellow to green indicates reduced GHG or improved AQ

(A) Typical energy and combustion sources with transportation from diesel and petrol (mobile combustion sources in cities; red GHG, red AQ), electricity generation from coal (cooling space and electricity generation; red GHG, red AQ), and residential energy (home heating and cooking) from natural gas (heating and cooking: impacts on indoor and outdoor air quality; yellow GHG, red AQ, especially indoors).

(B) Transportation changing petrol to diesel (yellow, reduced GHG; red/black, worse AQ), electricity changing coal to natural gas (yellow, reduced GHG; yellow, improved AQ), and residential changing natural gas to wood burning (yellow, reduced GHG; red to black, dramatically worse AQ).

(C and D) (C) Shows reduced GHG emissions and cleaner AQ for all sectors when electrifying the grid, assuming the source of electricity is natural gas. However, (D) shows that we need to change the electrical grid energy source to noncombustion sources to reduce the GHG emissions while maintaining clean AQ.

that NO_2 is not transported far from the point of emission. Surface ozone is produced through atmospheric reactions following oxidation of CO or volatile organic compounds (VOCs) in the presence of sunlight and NO_2. With an atmospheric lifetime of hours to weeks, ozone can be transported over long distances. Reduced and semireduced sulfur and nitrogen compounds can be oxidized

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as they move away from source regions and contribute to PM as sulfate and nitrate, respectively.

NO_2, ozone, and PM kill millions of people each year,[5] and combined are the leading cause of premature deaths worldwide.[6] NO_2 and ozone cause respiratory distress and irritation of the eyes, nose, and throat[5] and higher mortality rates as concentrations increase. Ozone also damages crops and stunts growth.[7] High concentrations of PM are related to respiratory distress, increased cancer rates,[8] adverse effects on the cardiovascular system, and increased mortality.[9] The World Health Organization (WHO) recommends that each country should limit their concentrations of NO_x, ozone, and PM, but few countries and cities are able to consistently achieve these limits.

Meteorology, topography, and local vegetation also influence how air pollutants are processed in the urban atmosphere. Temperature inversions, especially in valleys, can trap pollutants within a city (e.g., Santiago, Chile; Salt Lake City, Utah; Po Valley, Italy). In summer, the increased sunlight on emissions trapped in valleys can speed up photochemical processes that generate secondary pollutants such as ozone (e.g., Los Angeles, California). The tall chimney stacks of power plants are designed to inject the emissions ~100 m above ground, where winds will normally transport much of the emissions away, rather than settling locally, allowing secondary pollutants, such as ozone, to have the largest impact on cities or suburbs downwind. With increasing intensity, frequency and duration of heatwaves,[10] and stagnation events where wind speeds drop to zero,[11] emissions will more often settle close to the point of emission.

The type of fossil fuel used in, or upwind of, cities is a key component of a city's urban air quality (AQ). Air pollution abatement strategies focus on the removal of pollutants at the point of emission, but carbon removal strategies are less well developed. For example, planting trees is a natural climate mitigation strategy to remove carbon from the atmosphere, and tree planting programs have been successful at counteracting the urban heat island effect in cities, where tree canopies provide shade at street level and the transpiration of water from their leaves helps to reduce the ambient temperature.[12] However, to maintain AQ standards, we must also consider what *species* of trees are planted. Some tree species, such as oak and poplar, emit relatively large amounts of isoprene,[13] a potent VOC and ozone precursor. So, for cities with large NO_x emissions, planting large quantities of oak[14] and poplar[15] trees could lead to increased ozone production, and hence, other tree species should be considered instead.

Here, we will discuss how carbon mitigation and pollution abatement strategies depend on the type of combustion (mobile or stationary) and the fuel used. We will also discuss how the existing AQ monitoring networks could provide a framework to measure carbon emission reductions, and what future policy changes would both improve AQ and reduce carbon emissions in cities.

MOBILE COMBUSTION SOURCES IN CITIES

Diesel and petrol (gasoline) internal combustion engines are the dominant source of transportation-related CO_2 and air pollutant emissions in many cities (Figure 1A Transportation). The rate of emission is a function of the type and age of the engine, the power output, etc. In petrol engines, the fuel is ignited by spark plugs, resulting in *lower* temperature combustion and less NO_x and more CO, VOCs, and PM produced. In diesel engines, the fuel self-ignites when it is injected into air that has been heated by compression, resulting in a *higher* temperature combustion, with more NO_x and PM but less CO and VOCs than petrol.[16] Much of the NO_x emitted from engines is not from the fuel itself but is produced from the temperature-dependent oxidation of atmospheric nitrogen (N_2)[17]; hence, hotter diesel engines produce more NO_x than cooler petrol engines. Mopeds and motorcycles in countries across Africa[18] and Asia,[19] and leaf-blowers in North America and Europe use two-stroke engines that use petrol with oil added and have comparable NO_x emissions with petrol but emit much larger concentrations of CO, VOCs, and PM.[20]

The catalytic converter systems employed on each engine determines the type and quantity of air pollutants coemitted with CO_2. These catalytic converters use precious metals such as platinum, palladium, or rhodium. Petrol engines use 3-way *redox* catalysts to reduce NO_x to N_2, followed by a separate oxidization of CO and hydrocarbons. The NO_x reduction reactions are efficient below 0.5% oxygen but will not proceed if oxygen concentrations are above 1%, whereas the oxidation of CO and hydrocarbons requires >5% oxygen. Lean-burn diesel engines contain high oxygen concentrations in their exhaust; hence, petrol-specific redox catalysts (and the NO_x reduction reactions in particular) cannot be used. Instead, lean-burn engines use a *diesel oxidation* catalyst, which oxidizes CO and hydrocarbons to CO_2, coupled with soot traps or filters to remove PM. NO_x can then be removed afterward using a number of catalytic converters. A "lean NO_x trap" (LNT), which adsorbs NO_x onto a zeolite material, must be purged or will become inefficient. LNT can also be poisoned by SO_2 produced in cars using high sulfur fuels (e.g., mostly countries in South America, Africa, and the Middle East as of 2021[21]). "Exhaust gas recirculation" (EGR) routes up to 50% of cooled exhaust gas back into the engine, reducing the temperature of combustion and NO_x production rates. However, EGR also reduces engine efficiency and peak power output; hence, it is typically not employed at high loads and is also omitted at idle (low speed, zero load) because it would cause unstable combustion. "Selective catalytic reduction" (SCR) reduces NO_x by reaction with anhydrous or aqueous ammonia or, more commonly, dissolved urea, as it is the easiest of these reductants to store. SCR is the most efficient NO_x reduction method and is typically employed on large boilers, turbines, and locomotive; ship and truck; and, recently, larger light vehicle engines (e.g., SUVs).

New direct injection petrol engines that have been designed to reduce CO_2 emissions have increased nitrous oxide (N_2O) and ammonia (NH_3) emissions,[22] and in some vehicles, the ammonia emissions now exceed the NO_x emissions.[16] Newer diesel vehicles using SCR and LNT (Euro6 category) also release ammonia,[23] resulting in ammonia quickly becoming the dominant form of nitrogen emitted from transportation.[24] Ammonia and NO_2 can lead to the formation of ammonium nitrate (NH_4NO_3), especially at colder temperatures, which further contributes to PM and degrading AQ. Catalytic converters require temperatures greater than 150°C to effectively reduce NO_x; hence, although they are relatively efficient at air temperatures close to 20°C, cold starts or operation in cooler climates can lead to increased NO_x emissions.[16] Nearly half of all passenger car driving mileage in northern latitude countries (like Germany) is performed at temperatures below 10°C, when converter efficiency is reduced, leading to higher real-life NO_x, ammonia, CO, and VOC emissions than that evaluated in laboratory tests. Tests also indicate that CO_2 and N_2O emissions are higher by 30% and 90%, respectively, in vehicles tested at −7°C instead of 20°C.[16]

Heavy-duty diesel vehicles have diesel engines that are optimized for highway driving conditions, emitting five times the certification limit below 25 mph and almost twice the limit at 50 mph,[25] which is the NO_x equivalent of 100 light vehicles for each mile driven in city driving conditions. Trucks and public buses have been leading in the use of biodiesel blends (diesel mixed with long chain monoalkyl esters from vegetable or animal fat at 5%–20%). Although the combustion of biodiesel blends results in less CO_2 emissions, there is some uncertainty about the associated air pollutant emissions, with one recent study indicating that modern trucks using 20% biodiesel would emit more air pollutants than pure diesel.[26]

Off-road transportation, such as ships, trains, and aircraft, also contribute to air pollution in cities. Until recently, shipping was a large contributor to urban air pollution in port cities or those with ferry systems.[27] In 2020, the International Convention for the Prevention of Pollution from Ships (MARPOL) was amended to reduce the allowed sulfur content of marine fuel from 3.5% (often called "bunker" fuel) to 0.5% sulfur by weight or ensure ships install exhaust scrubbers to reduce SO_x emissions.[28] Some special areas (coastal United States (US), North Sea, etc.) have an even lower limit of 0.1% sulfur by weight. Trains powered by diesel locomotive engines emit all the air pollutants described for on-road vehicles, but only some newer locomotives employ the abatement strategies described above. Freight trains are more efficient than heavy-duty on-road vehicles, and one study indicated that a transition of freight from road to rail in the upper Midwestern US (Michigan and Wisconsin) could reduce CO_2 by 31% and NO_x by 28%.[29] Airport emissions in cities include NO_x, VOC, and CO emissions from aircraft as they taxi, take-off, and land, and ground-based auxiliary power units at the gate.[30] On-road transportation to get passengers to and from the airport must also be included in the total airport emissions.

Diesel is denser than petrol, with higher boiling range petroleum fractions resulting in more fuel efficiency and ~13% more energy produced than from a liter of petrol, leading to lower CO_2 emissions per trip from diesel compared with petrol engines, with no pollution abatement methods implemented. However, recent studies have found similar real-world CO_2 emissions for petrol and diesel vehicles in Europe since 2012.[31] Over the past 30 years, the European Union (EU) has enacted laws promoting diesel engine use as a carbon reduction strategy (compared with petrol), with little consideration of the AQ implications. This policy moves Figure 1 Transportation from A to B, reducing GHG emissions but degrading AQ. Observed concentrations of air pollutants around Europe had long suggested that the reported emissions for some transportation-related sectors were too low, and a series of long-distance tests in 2014 in the US confirmed NO_x emissions 30–40 times higher than standard laboratory test and on-road regulations for some Volkswagen (VW) Group light vehicles.[32] Under-reporting NO_x emissions in laboratory tests versus real road conditions was a known problem for most diesel vehicles. However, a team working for the US Environmental Protection Agency (EPA) found that the high emissions for the VW vehicles were made possible by an illegal defeat device; software that only activated the EGR NO_x catalytic converter when the car detected regulation testing conditions.[33] Employing the EGR to reduce NO_x also has the undesired effect of reducing engine efficiency and power output. The "dieselgate" scandal eventually affected nearly 11 million VW, Audi, and Fiat vehicles in the US, Europe, and elsewhere and resulted in VW paying out nearly 16bn USD in fines and settlements in the US alone. It is estimated that diesel emissions in excess of regulations contributed to at least 38,000 fine PM and ozone-related premature deaths globally in 2015.[34] In 2015, on average, 52% of new light vehicle purchases across Europe were diesel[35] (up to 71% in Ireland). However, by 2019, the media attention on the pollution emissions of diesel vehicles during the VW scandal was more influential on the public than the tax incentives, and diesel had dropped to 32% of new vehicle purchases on average across the EU.[35]

Steps to reduce pollution from mobile combustion

The AQ implications of reducing vehicle emissions in cities, especially NO_x, is complex: at high NO_2 concentrations, ozone will be destroyed through reaction with NO. In these NO_x-saturated/VOC-limited environments (urban centers in Europe and China, and New York City), ozone concentrations will initially *increase* as on-road NO_x emissions (often largest in the highly populated urban core) are decreased. However, once the region transitions to a NO_x-limited regime, reductions in NO_x emissions will also reduce ozone for both the urban and downwind/suburban areas. This nonlinear chemistry also means that for some cities, NO_x emissions reductions may have to be quite dramatic before a reduction in ozone in the city center is observed. Fortunately, many major cities around the world (e.g., New York City, London, and Seoul) have seen large NO_x reductions in recent years, and further NO_x emission controls will also reduce secondary ozone production much more than a decade ago.[36]

Moving away from petrol and diesel by electrifying the transportation fleet would dramatically improve AQ in urban areas worldwide (moving Figure 1 Transportation from B to C and D). This is especially true for cities in colder climates (latitudes north of 50°N, south of 50°S, or high altitudes[37]) and for heavy-duty vehicles that operate in slow-moving city traffic. Sales figures as of August 2020 suggest that the EU leads the way in electric vehicle (EV) sales (8% of all new vehicles purchased), followed by China (4%), and the US is far behind at <2%.[35] Moving to an EV fleet would change the temporal profile of emissions; peak daytime, on-road vehicle emissions would change to overnight power plant emissions from energy needed to charge the EVs.[38] Hybrid EVs have been promoted as a bridge to reducing CO_2 and air pollutant emissions from transportation, but their ammonia and some VOC emissions are similar to gasoline vehicles[39]; hence, hybrid EV use will still impact AQ but less than combustion engines.

The introduction of limited vehicle "congestion zones" (as implemented in London, sporadically in Paris, and planned for New York City) could also reduce air pollutants in some areas. These zones would reduce the overall emission by reducing the number of vehicles in the city core and allow heavy goods delivery vehicles to operate at more optimal speeds that reduce their NO_x emissions.

In cities dominated by two-stroke engines (e.g., many countries in Africa and Asia), the reduction in PM from converting the vehicle fleet would be immediately noticeable. In the meantime, the reduction of sulfur limits in countries that still allow the use of high sulfur fuels would allow catalytic converters to function properly and scrub SO_2, PM, and NO_x emissions and reduce ozone and secondary PM production. PM from sulfate sources cool the atmosphere[40]; hence, although we would quickly see a dramatic improvement in health outcomes, additional GHG emissions reductions will be needed to account for the warming associated with sulfate-based PM removal.

STATIONARY COMBUSTION SOURCES IN CITIES

Stationary combustion sources use diverse fossil and biofuels to generate electricity, provide heat, and facilitate cooking in buildings (Figure 1 Electricity and Residential

A and B). The CO_2 emissions from "building" sources often referred to in inventories include both the CO_2 produced from the materials the building is made from (~one-third of total building energy) and the CO_2 emitted from combustion processes when operating the building (~two-thirds of total building energy).[41] The spaces in buildings require heating, cooling, or both, depending on the local climate and season, with different AQ impacts depending on the process.

Cooling space and electricity generation

Electricity is generated for appliances and lights, but the cooling of spaces in buildings is the largest power demand of any process within a building. Cooling is usually powered by the electrical grid and currently contributes to ~20% of the total electricity use worldwide.[42] Extreme temperatures impact human health, especially in cities where the concrete buildings and pavements magnify temperatures. As more heatwaves are expected with warming climates,[10] demand for cooling is expected to grow.[42] Households have been found to increase their electricity use by 35%–42% when they adopt air conditioning.[43] Electricity generation units (EGUs) (commonly known as power plants) burn fossil fuels such as coal and natural gas and are generally located outside the urban core of most modern cities, limiting their impact on urban AQ. Many cities also rely on additional EGUs and smaller generators ("peaking units") within the city during times of high energy demand. New large EGUs are required to scrub NO_x and SO_x from their stack emissions in some countries, but older EGUs, peaking units, or generators are not. The source of electricity used in cooling is especially important as summer time photochemistry can intensify the air pollution impacts from these emissions.

Coal use, despite once dominating as the largest home heating and power generation source in cities, has started to decline in recent years (moving Figure 1 Electricity from A to B). Of fossil fuels burned in cities, coal is the least efficient, is the largest carbon emitter, and releases the most air pollutants. Anthracite coal (so-called "clean coal") contains slightly less sulfur than bituminous coal, but the combustion of all types of coal releases SO_x and NO_x, chlorine, and large quantities of heavy metals such as lead, arsenic, and mercury.[44] The combustion of coal has been linked to some of the most infamous AQ events in history: In London, by the 1950, the use of brown coal (similar to lignite) was widespread for industry, power generation, and home heating. In December 1952, an unusually cold and stagnant anticyclone, coupled with the toxic emissions from coal combustion, caused the infamous "London Smog" and resulted in the death of more than 12,000 people.[45] The incident eventually led to the UK Clean Air Act of 1956.

The use of coal has rapidly decreased in European cities over the past 5 years: coal use dropped to 13% of total power generation in 2020.[46] In the US, coal still contributes almost 20% of total power generation in 2020.[47] However, this coal use reduction has not been consistent across the globe. In China, 57% of total energy needs were met by coal,[47] with China consuming over half of coal burned globally in 2020.[48] Newer coal plants use flue gas desulfurization (FGD) to remove the SO_x, SCR to remove NO_x, and FGD and SCR combined to remove mercury from the exhaust gas. However, 40% of coal EGUs in the US are not fitted with FGD or SCR and continue to emit SO_x and NO_x, as well as release up to 50% of anthropogenic mercury emissions for the US.[44]

Natural gas emits fewer air pollutants than coal when completely burned. Once it is refined, natural gas provided to consumers is primarily composed of methane (CH_4), with some other hydrocarbons present (<10% ethane, butane, etc). Methane is a

GHG that warms the atmosphere over 80 times more than CO_2 over 20 years[40] and has an atmospheric lifetime of about 10 years until it is oxidized and produces ground-level ozone among other secondary pollutants.[49] As with any fuel, the combustion of natural gas is rarely complete, especially in inefficient or leaking consumer products,[50] and street level leaks of natural gas are common in urban areas with older pipeline infrastructure.[51–53]

Diesel is commonly used as the primary source of fuel for portable generators, which are most often employed in remote areas and after natural disasters (e.g., Puerto Rico after Hurricane Maria[54]). These diesel generators are very inefficient and emit large amounts of CO, hence, care must be taken that the exhaust of these portable generators does not vent indoors. Fuel oil is also used in EGUs for short periods when stores of other fuels, such as natural gas, run low or in peaking units when demand is high.

Heating and cooking: Impacts on indoor and outdoor air quality

In most colder winter climates, the heating of buildings in cities is often independent of the electrical grid and instead relies on a large number of on-site boilers that are fueled by natural gas, coal, fuel oil, or wood/biomass. If not dependent on electricity, restaurant and home cooking generally use similar fuels.

Residential and commercial boilers can use residual or distillate "home heating" fuel oil of different grades or impurities, which also corresponding to different air pollutant emissions. No. 2 distillate fuel oil is largely consumed in the northeastern US, Canada, UK, and Ireland. High sulfur fuels No. 6 (marine bunker fuel—until recently used in ships) and No. 4 fuel oil were used in New York City until 2015, where they were associated with increased SO_2, NO_x, and soot emissions.[55]

Wood burning has been framed as a carbon-neutral source of energy and an alternative to coal (moving Figure 1 Residential from A to B), but by burning this reservoir of carbon, we are releasing CO_2 emissions that occurred and were captured 30–50 years ago back into the atmosphere. Similar to wildfires, the combustion of wood and solid biofuels results in very high concentrations of particles in the form of elemental carbon/soot and carbon-rich PM, as well as hundreds of trace gases including CO, CO_2, CH_4, hydrogen cyanide, and acetonitrile.[56,57] Wood or coal stoves and open fires, often used without any PM filtration, contribute to toxic winter-time pollution events in cities and towns all around the world from Fairbanks, Alaska[58] to Paris[59] to Ulaanbaatar, Mongolia.[60]

Heating sources are designed to release the exhaust gas outside the building, but cooking still contributes to indoor air pollution. Commercial kitchens are required to actively ventilate the air pollutant released from combustion out of the enclosed space, but homes, and especially small city apartments, are often inadequately ventilated. Natural gas ovens and stoves emit large concentrations of NO_x, CO, and PM[61] into kitchens and impact indoor AQ. Three billion people use wood or other solid biofuel (dung, crop residue, charcoal, etc.) to meet their daily cooking needs,[62] and about 4 million premature deaths each year are attributed to burning of wood and other biomass in homes.[63]

Steps to reduce pollution from stationary combustion

All electricity used in cities should be sourced from noncombustion sources (see Figure 1). Until carbon-free electricity infrastructure is constructed, power plants should be moved out of urban areas, continue to transition to natural gas as their primary fuel, and stop the use of solid fuels, such as coal or wood/biomass, or secondary fuels, such as oil or diesel. For example, Figure 1 shows how moving electricity from natural gas (B) to coal (A) or residential heating from liquid (A) to solid fuels (B) results in greater GHG emissions and degraded AQ.

Coal or wood should not be burned in any urban area for either home heating or electricity generation (e.g., Figure 1 Residential moving A to B). Some progress has been made on reducing wood and coal pollution by encouraging transitions to natural gas: In China, after aggressive efforts to reduce household burning of coal because of air pollution impacts, the use of solid fuels in cooking decreased from 60% in 2005 to 32% in 2017.[64] Similar to electrifying vehicles, there could be a slight increase in ozone in NO_x-saturated cities as NO_x emissions are initially reduced, but the reduction in VOC and CO emissions from removing stationary combustion would also reduce ozone production.

In addition, the burning of wood/biomass in converted coal power plants needs to be reconsidered as a carbon-neutral source of electricity (e.g., Drax Power Plant in North Yorkshire, UK, which burns wood pellets from the US, Canada, and Brazil). Although the burned wood is renewable and may be salvaged from other industrial or manufacturing processes, carbon is nonetheless being released back into the atmosphere much quicker than it is sequestered through tree growth. The wood pellets are often shipped to the EGU from thousands of miles away. The massive net carbon emissions from the combustion of wood pellets are not included in many global carbon inventories as they are considered "sustainable." However, regardless of the label, the burning of wood pellets results in large copollutant emissions detrimental to AQ.

Further, by phasing out the combustion of carbon fuels for cooking, indoor AQ and the associated health impacts could be improved worldwide (moving from B to C and D in Figure 1 Residential). The impact of natural gas combustion in ovens and stoves[61,65] on indoor AQ is often underappreciated. Motivated by the climate impacts of fugitive natural gas emissions, stopping new natural gas connections has become law in some US cities. Programs have encouraged households in various countries to move away from cooking with biofuels to reduce PM indoors.[66–68]

MONITORING COMBUSTION EMISSIONS IN URBAN AREAS

Many cities do not accurately calculate their carbon emissions[3]; hence, it is difficult to assess the expected impact of carbon mitigation policies on CO_2 emissions within the city. Over the past decade, ground-based monitoring networks for GHGs have been developed for urban areas to address this problem directly. These networks focus on cities such as Baltimore and Washington, D.C.,[69] Boston,[70] Indianapolis,[71,72] Los Angeles,[73] Paris,[74] Salt Lake City,[75] and San Francisco.[76,77] The networks measure the CO_2 concentration enhancement in the atmosphere from emissions within the domain (so-called Scope 1 emissions) relative to the background CO_2 concentration in the air arriving from elsewhere. Using these urban-scale CO_2 enhancements aggregated over time, researchers can quantify the CO_2 emissions of a city through inverse methods.

Each network has a different sampling design, allowing them to leverage the trade-off between precision and measurement quantity. Some networks use fewer than 10 high precision instruments sampling from towers across the domain (e.g., Boston,[70] Salt Lake City[75]), some use more than 70 of these towers across a region (e.g., Baltimore[69]), whereas others use a high-density network of lower cost sensors across a city (e.g., BEACO$_2$N in San Francisco).[76,77] Many of the networks observed large

CO_2 changes during the COVID-19 lockdowns in 2020,[73,78] but future efforts to monitor more subtle changes in CO_2 due to carbon reduction policies will require further developments of these networks.

Atmospheric pollutants are coemitted with CO_2 at different ratios during the various carbon combustion processes. We can take advantage of this coemission by using the changes in air pollutants to monitor the combustion-driven carbon emissions in cities, both known and unknown. A few of the existing CO_2 monitoring networks have added low-cost sensors to measure gas species coemitted during combustion (e.g., CO, NO_2) and PM to attribute sources of CO_2.[78] Another alternative is to use existing AQ monitoring sites, which are already established in many cities and operated by local and municipal governments under guidance from State or Federal agencies (e.g., EPA in the US, European Environment Agency [EEA] in the EU, etc.). AQ sites are designed to sample highly reactive gases, and the operation of sites requires trained technical staff and long calibration histories. AQ sites should be augmented with calibrated CO_2 analyzers to determine a city-specific, spatially resolved, combustion signal of the NO_x/CO_2 ratio. The addition of CO_2 at established AQ sites would be a technically easy and cost-effective approach, assuming calibration standards (traceable to World Meteorological Organization [WMO]) are available.

There is currently no global, high-resolution satellite mission observing CO_2 at the urban scale. Existing satellites such as NASA's Orbiting Carbon Observatory-2 (OCO-2) are not designed to monitor CO_2 emissions on urban scales,[79] and OCO-3 on the International Space Station targets a limited number of urban areas.[80] However, suitable CO_2 satellites will start to come online over the next decade: NASA's Geostationary Carbon Cycle Observatory (GeoCarb) will measure CO_2 and CO over North America (launch planned for 2024), followed by detailed global CO_2 measurements by the Copernicus Carbon Dioxide Monitoring mission (CO2M by the European Space Agency [ESA], launch planned for 2025).

In the meantime, AQ satellites are being used to monitor city-specific estimates of CO_2 emissions from combustion: Recent studies have used the coemission of NO_x with CO_2 to estimate CO_2 emissions from coal power plants in rural areas[81] using the daily tropospheric NO_2 column product from the ESA's TropOMI satellite as a proxy for total NO_x emissions. The large number of co-located NO_x and CO_2 emissions sources in urban areas make it more difficult to use this approach to identify individual combustion changes within cities. However, hourly NO_2 products from GEMS (Asia) and soon to be launched TEMPO (US) and Sentinel-4 (Europe) will measure changes in the tropospheric column of NO_2 on a finer spatial resolution than previously possible.

By combining new satellite products with *in situ* NO_2 and CO_2 measurement networks, we could monitor changes in combustion-related CO_2 emissions on the city scale into the future or even back over the duration of the satellite record in some locations. Closer links between the carbon mitigation and AQ communities would greatly benefit both communities as we strive to reduce carbon emissions and improve air qualities in urban areas.

THE FUTURE OF COMBUSTION IN CITIES

The infrastructure investments required to remove combustion sources from cities may be economically daunting; however, numerous studies have shown that the human health cobenefits of reducing the air pollutants coemitted with CO_2 during combustion would more than offset the cost of climate policy implementation.[38] Cost-benefit analyses, such as the social cost of carbon dioxide (SCC), used in developing carbon mitigation policies in the US[82] calculate the economic cost of the damages that would result from emitting one additional ton of carbon into the atmosphere. Such calculations would be a more accurate estimate of costs and benefits if they included the health and economic impact of air pollutants coemitted with CO_2 during combustion in addition to the cost of the climate response and economic damages. The indoor air pollution impacts of natural gas combustion within the home are often overlooked in cost-benefit analyses. In large cities, many people live in small apartments with open plan kitchens and bad ventilation—apartments where the combustion of natural gas has a potent impact on indoor AQ. Using both indoor and outdoor AQ indicators to lead climate change policy would be an economically viable strategy to achieve climate change goals and protect human health and the environment.

In our opinion, combustion in cities must be phased out to improve AQ and reduce GHG emissions. All internal combustion engines, regardless of fuel type, need to be removed from cities (Figure 1 Transportation A and B need to move to C). For small towns and big cities alike, the burning of solid fuels (e.g., coal and wood), especially in winter, must be stopped immediately (Figure 1 Residential A and B move to C). Eventually *all* combustion (e.g., natural gas and propane) must be stopped and transitioned to electrical sources, especially the combustion of solid and liquid fuels indoors (Figure 1 C move to D).

Due to the sheer volume of emissions each year, CO_2 is the largest driver of climate warming of all the GHGs and is the focus of most climate mitigation policies. However, sometimes, CO_2 emissions reductions are a cobenefit of policies focused on other target species. Methane, CO, and ozone are potent GHGs in their own right. Ozone, a secondary pollutant of NO_x emissions, and CO (coemitted during incomplete combustion processes) are targets of National Ambient Air Quality Standards (NAAQS) in the US and WHO standards worldwide. Methane has become the focus of the many climate-related laws (e.g., New York State's Climate Leadership and Community Protection Act [CLCPA]) and international meetings (COP26 meeting in November 2021). The 2021 signing of a "Natural Gas Ban" in New York City will reduce methane emissions and vastly improve both outdoor and indoor AQ across the city, assuming that the additional electricity generation required to replace the combusted natural gas is from noncombustion sources. The reduction of combustion processes that release CO and methane and produce ozone alongside CO_2, will reduce climate warming while also improving AQ.

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AUTHOR CONTRIBUTIONS

R.C. conceptualized the topic, and R.C. and L.D.S. wrote the manuscript.

DECLARATION OF INTERESTS

The authors declare no competing interests.

REFERENCES

1. United Nations, Department of Economic and Social Affairs, Population Division (2018). The World's Cities in 2018: Data Booklet (United Nations).
2. Gurney, K.R., Romero-Lankao, P., Seto, K.C., Hutyra, L.R., Duren, R., Kennedy, C., Grimm, N.B., Ehleringer, J.R., Marcotullio, P., Hughes, S., et al. (2015). Climate change: track urban emissions on a human scale. Nature *525*, 179–181.
3. Gurney, K.R., Liang, J., Roest, G., Song, Y., Mueller, K., and Lauvaux, T. (2021). Under-reporting of greenhouse gas emissions in U.S. cities. Nat. Commun. *12*, 553.
4. NOAA Research News (2021). Carbon dioxide peaks near 420 parts per million at Mauna Loa Observatory. https://research.noaa.gov/article/ArtMID/587/ArticleID/2764.
5. Watts, N., Amann, M., Arnell, N., Ayeb-Karlsson, S., Beagley, J., Belesova, K., Boykoff, M., Byass, P., Cai, W., Campbell-Lendrum, D., et al. (2021). The 2020 report of the Lancet Countdown on health and climate change: responding to converging crises. Lancet *397*, 129–170.
6. Lelieveld, J., Pozzer, A., Pöschl, U., Fnais, M., Haines, A., and Münzel, T. (2020). Loss of life expectancy from air pollution compared to other risk factors: a worldwide perspective. Cardiovasc. Res. *116*, 1910–1917.
7. Avnery, S., Mauzerall, D.L., Liu, J., and Horowitz, L.W. (2011). Global crop yield reductions due to surface ozone exposure: 1. year 2000 crop production losses and economic damage. Atmos. Environ. *45*, 2284–2296.
8. Turner, M.C., Andersen, Z.J., Baccarelli, A., Diver, W.R., Gapstur, S.M., Pope, C.A., Prada, D., Samet, J., Thurston, G., and Cohen, A. (2020). Outdoor air pollution and cancer: an overview of the current evidence and public health recommendations. CA Cancer J. Clin. *70*, 460–479.
9. Dockery, D.W., Pope, C.A., Xu, X., Spengler, J.D., Ware, J.H., Fay, M.E., Ferris, B.G., and Speizer, F.E. (1993). An association between air pollution and mortality in six U.S. cities. N. Engl. J. Med. *329*, 1753–1759.
10. Perkins-Kirkpatrick, S.E., and Lewis, S.C. (2020). Increasing trends in regional heatwaves. Nat. Commun. *11*, 3357.
11. Horton, D.E., Skinner, C.B., Singh, D., and Diffenbaugh, N.S. (2014). Occurrence and persistence of future atmospheric stagnation events. Nat. Clim. Change *4*, 698–703.
12. Ziter, C.D., Pedersen, E.J., Kucharik, C.J., and Turner, M.G. (2019). Scale-dependent interactions between tree canopy cover and impervious surfaces reduce daytime urban heat during summer. Proc. Natl. Acad. Sci. USA *116*, 7575–7580.
13. Sharkey, T.D., Wiberley, A.E., and Donohue, A.R. (2008). Isoprene emission from plants: why and how. Ann. Bot. *101*, 5–18.
14. Sillman, S. (1999). The relation between ozone, NO_x and hydrocarbons in urban and polluted rural environments. Atmos. Environ. *33*, 1821–1845.
15. Wiedinmyer, C., Tie, X., Guenther, A., Neilson, R., and Granier, C. (2006). Future changes in biogenic isoprene emissions: how might they affect regional and global atmospheric chemistry? Earth Interact. *10*, 1–19.
16. Suarez-Bertoa, R., and Astorga, C. (2018). Impact of cold temperature on euro 6 passenger car emissions. Environ. Pollut. *234*, 318–329.
17. Lavoie, G.A., Heywood, J.B., and Keck, J.C. (1970). Experimental and theoretical study of nitric oxide formation in internal combustion engines. Combust. Sci. Technol. *1*, 313–326.
18. Assamoi, E.-M., and Liousse, C. (2010). A new inventory for two-wheel vehicle emissions in West Africa for 2002. Atmos. Environ. *44*, 3985–3996.
19. Hopke, P.K., Cohen, D.D., Begum, B.A., Biswas, S.K., Ni, B., Pandit, G.G., Santoso, M., Chung, Y.S., Rahman, S.A., Hamzah, M.S., et al. (2008). Urban air quality in the Asian region. Sci. Total Environ. *404*, 103–112.
20. Begum, B.A., Biswas, S.K., and Hopke, P.K. (2006). Impact of banning of two-stroke engines on airborne particulate matter concentrations in Dhaka, Bangladesh. J. Air Waste Manag. Assoc. *56*, 85–89.
21. Stratas Advisors. (2021). Gasoline Sulfur Limits. https://stratasadvisors.com/Insights/2021/07232021-Top-100-gasoline-sulfur-ranking.
22. Guan, B., Zhan, R., Lin, H., and Huang, Z. (2014). Review of state of the art technologies of selective catalytic reduction of NO_x from diesel engine exhaust. Appl. Therm. Eng. *66*, 395–414.
23. Suarez-Bertoa, R., Zardini, A.A., Lilova, V., Meyer, D., Nakatani, S., Hibel, F., Ewers, J., Clairotte, M., Hill, L., and Astorga, C. (2015). Intercomparison of real-time tailpipe ammonia measurements from vehicles tested over the new world-harmonized light-duty vehicle test cycle (WLTC). Environ. Sci. Pollut. Res. Int. *22*, 7450–7460.
24. Bishop, G.A., and Stedman, D.H. (2015). Reactive nitrogen species emission trends in three light-/Medium-duty United States fleets. Environ. Sci. Technol. *49*, 11234–11240.
25. Badshah, H., Posada, F., and Muncrief, R. (2019). Current State of NO_x Emissions from in-Use Heavy-Duty Diesel Vehicles in the United States International Council on Clean Transportation. https://theicct.org/publication/current-state-of-nox-emissions-from-in-use-heavy-duty-diesel-vehicles-in-the-united-states/.
26. O'Malley, J., and Searle, S. (2021). Air Quality Impacts of Biodiesel in the United States International Council on Clean Transportation. https://theicct.org/publication/air-quality-impacts-of-biodiesel-in-the-united-states/.
27. Sorte, S., Rodrigues, V., Borrego, C., and Monteiro, A. (2020). Impact of harbour activities on local air quality: a review. Environ. Pollut. *257*, 113542.
28. International Maritime Organization Amendments to MARPOL (2018). Annex VI. https://wwwcdn.imo.org/localresources/en/KnowledgeCentre/IndexofIMOResolutions/MEPCDocuments/MEPC.305%2873%29.pdf.
29. Bickford, E., Holloway, T., Karambelas, A., Johnston, M., Adams, T., Janssen, M., and Moberg, C. (2014). Emissions and air quality impacts of truck-to-rail freight modal shifts in the Midwestern United States. Environ. Sci. Technol. *48*, 446–454.
30. Masiol, M., and Harrison, R.M. (2014). Aircraft engine exhaust emissions and other airport-related contributions to ambient air pollution: a review. Atmos. Environ. (1994) *95*, 409–455.
31. Helmers, E., Leitão, J., Tietge, U., and Butler, T. (2019). CO_2-equivalent emissions from European passenger vehicles in the years 1995–2015 based on real-world use: assessing the climate benefit of the European "diesel boom. Atmos. Environ. *198*, 122–132.
32. Thompson, G.J., Carder, D.K., Besch, M.C., Thiruvengadam, A., and Kappanna, H.K. (2014). In-Use Emissions Testing of Light-Duty Diesel Vehicles in the United States. https://theicct.org/publication/in-use-emissions-testing-of-light-duty-diesel-vehicles-in-the-u-s/.
33. Contag, M., Li, G., Pawlowski, A., Domke, F., Levchenko, K., Holz, T., and Savage, S. (2017). How they did it: an analysis of emission defeat devices in modern automobiles. In IEEE Symposium on Security and Privacy (SP) (IEEE), pp. 231–250.
34. Anenberg, S.C., Miller, J., Minjares, R., Du, L., Henze, D.K., Lacey, F., Malley, C.S., Emberson, L., Franco, V., Klimont, Z., et al. (2017). Impacts and mitigation of excess diesel-related NO_x emissions in 11 major vehicle markets. Nature *545*, 467–471.
35. Diaz, S. (2020). European Vehicle Market Statistics 2020/21. 55. https://theicct.org/publication/european-vehicle-market-statistics-2020-21/.
36. Jin, X., Fiore, A.M., Murray, L.T., Valin, L.C., Lamsal, L.N., Duncan, B., Folkert Boersma, K., De Smedt, I., Abad, G.G., Chance, K., et al. (2017). Evaluating a space-based indicator of surface ozone-NO_x-VOC sensitivity over midlatitude source regions and application to decadal trends: space-based indicator of O_3 sensitivity. J. Geophys. Res. Atmos. *122*, 10439–10461.
37. Mourshed, M. (2016). Climatic parameters for building energy applications: a temporal-geospatial assessment of temperature indicators. Renew. Energy *94*, 55–71.
38. Gallagher, C.L., and Holloway, T. (2020). Integrating air quality and public health benefits in U.S. Decarbonization strategies. Front. Public Health *8*, 563358.
39. Suarez-Bertoa, R., and Astorga, C. (2016). Unregulated emissions from light-duty hybrid electric vehicles. Atmos. Environ. *136*, 134–143.
40. Masson-Delmotte, V., Zhai, P., Pirani, A., Connors, S.L., Pean, C., Berger, S., Caud, N., Chen, Y., Goldfarb, L., Gomis, M.I., et al. (2021). Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the

Intergovernmental Panel on Climate Change.IPCC. https://www.ipcc.ch/report/ar6/wg1/.

41. UN Environment and International Energy Agency (2017). Global Status Report 2017: Towards a Zero-Emission, Efficient, and Resilient Buildings and Construction Sector. https://www.worldgbc.org/sites/default/files/UNEP%20188_GABC_en%20%28web%29.pdf.

42. IEA Paris (2018). The Future of Cooling. https://www.iea.org/reports/the-future-of-cooling.

43. Randazzo, T., De Cian, E., and Mistry, M.N. (2020). Air conditioning and electricity expenditure: the role of climate in temperate countries. Econ. Modell. *90*, 273–287.

44. Streets, D.G., Lu, Z., Levin, L., ter Schure, A.F.H., and Sunderland, E.M. (2018). Historical releases of mercury to air, land, and water from coal combustion. Sci. Total Environ. *615*, 131–140.

45. Bell, M.L., Davis, D.L., and Fletcher, T. (2004). A retrospective assessment of mortality from the London smog episode of 1952: the role of influenza and pollution. Environ. Health Perspect. *112*, 6–8.

46. Agora Energiewende and Ember (2021). The European Power Sector in 2020: up-to-date analysis on the electricity transition. https://www.agora-energiewende.de/en/publications/the-european-power-sector-in-2020/.

47. BP. (2021). Statistical review of world energy. https://www.bp.com/en/global/corporate/energy-economics/statistical-review-of-world-energy.html.

48. Dave Jones. (2021). Global electricity review 2021 - global trends.Ember. https://ember-climate.org/project/global-electricity-review-2021/.

49. Fiore, A.M., Naik, V., and Leibensperger, E.M. (2015). Air quality and climate connections. J. Air Waste Manag. Assoc. *65*, 645–685.

50. Saint-Vincent, P.M.B., and Pekney, N.J. (2020). Beyond-the-meter: unaccounted sources of methane emissions in the natural gas distribution sector. Environ. Sci. Technol. *54*, 39–49.

51. Ars, S., Vogel, F., Arrowsmith, C., Heerah, S., Knuckey, E., Lavoie, J., Lee, C., Pak, N.M., Phillips, J.L., and Wunch, D. (2020). Investigation of the spatial distribution of methane sources in the greater Toronto area using mobile gas monitoring systems. Environ. Sci. Technol. *54*, 15671–15679.

52. McKain, K., Down, A., Raciti, S.M., Budney, J., Hutyra, L.R., Floerchinger, C., Herndon, S.C., Nehrkorn, T., Zahniser, M.S., Jackson, R.B., et al. (2015). Methane emissions from natural gas infrastructure and use in the urban region of Boston, Massachusetts. Proc. Natl. Acad. Sci. USA *112*, 1941–1946.

53. Lamb, B.K., Cambaliza, M.O.L., Davis, K.J., Edburg, S.L., Ferrara, T.W., Floerchinger, C., Heimburger, A.M.F., Herndon, S., Lauvaux, T., Lavoie, T., et al. (2016). Direct and indirect measurements and modeling of methane emissions in Indianapolis, Indiana. Environ. Sci. Technol. *50*, 8910–8917.

54. Subramanian, R., Ellis, A., Torres-Delgado, E., Tanzer, R., Malings, C., Rivera, F., Morales, M., Baumgardner, D., Presto, A., and Mayol-Bracero, O.L. (2018). Air quality in Puerto Rico in the aftermath of Hurricane Maria: a case study on the use of lower cost air quality monitors. ACS Earth Space Chem. *2*, 1179–1186.

55. Clougherty, J.E., Kheirbek, I., Eisl, H.M., Ross, Z., Pezeshki, G., Gorczynski, J.E., Johnson, S., Markowitz, S., Kass, D., and Matte, T. (2013). Intra-urban spatial variability in wintertime street-level concentrations of multiple combustion-related air pollutants: the New York City Community Air Survey (NYCCAS). J. Expo. Sci. Environ. Epidemiol. *23*, 232–240.

56. Akagi, S.K., Yokelson, R.J., Wiedinmyer, C., Alvarado, M.J., Reid, J.S., Karl, T., Crounse, J.D., and Wennberg, P.O. (2011). Emission factors for open and domestic biomass burning for use in atmospheric models. Atmos. Chem. Phys. *11*, 4039–4072.

57. Andreae, M.O. (2019). Emission of trace gases and aerosols from biomass burning—an updated assessment. Atmos. Chem. Phys. *19*, 8523–8546.

58. Ward, T., Trost, B., Conner, J., Flanagan, J., and Jayanty, R.K.M. (2012). Source apportionment of PM2.5 in a subarctic airshed—Fairbanks, Alaska. Aerosol Air Qual. Res. *12*, 536–543.

59. Favez, O., Cachier, H., Sciare, J., Sarda-Estève, R., and Martinon, L. (2009). Evidence for a significant contribution of wood burning aerosols to PM2.5 during the winter season in Paris, France. Atmos. Environ. *43*, 3640–3644.

60. Guttikunda, S.K., Lodoysamba, S., Bulgansaikhan, B., and Dashdondog, B. (2013). Particulate pollution in Ulaanbaatar, Mongolia. Air Qual. Atmos. Health *6*, 589–601.

61. Singer, B.C., Pass, R.Z., Delp, W.W., Lorenzetti, D.M., and Maddalena, R.L. (2017). Pollutant concentrations and emission rates from natural gas cooking burners without and with range hood exhaust in nine California homes. Build. Environ. *122*, 215–229.

62. Bonjour, S., Adair-Rohani, H., Wolf, J., Bruce, N.G., Mehta, S., Prüss-Ustün, A., Lahiff, M., Rehfuess, E.A., Mishra, V., and Smith, K.R. (2013). Solid fuel use for household cooking: country and regional estimates for 1980–2010. Environ. Health Perspect. *121*, 784–790.

63. Lim, S.S., Vos, T., Flaxman, A.D., Danaei, G., Shibuya, K., Adair-Rohani, H., Amann, M., Anderson, H.R., Andrews, K.G., Aryee, M., et al. (2012). A comparative risk assessment of burden of disease and injury attributable to 67 risk factors and risk factor clusters in 21 regions, 1990–2010: a systematic analysis for the Global Burden of Disease Study 2010. Lancet *380*, 2224–2260.

64. Yin, P., Brauer, M., Cohen, A.J., Wang, H., Li, J., Burnett, R.T., Stanaway, J.D., Causey, K., Larson, S., Godwin, W., et al. (2020). The effect of air pollution on deaths, disease burden, and life expectancy across China and its provinces, 1990–2017: an analysis for the Global Burden of Disease Study 2017. Lancet Planet. Health *4*, e386–e398.

65. Logue, J.M., Klepeis, N.E., Lobscheid, A.B., and Singer, B.C. (2014). Pollutant exposures from natural gas cooking burners: a simulation-based assessment for Southern California. Environ. Health Perspect. *122*, 43–50.

66. Johnson, M., Edwards, R., Alatorre Frenk, C., and Masera, O. (2008). In-field greenhouse gas emissions from cookstoves in rural Mexican households. Atmos. Environ. *42*, 1206–1222.

67. Christian, T.J., Yokelson, R.J., Cárdenas, B., Molina, L.T., Engling, G., and Hsu, S.-C. (2010). Trace gas and particle emissions from domestic and industrial biofuel use and garbage burning in central Mexico. Atmos. Chem. Phys. *10*, 565–584.

68. Mukhopadhyay, R., Sambandam, S., Pillarisetti, A., Jack, D., Mukhopadhyay, K., Balakrishnan, K., Vaswani, M., Bates, M.N., Kinney, P.L., Arora, N., et al. (2012). Cooking practices, air quality, and the acceptability of advanced cookstoves in Haryana, India: an exploratory study to inform large-scale interventions. Glob. Health Action *5*, 1–13.

69. Karion, A., Callahan, W., Stock, M., Prinzivalli, S., Verhulst, K.R., Kim, J., Salameh, P.K., Lopez-Coto, I., and Whetstone, J. (2020). Greenhouse gas observations from the Northeast Corridor tower network. Earth Syst. Sci. Data *12*, 699–717.

70. Sargent, M., Barrera, Y., Nehrkorn, T., Hutyra, L.R., Gately, C.K., Jones, T., McKain, K., Sweeney, C., Hegarty, J., Hardiman, B., et al. (2018). Anthropogenic and biogenic CO_2 fluxes in the Boston urban region. Proc. Natl. Acad. Sci. USA *115*, 7491–7496.

71. Lauvaux, T., Gurney, K.R., Miles, N.L., Davis, K.J., Richardson, S.J., Deng, A., Nathan, B.J., Oda, T., Wang, J.A., Hutyra, L., et al. (2020). Policy-relevant assessment of urban CO_2 emissions. Environ. Sci. Technol. *54*, 10237–10245.

72. Mueller, K.L., Lauvaux, T., Gurney, K.R., Roest, G., Ghosh, S., Gourdji, S.M., Karion, A., DeCola, P., and Whetstone, J. (2021). An emerging GHG estimation approach can help cities achieve their climate and sustainability goals. Environ. Res. Lett. *16*, 084003.

73. Yadav, V., Ghosh, S., Mueller, K., Karion, A., Roest, G., Gourdji, S.M., Lopez-Coto, I., Gurney, K.R., Parazoo, N., Verhulst, K.R., et al. (2021). The impact of COVID-19 on CO_2 emissions in the Los Angeles and Washington DC/Baltimore metropolitan areas. Geophys. Res. Lett. *48*, e2021GL092744.

74. Staufer, J., Broquet, G., Bréon, F.-M., Puygrenier, V., Chevallier, F., Xueref-Rémy, I., Dieudonné, E., Lopez, M., Schmidt, M., Ramonet, M., et al. (2016). The first 1-year-long estimate of the Paris region fossil fuel CO_2 emissions based on atmospheric inversion. Atmos. Chem. Phys. *16*, 14703–14726.

75. Mitchell, L.E., Lin, J.C., Bowling, D.R., Pataki, D.E., Strong, C., Schauer, A.J., Bares, R., Bush, S.E., Stephens, B.B., Mendoza, D., et al. (2018). Long-term urban carbon dioxide observations reveal spatial and temporal dynamics related to urban characteristics and growth. Proc. Natl. Acad. Sci. USA *115*, 2912–2917.

76. Shusterman, A.A., Teige, V.E., Turner, A.J., Newman, C., Kim, J., and Cohen, R.C. (2016). The Berkeley atmospheric CO_2 observation network: initial evaluation. Atmos. Chem. Phys. *16*, 13449–13463.

77. Kim, J., Shusterman, A.A., Lieschke, K.J., Newman, C., and Cohen, R.C. (2018). The BErkeley atmospheric CO_2 observation network: field calibration and evaluation of low-cost air quality sensors. Atmos. Meas. Tech. *11*, 1937–1946.

78. Turner, A.J., Kim, J., Fitzmaurice, H., Newman, C., Worthington, K., Chan, K., Wooldridge, P.J., Köehler, P., Frankenberg, C., and Cohen, R.C. (2020). Observed impacts of COVID-19 on urban CO_2 emissions. Geophys. Res. Lett. *47*, e2020GL090037.

79. Peiro, H., Crowell, S., Schuh, A., Baker, D.F., O'Dell, C., Jacobson, A.R., Chevallier, F., Liu, J., Eldering, A., Crisp, D., et al. (2022). Four years of global carbon cycle observed from the orbiting carbon observatory 2 (OCO-2) version 9 and *in situ* data and comparison to OCO-2 version 7. Atmos. Chem. Phys. *22*, 1097–1130.

80. Kiel, M., Eldering, A., Roten, D.D., Lin, J.C., Feng, S., Lei, R., Lauvaux, T., Oda, T., Roehl, C.M., Blavier, J.-F., et al. (2021). Urban-focused satellite CO_2 observations from the orbiting carbon observatory-3: a first look at the Los Angeles megacity. Remote Sens. Environ. *258*, 112314.

81. Liu, F., Duncan, B.N., Krotkov, N.A., Lamsal, L.N., Beirle, S., Griffin, D., McLinden, C.A., Goldberg, D.L., and Lu, Z. (2020). A methodology to constrain carbon dioxide emissions from coal-fired power plants using satellite observations of co-emitted nitrogen dioxide. Atmos. Chem. Phys. *20*, 99–116.

82. Biden Whitehouse. (2021). Executive order on protecting public health and the environment and restoring science to tackle the climate crisis. https://www.whitehouse.gov/briefing-room/presidential-actions/2021/01/20/executive-order-protecting-public-health-and-environment-and-restoring-science-to-tackle-climate-crisis/?utm_source=link.