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Clearing the air: incorporating air quality and environmental justice into climate policy

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Abstract In addition to lower carbon dioxide emissions, policies to reduce fossil fuel combustion can yield substantial air quality co-benefits via reduced emissions of co-pollutants such as particulate matter and air toxics. If co-pollutant intensity (the ratio of co-pollutant impacts to carbon dioxide emissions) varies across pollution sources, efficient policy design would seek greater emissions reductions where co-benefits are higher. The distribution of co-benefits also raises issues of environmental equity. This paper presents evidence on intersectoral, intrasectoral and spatial variations in co-pollutant intensity of industrial point sources in the United States, and discusses options for integrating co-benefits into climate policy design to advance efficiency and equity.

1 Introduction

A central objective in climate policy is to reduce the burning of fossil fuels—coal, oil, and natural gas—so as to curb emissions of carbon dioxide (CO_2). In addition to climate benefits, this generates potential air quality “co-benefits” in the form of reduced emissions of nitrogen oxides (NO_x), sulfur dioxide (SO_2), particulate matter (PM), and other hazardous air pollutants generated in fossil fuel combustion.

Collectively, these associated emissions are termed “co-pollutants.” The importance of such co-pollutants can be seen in the contrast between two facilities in California which each emit 2.5–3 million tons per year (t/year) of CO_2. One, the La Paloma power plant, is a natural gas-fired electricity-generation facility in the Central Valley that emits about 50 t/year of particulate matter (PM) and has fewer than 600 residents living in a 6-mile (9.7 km) radius (6 miles is the buffer that California regulators use to assess environmental justice issues in power-plant siting). The other is the ExxonMobil petroleum refinery in Torrance, which

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emits about 350 t/year of PM and has about 800,000 residents living within a 6-mile radius (Pastor et al. 2010). Clearly, the co-benefits for population health associated with emission reductions in the two facilities are likely to be different.

The inclusion of co-benefits provides a rationale for more ambitious emissions reduction targets than an exclusive focus on CO_2. Moreover, if co-pollutant intensity—defined as co-benefits per unit of CO_2 emissions—varies across sources, there is an efficiency rationale for designing policies to achieve greater reductions where co-benefits are higher. Insofar as co-pollutants tend to be concentrated in economically and socially disadvantaged communities, there is also an equity rationale for incorporating them into policy design.

The overall magnitude of air quality co-benefits, measured in terms of human health impacts and their economic valuation, has received considerable attention. However, relatively little research has examined variations in co-pollutant intensity across industrial sectors, across facilities within sectors, or across spatial jurisdictions. This paper reviews evidence on co-pollutant intensity for industrial point sources in the United States, highlighting distributional as well as efficiency concerns, and discusses options for incorporating co-benefits into climate policy. To our knowledge, it is the first study to empirically analyze these variations on a national scale.

2 Policy significance of air quality co-benefits

A large number of studies on the magnitude of air quality co-benefits associated with climate policy have concluded that they are likely to be significant. In a review of more than a dozen studies from multiple locations, including industrialized and developing countries, Bell et al. (2008) concluded that there is strong evidence that the health co-benefits of policies to reduce greenhouse gas (GHG) emissions are “substantial,” and that the results are “likely to be underestimates because there are a number of important unquantified health and economic endpoints.” In a survey of 37 studies from around the world, Nemet et al. (2010) found a mean co-benefit of \$49 per ton of CO_2 (tCO_2). By way of comparison, the U.S. government’s Interagency Working Group on the Social Cost of Carbon (2010) put the “central value” of the benefit of climate-change mitigation at \$19/tCO_2.

In a study of emissions reductions in the European Union conducted for the Netherlands Environmental Assessment Agency, Berk et al. (2006) found that the health co-benefits of a “stringent climate change policy scenario” in the European Union would be sufficient alone to offset the policy’s costs. “The welfare effects of climate policy seem to be positive,” they conclude, “*even when the long-term benefits of avoided climate impacts are not taken into account*” (emphasis added).

The National Academy of Sciences (2009) estimates that air pollution from the burning of fossil fuels in the United States is responsible for roughly 20,000 premature deaths each year, translating into \$120 billion per year in health damages. Muller et al. (2011) estimate that air pollution health damages from coal-fired electricity generation in the United States—not including damages from CO_2 emissions—exceed the industry’s value added. Muller (2012) reports that the average co-pollutant damage per ton of CO_2 emissions from bituminous coal-burning electric generation units is 34 times higher than from natural gas-fired units. Observing that a small number of facilities account for a large fraction of the total damages, Muller concludes that if co-pollutants were incorporated into an optimal climate policy, almost two-thirds of the welfare gain would come from just 1 % of pollution sources.

What does this imply for carbon pricing policies? The air quality co-benefits of emissions reduction warrant higher prices than those that would be based on CO_2 emissions alone. If co-benefits vary across sources, then efficiency requires more emissions reductions where the co-benefits are greater. Hence a policy that ignores co-benefits would be inefficient in two respects: it would choose suboptimal emissions reduction targets overall, and it would fail to take account of differences in abatement benefits across emission sources.

A substantial body of literature has found that in the United States, racial and ethnic minorities and low-income communities tend to bear disproportionate pollution burdens (Szasz and Meuser 1997; Pastor 2003; Ringquist 2005; Mohai 2008; Morello-Frosch et al. 2011). In an analysis of the demographic correlates of exposure to PM and NO_x from 146 California power plants, petroleum refineries, and cement plants, Pastor et al. (2010, 2013) find that minorities are more likely than non-Hispanic whites to reside in close proximity to these facilities, even when controlling for household income, and more likely to reside near facilities that pose greater co-pollutant burdens. This suggests that incorporating air quality co-benefits into climate policy could advance the goal of environmental equity as well as efficiency.

Even where co-pollutants are already regulated, the possible co-benefits from further reductions in emissions may be substantial. Furthermore, many co-pollutants are still not regulated. Fewer than half of the chemicals reported in the USEPA's annual Toxics Release Inventory, for example, are subject to USEPA restrictions on point-source emissions. In cases where regulatory standards do exist, some facilities are exempted by virtue of their vintage (having been built prior to introduction of regulations), size (small), or location (in areas that meet overall air-quality standards). For instance, coal-fired power plants that together account for 37 % of U.S. capacity still had no emissions control equipment in place as of 2010 (Credit Suisse 2010, p. 20).

3 Measuring co-pollutant intensity

In measuring co-pollutant intensity, here defined as the ratio of co-pollutant impacts to CO_2 emissions, two issues must be addressed: the choice of the numerator and the breadth of the measure.

In choosing the numerator, we must consider which co-pollutant(s) to analyze, the method of aggregation when multiple pollutants are combined into a single measure, and the units in which impacts are expressed. The choice of co-pollutants is driven by policy relevance and data availability. The simplest way to aggregate multiple pollutants is to add their mass, but a shortcoming of this method is that some chemicals are more hazardous pound for pound than others. To address this problem, one can assign weights to different chemicals based on their relative toxicity.

In addition, policy analysts may want to take into account the number of people impacted. For point-source emissions, a straightforward way to do so is to use census data on the number of people living within a given radius of a facility (Mohai and Saha 2006). However, pollutants are not distributed uniformly within the resulting areas. Fate-and-transport models, such as the Risk-Screening Environmental Indicators used in this paper for air toxics and the Air Pollution Emission Experiments and Policy model used by Muller and Mendelsohn (2009), address this problem by taking into account stack heights, exit gas velocities, prevailing winds, and chemical decay rates in estimating exposure levels.

A further possibility, not attempted here, would be to monetize the costs of morbidity and premature mortality (for discussion, see Dorman 1996; Viscusi and Aldy 2003; Landrigan 2012). Monetization allows policymakers to compare costs and benefits of abatement for individual pollutants. As an aggregation method, this simply uses alternative units to express toxicity-weighted population exposure (unless the health and lives of different individuals are valued differently). A monetized measure could also incorporate ecosystemic and other impacts.

In this paper we report co-pollutant intensities for air toxics using three aggregation methods—simple mass, toxicity-weighted mass, and toxicity-weighted population exposure—to illustrate the sensitivity of results to this choice. We do not attempt to incorporate pollution impacts apart from human health, nor do we seek to aggregate criteria air pollutants along with air toxics into a single measure. We view these as important areas for further research.

The narrowest measure of co-pollutants would refer solely to emissions generated by fossil fuels combustion. A slightly broader definition would include combustion emissions attributable to fuel additives. A still broader definition would include additional emissions generated in the production and use of fossil fuels. The magnitude of the difference between narrow and broad measures varies across co-pollutants and industrial sectors. In the case of electrical power generation, for example, the difference is minor since most co-pollutants are generated by fuel combustion itself. In cement manufacturing, on the other hand, substantial PM emissions result from physical attrition of raw materials as well as from fuel combustion.

In practice, the breadth of the measure may be driven by data availability. Here we report broad measures of co-pollutant intensity, since the available data for U.S. industrial facilities refer to total emissions rather than emissions from fossil-fuel combustion alone.

4 Data

Collection of data on CO_2 emissions has been largely independent of collection of data on co-pollutants. The construction of co-pollutant intensity measures therefore requires merging data from multiple sources, a task complicated by differences in coverage, definitions, identification codes, and timing. In this section we discuss the datasets we use and the process for merging them.

4.1 CO_2 emissions

In January 2012 the USEPA's Greenhouse Gas Reporting Program (GHGRP) released the first inventory of GHG emissions from large industrial facilities in the United States (see http://www.epa.gov/ghgreporting/ghgdata/2010data.html). The data cover emissions of CO_2 and five other GHGs in the year 2010 from facilities that together accounted for more than half of total U.S. emissions.

More fragmentary data on CO_2 emissions from industrial facilities were reported previously in the 2008 edition of the USEPA's National Emissions Inventory (NEI), which relies primarily on data provided by state and local authorities (see http://www.epa.gov/ttnchie1/net/2008inventory.html). For 520 facilities we can compare 2008 CO_2 emissions recorded in the NEI to 2010 emissions recorded in the GHGRP. The correlation is 0.97, suggesting that year-to-year variations in CO_2 emissions at the facility level typically are small. This is important since our data on co-pollutants pertain to different years.

4.2 Criteria air pollutants

The Clean Air Act requires the USEPA to set National Ambient Air Quality Standards (NAAQS) for pollutants considered harmful to public health and the environment. In response, the USEPA has developed standards based on health and environmental criteria for particulate matter (PM), sulfur dioxide (SO_2), nitrogen oxides (NO_x), and several other air pollutants. Facility-level emissions of these "criteria air pollutants" are reported by the NEI. A limitation of these data is that they are not produced annually. Here we use 2008 NEI data on emissions of NO_x, $PM_{2.5}$ (particulate matter with a diameter of 2.5 micrometers or less) and SO_2.

Since the NEI does not include fate-and-transport modeling, we calculated population-weighted indices by multiplying tons of emissions of these co-pollutants by the number of people residing within 2.5 miles of the facility, a radius often used to define proximity in environmental justice studies (Pastor et al. 2013). For this purpose, we included the population counts for census block groups whose centroids were located within 2.5 miles of the facilities, using data from the 5-year American Community Survey (2005–2009). The effects of population weighting were similar across the three co-pollutants: to conserve space, here we report only the population-weighted results for $PM_{2.5}$.

Proximity serves as a relatively crude basis for measurement of population exposure for several reasons. The concentration-distance relationship varies by pollutant; for example, the gradient is steeper for NO_x than for $PM_{2.5}$ (Karner et al. 2010). Moreover, stack heights, exit gas velocities, prevailing wind patterns, and local topography affect the spatial distribution of pollutants. Facility-specific fate-and-transport modeling for criteria air pollutants would allow more precise measurement of population-weighted health risks and co-pollutant intensity.

4.3 Air toxics

Air toxics, also known as hazardous air pollutants, are chemicals that are not subject to NAAQS but are also known or suspected to cause serious health or environmental effects. The principal source of data on air-toxics emissions in the United States is the Toxics Release Inventory (TRI), which requires industrial facilities to report annual emissions of hundreds of toxic chemicals into air, water, and land.

The USEPA's Risk-Screening Environmental Indicators (RSEI) augments TRI data with information on the relative toxicity of TRI chemicals, fate-and-transport modeling of chemical releases, and the population densities in each of roughly 10,000 one-km^2 grid cells around each facility. The RSEI data for the year 2007, used here, include inhalation toxicity weights for 417 TRI chemicals and chemical compound groups (see http://www.epa.gov/oppt/rsei/pubs/get_rsei.html). As noted above, we compare three different aggregation methods in measuring air-toxics co-pollutant intensity: simple mass (pounds); toxicity-weighted mass ("hazard" in RSEI parlance); and total estimated human health impact ("score" in RSEI parlance).

4.4 Constructing the dataset

Matching facility-level data across these three USEPA databases is complicated by inconsistencies among facility identifiers. The GHGRP and RSEI sometimes use different Federal Registration System Identifiers for the same facility, and the NEI uses Emissions Inventory System Identifiers instead. To match facilities across databases, we supplemented these identifiers with information on facility names, addresses, and latitude-longitude coordinates; for methodological details, see Boyce and Pastor (2012).

Since our primary aim is to consider policy options with regard to major CO_2 emitters, we limit our analysis to industrial sectors in which the GHGRP data indicate that the average facility emits at least 50,000 tons of CO_2 annually, and to individual facilities that emit at least 50,000 tons annually in the case of power plants and at least 10,000 tons annually in the case of other industries. Our resulting database consists of 1,542 facilities that together account for 66 % of the total CO_2 emissions reported in the GHGRP database. Ninety-five percent of the facilities are in eight industrial sectors, with at least 30 facilities each. To avoid small-sample biases, when we examine co-pollutant intensity variations across industrial sectors, we restrict our analysis to this subset of sectors.

5 Results

5.1 Intersectoral variations

Average co-pollutant intensities for the eight industrial sectors vary markedly, as reported in Table 1. These differences imply corresponding intersectoral variations in the co-benefits of CO_2 emissions reductions. For example, the average co-pollutant intensity for air toxics, aggregated by the RSEI full-model score, is ten times higher for the petroleum refineries sector than for electrical power plants, and for population-weighted $PM_{2.5}$ it is three times higher. This implies that *ceteris paribus*, it would be desirable to achieve greater CO_2 emissions reductions at refineries.

The pattern of intersectoral variations is not uniform across co-pollutants. Primary metal manufacturers rank fairly low in NO_x, for example, but fairly high in $PM_{2.5}$, while for power plants the reverse is true. This means that estimates of co-benefits are sensitive to the choice of co-pollutants in the numerator.

A look at the three air toxics (RSEI) measures illustrates the importance of aggregation method for multiple pollutants. Transportation equipment manufacturers top the list in terms of simple mass, that is, pounds of air toxics per ton CO_2. Chemical manufacturers top the list, however, in toxicity-weighted pounds. In the full-model score, which also accounts for fate-and-transport and population densities, transportation equipment manufacturers again top the list. This sector also has the highest population-weighted $PM_{2.5}$ intensity.

5.2 Intrasectoral variations

Co-pollutant intensity varies substantially among facilities *within* the same sector, as well. Table 2 reports the coefficient of variation across facilities within sectors and across all facilities combined for each of our seven measures of co-pollutant intensity. A higher coefficient of variation indicates more variability.

Why is intrasectoral variation important? If co-pollutant intensity were fairly constant across facilities within any given industry, then an industry-by-industry policy approach could be reasonably efficient. If co-pollutant intensity is highly variable across facilities within the same industry, however, an approach designed to ensure greater GHG reductions from specific facilities where co-benefits are higher could be more efficient.

As can be seen, power plants show a relatively high degree of variation in several measures, a pattern partly explained by differences between coal-fired and natural gas-fired plants. On the other hand, refineries show the lowest degree of variation for both the RSEI full-model score and population-weighted $PM_{2.5}$. This suggests that in the case of refineries a sector-wide approach to carbon pricing (such as regulating trades into and out of the sector under a cap-and-trade system) may be reasonably efficient.

With the possible exception of the refinery sector, the relatively large intrasectoral variations in co-pollutant intensities imply a rationale for designing climate policy to steer CO_2 reductions toward particularly high-intensity facilities. To shed light on the importance of such high-intensity facilities, we calculate the share of total emissions that is attributable to the top 1 % of facilities. We also calculate a Gini index in which 0 would indicate that each facility emits the same quantity of co-pollutants and 1 would indicate that all co-pollutants are emitted from a single facility.

Table 1 Co-pollutant intensity ratios by industry

Industry	Mean CO2 output (thousand tons)	NOx (pounds/tCO2)	PM2.5 (pounds/tCO2)	SO2 (pounds/tCO2)	RSEI chemicals (pounds/tCO2)	RSEI toxicity-weighted (pounds/tCO2)	RSEI full model score (per tCO2)	Population-weighted PM2.5 (per tCO2)
Power plants	3,680	2.97	0.33	8.38	0.34	119	0.006	2,994
Petroleum refineries	1,170	1.31	0.32	1.72	0.20	1,091	0.059	8,414
Primary metal manufacturers	382	1.74	1.04	5.02	0.39	2,864	0.200	19,707
Chemical manufacturers	348	2.45	0.49	4.32	1.02	10,300	0.244	5,198
Nonmetallic mineral product manufacturers	325	6.52	0.72	3.55	0.29	211	0.066	8,634
Paper mills	220	9.51	1.74	13.69	3.30	2,397	0.056	10,886
Food manufacturers	215	2.33	0.52	5.53	1.16	1,033	0.034	13,132
Transportation equipment manufacturers	54	4.13	0.90	2.11	4.28	2,883	9.507	53,359
Average (unweighted)	799	3.87	0.76	5.54	1.37	2,612	1.272	15,291

Table 2 Coefficient of variation for co-pollutant intensity by industrial sector

	NOx (pounds/tCO2)	PM2.5 (pounds/tCO2)	SO2 (pounds/tCO2)	RSEI chemicals (pounds/tCO2)	RSEI toxicity-weighted (pounds/tCO2)	RSEI full model score (per tCO2)	Population-weighted PM2.5 (per tCO2)
All facilities	2.51	4.19	4.97	3.49	9.72	12.95	3.88
Power plants	2.98	10.19	4.81	4.28	4.31	4.86	4.03
Petroleum refineries	1.03	1.42	2.61	3.64	2.76	2.45	1.97
Primary metal manufacturers	0.95	2.52	5.53	6.54	4.96	4.46	2.10
Chemical manufacturers	1.98	2.33	4.18	3.21	4.86	6.67	3.67
Nonmetallic mineral product manufacturers	1.44	2.59	1.36	4.05	4.83	7.75	2.89
Paper mills	2.21	1.48	1.43	1.60	2.12	7.03	2.11
Food manufacturers	1.02	1.22	1.41	1.40	3.99	3.52	2.38
Transportation equipment manufacturers	1.79	2.65	2.66	1.17	1.31	6.03	5.24

Table 3 reports both indicators. In the case of air toxics as measured by the RSEI score, for example, the top 1 % of facilities (15 of the 1,542 facilities in the sample) accounted for more than half of the impact; in the case of population-weighted $PM_{2.5}$, the top 1 % accounted for 35 % of the total impact. These results exhibit a high degree of disproportionality, a skewed distribution in which a small number of facilities have far greater impact than the sectoral average (Berry 2008; Freudenberg 2006). Specific policy attention to this small number of facilities could yield commensurately disproportional results.

5.3 Spatial variations

To examine spatial variations that might have implications for environmental justice, we use two measures of how co-pollutant burdens are distributed by race and income. The first

Table 3 Measures of facility-level co-pollutant and CO_2 concentration

Co-pollutant (absolute quantity)	Share for top 1 % of facilities	Gini index
NOX	13 %	0.76
PM2.5	26 %	0.78
SO2	23 %	0.86
RSEI pounds	22 %	0.80
RSEI toxicity-weighted pounds	44 %	0.90
RSEI full-model score	59 %	0.93
Population-weighted PM2.5	35 %	0.84
CO2	13 %	0.78

Table 4 Environmental justice by industry: air toxics

Industry	Black share	Hispanic share	Minority share	Poor share
Power plants	16.3 %	16.9 %	38.5 %	12.6 %
Chemical manufacturers	15.5 %	22.9 %	41.8 %	14.6 %
Nonmetallic mineral product manufacturers	8.8 %	7.3 %	22.7 %	10.4 %
Primary metal manufacturers	14.2 %	9.6 %	26.8 %	14.0 %
Paper mills	11.5 %	6.8 %	24.5 %	14.4 %
Petroleum refineries	24.8 %	20.8 %	50.3 %	16.3 %
Food manufacturers	28.5 %	3.0 %	34.4 %	14.3 %
Transportation equipment manufacturers	13.5 %	12.2 %	35.3 %	12.4 %
All sectors	15.7 %	14.4 %	36.3 %	13.6 %
U.S. population distribution, 2000	12.3 %	12.5 %	30.9 %	12.4 %

calculates how much of each facility's air toxics impact, as measured by their RSEI score, is borne by African Americans, Latinos, and all racial and ethnic minorities combined (including Asian-Pacific Islanders and Native Americans) and by households with incomes below the federal poverty line. For this purpose we use geographic microdata from the RSEI to track each facility's air toxics to specific neighborhoods, and then use census data on the demographics of those neighborhoods (Ash et al. 2009; Ash and Boyce 2011). The second measure, which we use for population-weighted $PM_{2.5}$, calculates the share of each group in the population living within 2.5 miles of the facility.

Tables 4 and 5 report the results for air toxics and population-weighted $PM_{2.5}$, respectively. If co-pollutant exposure were evenly distributed across racial, ethnic, and income groups, their impact shares would correspond to their shares in the national population. For comparison, the latter are reported in the final row of the tables (for 2000 in the case of air toxics because the USEPA uses 2000 Census data to calculate RSEI scores, and for 2005–2009 in the case of $PM_{2.5}$ because we used census-tract information for these years from the 5-year pooled American Community Survey (Ruggles et al. 2011).

In the case of air toxics, we find that petroleum refineries impose the most disparate burden on people of color, followed by chemical manufacturers and power plants. Refineries and chemical manufacturers also impose the most disparate burden on low-income people.

Table 5 Environmental justice by industry: population-weighted $PM_{2.5}$

Industry	Black share	Hispanic share	Minority share	Poor share
Power plants	13.5 %	17.9 %	38.8 %	15.8 %
Chemical manufacturers	24.4 %	15.8 %	43.9 %	21.1 %
Nonmetallic mineral product manufacturers	14.8 %	17.3 %	39.8 %	16.1 %
Primary metal manufacturers	35.4 %	9.2 %	47.5 %	23.2 %
Paper mills	17.7 %	4.9 %	27.2 %	17.9 %
Petroleum refineries	33.3 %	20.2 %	59.5 %	24.0 %
Food manufacturers	13.7 %	16.4 %	33.5 %	18.2 %
Transportation equipment manufacturers	11.0 %	29.6 %	44.3 %	28.4 %
All sectors	20.1 %	16.4 %	42.6 %	18.6 %
U.S. population distribution, 2005–2009	12.1 %	15.1 %	34.2 %	13.5 %

In the case of population-weighted $PM_{2.5}$, refineries again have the most disparate impacts on minorities. Six of the eight sectors disproportionately impact minorities, and all disproportionately impact the poor.

These findings suggest that any climate policy that fails to incorporate air-quality co-benefits into its design will not only forgo public health benefits, but also is likely to violate U.S. federal government directives to consider environmental equity in rule and decision making. This is mandated by Executive Order 12898, "Federal Actions to Address Environmental Justice in Minority Populations and Low-Income Populations," 1994, and most recently articulated in the USEPA's "Plan EJ 2014" (see http://www.epa.gov/environmentaljustice/plan-ej/index.html).

The clustering of industrial facilities in specific locations can exacerbate health disparities resulting from pollution exposure (Alexeeff et al. 2010). While clustering is not consequential in the case of CO_2 emissions, whose effects are global rather than local, it can be highly relevant in the case of co-pollutants.

Figure 1 illustrates the existence of clusters by mapping $PM_{2.5}$ emissions in Los Angeles, Houston, and Pittsburgh. The size of the facility icons in the map varies with the magnitude of population-weighted $PM_{2.5}$ emissions, while the darker shades in the neighborhood map insets indicate census block groups with higher numbers of proximate facilities. The patterns

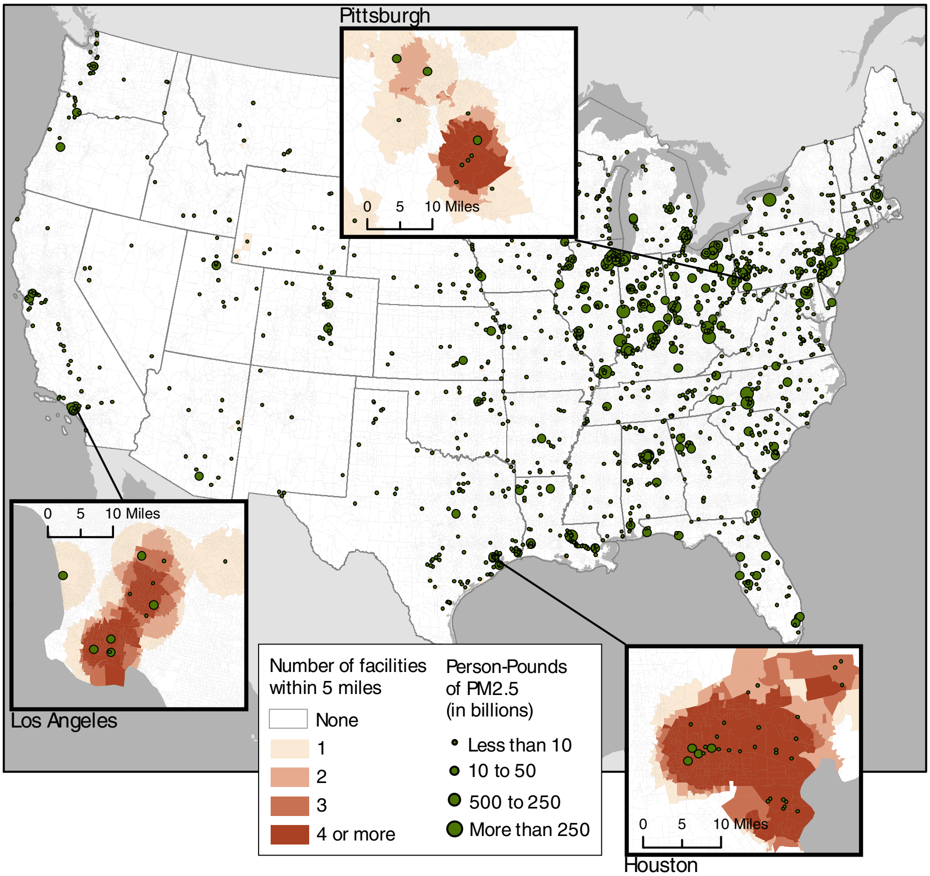

Fig. 1 Clustering of population-weighted $PM_{2.5}$ emissions

shown here are consistent with an earlier study of California's electric utilities, petroleum refineries and cement industries, which found that clustering results in significant cumulative PM impacts in some locations (Pastor et al. 2010, 2013).

6 Policy options

Two broad types of policies have been proposed to curb carbon emissions from fossil-fuel combustion: quantitative regulations, such as fuel economy standards and mandated technologies; and price-based policies, such as a carbon tax or marketed permits. These are not mutually exclusive, as illustrated in California where the state's Global Warming Solutions Act is being implemented via measures that include renewable portfolio standards for electricity, low-carbon fuel standards for transportation fuels, and a cap-and-trade program for CO_2 emissions.

A frequently noted strength of price-based policies is that they provide incentives not only to use available emissions-reducing technologies, but also to invest in research and development of new technologies (Burtraw 1996; Carlson et al. 2000). Price-based policies have met opposition, however, from some environmental justice proponents on the grounds that they can allow co-pollutant "hot spots" to persist or maybe even worsen in overburdened communities. Environmental justice advocates in California, for example, filed a lawsuit in an attempt to block implementation of the state's cap-and-trade program (Farber 2012).

Some have responded to these concerns by maintaining that co-pollutants are best regulated separately, and that they should not be considered when designing climate policy (Schatzki and Stavins 2009). We cannot safely assume, however, that co-pollutant impacts are or will be adequately addressed by other policies—and if air quality co-benefits vary across polluters, an efficient and equitable climate policy should take this into account. Such an approach is consistent with growing interest in multi-pollutant as opposed to single-pollutant strategies for air-quality management (National Academy of Sciences 2004; Napolitano et al. 2009; McCarthy et al. 2010).

The administrative costs of incorporating co-pollutants into carbon pricing policies could be modest, particularly where a small number of sectors or facilities present the greatest opportunities to achieve air quality co-benefits. The evidence presented above suggests that this is the case for industrial point sources in the United States.

Options to incorporate air quality co-benefits into climate policy include the following:

1. *Monitor impacts on co-pollutants*: A minimalist option is to monitor co-pollutant emissions, with a view to adopting remedial measures if the climate policy has unacceptable impacts. This is the approach taken by the California Air Resources Board (2011) in its adaptive management plan for the state's cap-and-trade policy. Such monitoring is particularly important for high-emissions sectors and facilities and in the most polluted neighborhoods. Policymakers could then respond to failures to reduce co-pollutant emissions as well as to increases in emissions (as could happen, for example, if closure of coal-fired power plants reduces pressure on the existing SO_2 emissions cap, allowing other facilities to emit more SO_2).
2. *Zonal tax or permit systems*: Zonal tax or permit systems can be designed to ensure emissions reductions in high-priority locations where potential public health benefits are greatest. Polluters in high-priority zones would face a higher carbon tax or permit price and therefore have a stronger incentive to reduce emissions. If the policy allows permit trading, they would be barred from buying out of emissions reduction by purchasing offsets or permits from other localities. Such zone-specific caps were established, for

example, in California's Regional Clean Air Incentives Market, which was initiated in 1994 to reduce point-source emissions of NO_x and SO_2 in the Los Angeles basin (Gangadharan 2004).

3. *Sectoral tax or permit systems*: Similarly, sectoral tax or permit systems can be designed to ensure emissions reductions in high-priority sectors. Our findings suggest that the petroleum refinery and chemical manufacturing sectors have above-average co-pollutant intensities of toxicity-weighted and population-weighted emissions, and that their emissions rank high in disproportionate impacts on minorities and low-income communities. Sector-specific caps on carbon emissions can be used to ensure reductions in such sectors that are greater than, or at least equal to, those mandated by the overall cap. Alternatively, carbon tax rates can be calibrated to reflect intersectoral differences in air quality co-benefits.
4. *Trading ratios*: In a tradable permit system where damages per unit of emissions vary across sources, the exchange rate at which permits are traded can be another policy instrument. For example, if total (CO_2 + co-pollutant) damages per ton CO_2 are twice as high in location A as in location B owing to higher co-pollutant damages in location A, the exchange rate (or "trading ratio") would equate one permit in location A to two permits in location B (Tietenberg 1995; Muller and Mendelsohn 2009). A similar policy can be applied to trades between sectors.
5. *Community benefit funds*: Pricing proposals often envision that some fraction of the revenue from carbon taxes or permit auctions ("carbon rent") will be channeled into public investments. Part of this investment can be allocated to community benefit funds to mitigate co-pollutant impacts and protect public health in vulnerable communities (Prasad and Carmichael 2008). In September 2012, California governor Jerry Brown signed into law such a policy for revenues from permit auctions under the state's climate policy. Community eligibility can be determined by screening methods that take into account hazard proximity, local air quality, climate change vulnerability, and other measures of social vulnerability (Sadd et al. 2011).
6. *Regulatory instruments*: Finally, conventional regulatory instruments, such as emissions standards or mandated technologies, can be used to ensure emissions reductions in specific locations, sectors, or facilities that are assigned priority due to their high co-pollutant intensities or environmental justice impacts.

7 Conclusions

Studies of the magnitude of air quality co-benefits from climate policy have concluded that they are large enough to warrant policy attention. Indeed, by some measures these co-benefits are as large as the climate benefits that constitute the primary rationale for the policy. To integrate co-pollutants into climate policy design, policymakers need information on source-wise variations in co-pollutant intensity—the ratio of co-pollutant damages to CO_2 emissions.

Evidence on variations across industrial point-source emitters indicates that co-pollutant intensity varies substantially. This finding implies that a one-size-fits-all approach to climate policy, in which emissions reductions are treated as equivalent regardless of where they occur, is less than optimal. From the standpoint of efficiency, policy should aim for greater reductions where air quality co-benefits are larger.

Differentiation across pollution sources that takes these variations into account can serve the goal of environmental equity, too, since low-income and minority communities often bear disproportionate co-pollutant burdens. Federal directives mandate that environmental justice considerations enter into the design of environmental policy and implementation. Any carbon tax or permit policy that gives polluters the choice to "clean up or pay up" should ensure that their decisions fully reflect the benefits of clean up, including the public health and environmental justice implications of co-pollutant emissions.

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