Cooling the Planet, Clearing the Air: Climate Policy, Carbon Pricing, and Co-Benefits

James K. Boyce and Manuel Pastor

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Written by James K. Boyce and Manuel Pastor

ABSTRACT: Policies to reduce carbon dioxide emissions can yield substantial co-benefits via reduced emissions of co-pollutants such as particulate matter, nitrogen oxides, and air toxics. Valuation studies suggest that these benefits may be comparable in magnitude to the value of reduced carbon emissions. However, co-pollutant intensity (the ratio of co-benefits to carbon dioxide emissions) varies across pollution sources, and so efficient policy design would seek greater emissions reductions where co-benefits are higher. Moreover, because co-pollutant impacts are localized, the distribution of co-benefits raises important issues of equity, particularly with regard to the unintentional income, racial, and geographic disparities that might result from carbon-charge programs, whether they are trading or fee approaches. This paper presents evidence on intersectoral and spatial variations in co-pollutant intensity and discusses options for integrating co-benefits into climate policy to advance the goals of efficiency and equity.

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Communities of color have a tremendous stake in efforts to reverse climate change and mitigate its impacts. They are among the first to experience the effects of climate disruption, which can include "natural" disasters, rising levels of respiratory illness and infectious disease, and heat-related sicknesses and deaths. They face greater risk from ill-conceived solutions to climate disruption, and they stand to gain considerable health and economic benefits from policies that are carefully constructed to maximize and broaden the impacts of addressing climate change and pollution.

Indeed, these were principal reasons why, in 2008, the Joint Center established a special commission to promote wider engagement and participation by African Americans in the climate change debate – a panel that has more recently become the Commission to Engage African Americans on Energy and the Environment. Over the years, we have worked with our commissioners and a broad range of partners to bring new voices to the table and advance climate and environmental policy discussions toward effective and equitable solutions.

One of our key objectives has been to build an evidentiary record to support these efforts, and this report – "Cooling the Planet, Clearing the Air: Climate Policy, Carbon Pricing, and Co-Benefits" – is an important step in furthering our knowledge and understanding. By highlighting the need for and opportunities to develop policies that both reduce emissions of harmful greenhouse gases and improve overall air quality, this report provides a roadmap for improving lives in communities of color. In particular, the evidence presented in this report outlines ways that policies to reduce carbon dioxide emissions can yield substantial additional benefits as co-pollutants such as particulate matter, nitrogen oxides and air toxics. Progress on these co-pollutants would yield additional positive health impacts for African Americans and other people of color, who are more likely than others to live near their point sources, and greatly increase the value of climate change policies, particularly in the short-term.

As this study shows, enormous progress could be made on reducing emissions of both greenhouse gases and other air pollutants with only modest adjustments to climate change mitigation approaches currently under consideration. We look forward to bringing this report into the climate change debate, and helping raise the level of awareness – within communities of color and among the broader population – about what can be done in the climate change framework to ensure a clean and healthy environment for all our citizens.

Ralph B. Everett, Esq.
President, The Joint Center for Political and Economic Studies
September 20, 2012

ECONOMICS FOR EQUITY & ENVIRONMENT

Today's challenges demand new economic thinking. Economics for Equity and the Environment Network is dedicated to applied research and dissemination of new economic arguments for protecting human health and the environment.

For economists in E3 Network, environmental protection and social justice are inextricably linked. Yet there are many examples of environmental policies designed without full consideration of the implications for vulnerable populations. E3's latest report, Cooling the Planet, Clearing the Air: Climate Policy, Carbon Pricing, and Co-Benefits, produced in partnership with the Joint Center for Political and Economic Studies, the Political Economy Research Institute at the University of Massachusetts, and the Program for Environmental and Regional Equity at the University of Southern California, takes a fresh look at the relationship between social equity and economic efficiency in the design of climate policies.

Climate policy is very heavily focused on reducing carbon emissions. The same power plants and refineries that emit carbon, however, produce other pollutants that have immediate and direct impacts on the health of nearby residents. These point sources are often disproportionately located in low-income and minority communities. This report examines the inter-sectoral and spatial variations in the intensity of co-pollutants with important findings for how we approach carbon reduction. Failure to consider co-pollutants in carbon cap and pricing strategies can exacerbate existing disparities while leaving valuable health care dollars on the ground. We could lose substantial economic benefits by excluding co-pollutants from our carbon strategies, and those losses would fall disproportionately on the most vulnerable amongst us.

The good news is that we can account for the co-benefits of carbon reduction through modest adjustments in our approaches to carbon reduction. Doing so would greatly enhance the benefits of climate change policies, especially for communities of color and low-income communities. There is progress to be made cleaning the air and protecting the climate. This report provides the evidence and recommendations policy makers need to forge a more equitable and efficient approach to climate policy.

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September 20, 2012

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EXECUTIVE SUMMARY

Consider two emitters of greenhouse gases (GHGs) in California. One is a natural gas–fired electricity-generation facility in a rural area with no other major industrial facilities in the immediate vicinity. The other is a petroleum refinery in a densely populated urban center, with so many other adjacent pollution sources that the surrounding community is a poster child for what public-health researchers call "cumulative exposure." Each of these facilities, it turns out, emits roughly the same amount of carbon dioxide (CO_2)—but the refinery emits seven times more particulate matter (PM, a pollutant that leads to premature death, asthma, and other respiratory illnesses) and has hundreds of thousands more people living nearby.

Carbon-pricing strategies—in which polluters either are charged a set fee for carbon emissions or must surrender emissions permits whose total number is set by a cap and whose price is determined by the market—essentially treat these two sources as equal: a reduction of GHGs at one is the same as the reduction of GHGs at the other. However, the potential health benefits of reducing emissions of the various "co-pollutants" at these sources—particulates and other hazardous chemicals also emitted in the burning of fossil fuels—are very unequal. Where the emissions reductions occur can have dramatic effects on the number of people who benefit (or fail to benefit) from the ways that GHG reductions are coupled with other pollution cutbacks.

The failure of carbon-pricing strategies to consider co-pollutant externalities is a striking contradiction, since the point of such pricing is to build in the externality of global warming via a carbon charge. It is a source of inefficiency: potential health-care savings are left lying on the ground (or drifting in the air). And because point sources often are disproportionately located in low-income and minority communities, carbon pricing that does not take account of co-pollutants runs the risk of exacerbating existing disparities and thus running afoul of the nation's commitment to environmental justice.

This study explores the issue of co-pollutants and co-benefits in carbon-pricing policies and draws conclusions that are both disturbing and hopeful. The disturbing news is that significant benefits could be lost by failing to address this issue in designing climate policies, and these losses would fall disproportionately on more vulnerable communities. But there is also some important good news: the problems are concentrated in certain sectors and emitters, suggesting that a relatively modest set of market-constraining actions could yield big positive results.

Why Co-Benefits Matter

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 large. In fact, one study of carbon emissions reductions in the European Union found that "the welfare effects of climate policy seem to be positive even when the long-term benefits of avoided climate impacts are not taken into account." Several studies in the United States have also found that there are potentially large health gains apart from those that arise from curbing climate change, particularly through reductions in coal-based electrical power. Indeed, international data from the World Bank on damages from emissions of particulate matter—an air pollutant that poses serious health risks—suggest that co-pollutant damages per unit of carbon dioxide emissions in the United States, while lower than in newly industrializing countries, are notably higher than in a number of other high-income countries, including Germany, France, and Canada (see Appendix).

Perhaps as important as the scale of air-quality co-benefits is their immediacy. Although environmentalists may lament the failure of policy makers to think generations ahead, shifting the gains from climate policy forward in time can help to build political support to stay the course on tackling global warming. Indeed, in a recent California campaign to protect the state's landmark 2006 global-warming legislation against an initiative funded largely by oil refiners, advocates found that stressing the policy's immediate health benefits was highly persuasive, particularly among communities of color, who often feel the brunt of dirty air.

These communities also face what some have called a "climate gap"—a set of higher risks from climate change that run the gamut from a lack of shade cover (in "urban heat islands") to a sharper hit from rising energy costs to inadequate disaster preparedness (as evidenced during Hurricane Katrina and several recent heat waves).

One California study has shown that large GHG emitters are also disproportionately located in communities of color—even when controlling for differences in income. So there are good reasons to worry about what may occur when some facilities decide to clean up and others decide to buy out. Climate policy will bring about changes in the geographic location of co-pollutant burdens. There may, for example, be intrafacility technological changes that reduce (or capture and sequester) CO_2 emissions but increase emissions of co-pollutants. There are likely to be interfacility shifts, as in electricity generation when coal-fired plants decline in importance and natural-gas plants replace them. And there certainly could be intersectoral shifts, for example, between power plants and refineries, as in the stark example with which we began this executive summary.

One recent strand of literature suggests the differences between point-source facilities may not be all that important, because cancer risks from air toxics are driven primarily by mobile sources. Differences between point-source polluters—and which polluters choose to buy permits under cap-and-trade (or pay fees under a carbon tax) rather than cutting their emissions—will be a ripple in a larger ocean of air pollution. Of course, one person's ripple is another community's wave: in certain locations, stationary sources are quite important. But we also show in this study that the relative importance of point sources for neurological health effects and for particulate matter emissions is much higher than it is for the single measure of air-related cancer risk. Similarly, sulfur dioxide (SO_2) and nitrogen oxides (NO_X) emissions are more strongly associated with stationary sources. There is, in short, reason to be concerned about both the size of the effects from interfacility differences and the geographic and social inequalities that might result.

Taking the Measure of Co-Pollutant Burdens

Developing measures to gauge whether concern about co-pollutants in climate policy is not just theoretically interesting, but also empirically important is no easy task. The U.S. Environmental Protection Agency (USEPA) has finally assembled an inventory of GHG emitters across the country (under its Greenhouse Gas Reporting Program, or GHGRP), but the resulting data do not mesh readily with data on co-pollutants reported in the agency's National Emissions Inventory or with data on air toxics in its Risk-Screening Environmental Indicators (which take into account the inhalation toxicities of different chemicals and use a fate-and-transport model to analyze where such toxics end up and how many people they affect).

Going where most researchers have feared to tread—or, better put, sending out plucky graduate students as the initial scouts, and then enlisting them to grind through the mechanics of data assembly and Geographic Information System (GIS) mapping—we put together a unique data set that includes over 1,500 large facilities (which together account for two-thirds of the CO_2 emissions reported in the GHGRP) for which we have matched data on SO_2, NO_X, $PM_{2.5}$, and air toxics. For the first three co-pollutants we have the simple mass of emissions; for air toxics, we have not only the mass of emissions, but also the toxicity-weighted mass and a score that takes into account the size of the impacted population. We also report a proximity-based version of the population-impact measure for $PM_{2.5}$ (particulate matter with a diameter of 2.5 micrometers or less, also known as "fine particles," which are considered particularly hazardous, because they can penetrate deeply into the lungs).

Our first main finding is that co-pollutant intensity—the ratio of co-pollutant emissions, or damages, to carbon emissions—varies widely across pollutants, sectors, and firms. For example, power plants are responsible for nearly 80 percent of the CO_2 emissions in our sample, but for a lower share of $PM_{2.5}$ emissions and for a markedly smaller share of the toxicity-weighted air toxics emissions and their human health impacts. Petroleum refineries, in contrast, account for less than one-tenth as much carbon emissions as the power plants, yet they have roughly the same air-toxics health impact.

One can immediately see that any carbon-charge system in which refineries en masse buy their way out of cleanup and instead let all the emissions reductions come from power plants (or other sectors) would forego significant health benefits from reducing co-pollutants. This concern is heightened when we carry out an analysis of variations in co-pollutant intensity within industrial sectors. If facilities within a particular sector are all over the map, not just with regard to geography but also in their co-pollutant intensities, it could be best to go plant by plant in analyzing the health and equity impacts of climate policies such as cap-and-trade. As it turns out, refineries have the lowest variance, suggesting once again that this industry is of particular concern.

At the same time, it is important to look for outlier co-pollutant emissions producers. For example, in our sample, the top 1 percent of SO_2 polluters are responsible for nearly one-quarter of the SO_2 emissions. The top 1 percent of the population-weighted $PM_{2.5}$ producers are responsible for over one-third of the total. In general, we find high levels of disproportionality, in which some facilities are far more problematic than the "typical" facility. This is an important finding, because it suggests that specific policy attention to a small number of "bad actors"—bad in the sense of high co-pollutant impacts by virtue of the quantity and toxicity of their emissions and their proximity to vulnerable populations—could likewise yield disproportionately positive results.

Benefits and Burdens

In a carbon-pricing policy, such as the cap-and-trade system now being developed by the state of California, the cap offers widely shared benefits with regard to GHG emissions—no matter where you live, virtually everyone gains from climate protection. On the other hand, the effects are unequal with regard to co-pollutants—some places will see more reductions in, say, $PM_{2.5}$ emissions than other places. This is inherent in a policy that gives polluters the option of paying to pollute rather than reducing their emissions.

One key question is whether there are systematic patterns of inequity in the distribution of co-pollutant burdens by salient socioeconomic characteristics such as race, ethnicity, and income. To get at this, we looked at the share of $PM_{2.5}$ and air-toxics burdens borne by different demographic groups by industrial sector. If co-pollutant exposures were evenly distributed across all racial, ethnic, and economic groups, these shares would correspond to their respective shares in the national population.

In the case of air toxics, we find disproportionate exposures for African Americans in most sectors (the exceptions are nonmetallic mineral product manufacturing and paper mills), with particularly high shares of exposure in the petroleum-refining sector. Latinos are disproportionately burdened in four of the eight sectors, with chemical manufacturing and petroleum refining topping the list and power plants not far behind. Overall, petroleum refineries pose the most disparate air-toxics burden on people of color. They also pose the most disparate burden on the poor.

Refineries also top the list for disproportional impacts on minorities in the case of population-weighted $PM_{2.5}$ emissions and rank second in disparate impacts on the poor. In only two sectors (paper mills and food manufacturers) is the minority share of the co-pollution burden less than the minority share of the population—and there is no sector in which the share of the poor in the burden is less than their share in the population.

Comparing these rankings to the sectoral sources of carbon emissions, we find that the three industrial sectors that produce the most carbon emissions—power plants, refineries, and chemical manufacturing, which together account for more than 90 percent of industrial CO_2 emissions in our sample—also have the most environmentally inequitable impacts on minorities with regard to the air-toxics measure and rank in the top five in population-weighted $PM_{2.5}$. Any climate policy that reduces co-pollutants along with GHG emissions, therefore, is likely to reduce environmental disparities and thereby advance environmental justice objectives. By the same logic, any regulatory program that sacrifices air-quality co-benefits not only will forgo public health savings, but also is likely to violate the official federal directives to consider environmental equity in rule and decision making.

Many industrial facilities are clustered together. Such clustering of CO_2 emitters is not consequential with regard to carbon—again, wherever you reduce a certain amount of carbon emissions, whether from a single industrial facility or from a group of facilities, the effect on climate change is the same. On the co-pollutant side, however, clustering can matter a great deal: if a cluster of facilities reduces its pollution rather than, say, buying emission allowances or offset credits, then the neighborhood would find its overall air quality substantially improved.

Where facilities cluster, the share of overall cancer and neurological risk from industrial point sources rises dramatically. This is particularly pronounced in three clusters we map in detail—in Houston, Los Angeles, and Pittsburgh—but our analysis of the data overall suggests that large GHG-emitting facilities are likely important contributors to the health risk of their residential neighbors. This spatial analysis provides further insight into the equity impacts of climate policy.

Looking Forward

In our view, there is a strong case for integrating co-pollutants into climate-policy design on both efficiency and equity grounds. From an efficiency standpoint, failure to account for variations in air-quality co-benefits across carbon emission sources is tantamount to leaving health-care dollars lying on the floor. From an equity standpoint, co-pollutant burdens lie at the critical interface between climate policy and environmental justice.

Our recommendations include suggestions for improving the informational basis for policy making and for how to incorporate co-pollutant impacts into climate-policy design. The recommendations are summarized in Tables 1 and 2, respectively.

Table 1: Recommendations to Improve Information for Policy Design

Policy	Summary
Co-pollutant monitoring	Climate policy implementation should be accompanied by monitoring of co-pollutant emissions. Remedial policies should be introduced if monitoring reveals the widening of disproportional co-pollutant impacts on low-income communities and minorities.
Synchronize facility identification codes	Databases of the U.S. Environmental Protection Agency and other government agencies should include a consistent set of IDs for industrial facilities to improve the ability of researchers to analyze co-pollutant emissions in relation to carbon emissions.
Develop aggregate measures of co-pollutant impacts	Measures of co-pollutant impacts should be developed on a more granular neighborhood level, using fate-and-transport modeling of population exposures for criteria air pollutants for areas where monitoring is sparse, and combining this information with fate-and-transport models for air toxics.
Environmental justice screening	Environmental justice screening tools for the identification of disadvantaged communities should be developed to incorporate information on vulnerability to climate change.
Extend data collection and analysis to non-industrial sources of pollution	Spatial variation in the co-pollutant burdens posed by mobile sources, such as motor vehicles and aircraft, and by small point sources, such as dry cleaners and gas stations, should be analyzed, too.

Table 2: Recommendations to Improve Policy Outcomes

Policy	Summary
Strengthen carbon emission reduction targets	Air quality co-benefits should be counted in setting policy objectives for carbon emission reduction.
Designate high-priority zones	Climate policy design should include identification of high-priority zones where air quality co-benefits are especially large. Policy should ensure that emission reductions in these zones equal or exceed the average reductions achieved by the policy as a whole.
Designate petroleum refineries and chemical manufacturers as high-priority sectors	These two industrial sectors not only account for substantial carbon emissions but also for disproportionate shares of overall co-pollutant burdens and impacts on minorities and low-income communities. Policy should ensure that emission reductions in these sectors equal or exceed the average reductions achieved by the policy as a whole.
Designate high-priority facilities	Industrial facilities that rank in the top 5% in co-pollutant emissions should be designated as high-priority facilities for carbon emission reductions. Policy should ensure that emission reductions at these facilities equal or exceed the average reductions achieved by the policy as a whole.
Allocate a share of carbon revenues to community benefit funds	Part of the carbon rent generated by price-based climate policy instruments that is devoted to public investments should be allocated to community benefit funds to support environmental and public health improvements in disadvantaged communities.

Some proponents of a cap-and-trade approach to GHG reduction have argued against the integration of co-pollutants into climate policy, contending that co-pollutants are already regulated and that bringing them into a climate regime would unnecessarily complicate the construction of a new market and generate too many targets from a single policy. Although the existing regulatory regime may affect the extent of co-benefits, this does not mean that co-benefits from climate policy are inconsequential or irrelevant to climate-policy design. Moreover, the fact that co-benefits may be concentrated in certain key sectors and facilities suggests that administrative efficiencies are possible.

Another false dilemma is that posed between market and nonmarket mechanisms for pollution control. The climate debate has often focused on price-based policies, such as marketed permits or a carbon tax, partly because this is a new and intellectually exciting policy arena and partly because some are eager to capture the efficiencies and innovation incentives that price-based policies could provide. It is important to recognize, however, that quantitative controls on CO_2 emissions will be important too and are complementary; indeed, the bulk of the emissions reductions from California's Global Warming Solutions Act of 2006 (AB 32) will come from policies like the renewable portfolio standard for electricity supply and low-carbon fuel standard for transportation fuels, with a smaller share coming from its cap-and-trade component. That said, some part of emissions reduction in California will come from carbon-pricing strategies, and in the future these could play an even larger role nationwide.

Measures that could enhance efficiency and equity as carbon-pricing policies are adopted include the following (with one drawn from our list of information improvements and the rest from our list of policy suggestions):

1. **Strengthen carbon emission reduction targets:** A large body of evidence has established that the impacts of co-pollutants on public health are substantial. Air-quality co-benefits therefore should be included in setting targets for carbon emissions reductions. The concept of the "social cost of carbon" should be expanded to include the social cost of co-pollutants. One result of incorporating this information into climate-policy design will be more ambitious carbon emissions reduction targets.

2. **Develop mechanisms for co-pollutant monitoring:** Climate-policy design should include provisions for monitoring policy impacts on emissions of co-pollutants, particularly at facilities and locations with high emissions. Annual reviews of monitoring results should be conducted with a view to introducing remedial measures if the climate policy is found to widen the extent of disproportionate impacts of co-pollutants on minorities and low-income communities. Findings of absolute increases in co-pollutant burdens associated with climate-policy implementation should trigger immediate policy actions to ensure co-pollutant abatement.

3. **Designate high-priority zones:** Climate-policy design should include identification of high-priority zones where the co-benefits from reduced carbon emissions have the potential to be particularly large. In these zones, the policy should ensure that emissions reductions will equal or exceed the average level of reductions achieved by the policy as a whole. Insofar as the climate policy relies on price-based instruments, this can be achieved by introducing specific caps for these zones that limit the number of permits to be auctioned or allocated to facilities in these zones and prevent the purchase of offsets or permits from elsewhere.

4. **Designate high-priority sectors and facilities:** Petroleum refineries and chemical manufacturers tend to have the biggest health impacts and the most disproportionate impacts on minorities and the poor. Also, there is a high degree of disproportionality in co-pollutant emissions—that is, a small number of facilities often account for a large share of emissions in a given sector. If this pattern is confirmed by more research, these sectors could be designated as high-priority; co-pollution reductions could be accelerated for them either by conventional regulatory instruments or by sector-specific emission caps that limit the number of permits allocated to these sectors and facilities and bar purchases of permits from other sectors and facilities.

5. **Allocate a share of carbon revenues to community benefit funds:** To ensure that disadvantaged communities that bear disproportionate air-pollution burdens obtain a fair share of the benefits from public investments in the clean energy transition, a fraction of the carbon rent generated by the use of price-based instruments in climate policy should be directed to community benefit funds to support environmental improvements and public health in these localities. Screening methods that incorporate social vulnerability to both pollution and climate change can be used to identify high-priority zones for such funds.

Why have these relatively straightforward modifications to market-based climate policies not been fully considered? Policy makers and advocates may have felt overburdened by other issues: the debates over conventional regulations versus price-based instruments, the social cost of carbon, and even the scientific basis for climate change itself. But part of the answer may also lie in the marginalization of the constituencies that are most burdened by co-pollutants.

In the end, this imbalance in policy priorities can be redressed only by ensuring that those advocating for environmental justice have a secure place at the climate-policy table. The benefit of this inclusion will be not only a more robust policy discussion, but also a wider base of support for the climate strategies that will be necessary to cool a warming planet. Working together, we can ensure that climate policy helps to secure a better environment, greater efficiencies in implementation, and more equitable outcomes for all Americans.

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 can generate "co-benefits" in the form of reduced emissions of other hazardous compounds generated in fossil-fuel combustion, such as nitrogen oxides (NO_X), sulfur dioxide (SO_2), particulate matter (PM), benzene, toluene, xylenes, and polycyclic aromatic hydrocarbons. Collectively these other compounds are termed "co-pollutants."

Co-benefits are relevant to climate-policy design for three reasons. First, by augmenting the benefits from reduced CO_2 emissions, co-benefits provide an efficiency rationale for more ambitious reduction targets. Second, if co-pollutant intensity—here defined as co-benefits per unit of carbon emissions—varies across sources of CO_2 emissions, there may also be an efficiency rationale for designing policies to achieve greater emissions reductions where co-benefits are higher. Third, insofar as co-pollutants tend to be concentrated in economically and socially disadvantaged communities, there is an equity rationale for incorporating co-benefits into policy design.

The equity aspect is frequently overlooked, because the geographic distribution of greenhouse gas (GHG) reductions is not consequential: because climate change is a global problem and we all share the same atmosphere, reducing GHG emissions in any location will benefit the planet equally. However, insofar as GHG reductions go hand in hand with co-pollutant reductions, location can matter greatly to communities near major GHG sources. Because co-pollutants have clear and immediate health impacts, recognition of the magnitude and distribution of co-benefits can broaden and deepen support for climate policy among diverse sectors of the public and legislators.

Although the overall magnitude of co-benefits has received considerable attention in the climate-policy literature, little research has examined spatial and intersectoral variations in co-pollutant intensity and the implications of these variations for efficiency and equity. This paper reviews the available evidence, identifies needs for further research, and points to viable policy options that can address these issues moving forward.

We specifically suggest that any carbon-pricing system—whether marketable permits or a carbon tax—should carefully monitor and track the co-pollutant impacts of decisions by polluters to either clean up or pay to pollute. Beyond that, we argue for more ambitious emissions reduction targets for industrial facilities known to have large co-pollutant health impacts, the implementation of zonal pollution restrictions to protect overburdened communities, and the creation of community benefits funds to improve environmental equity. We recognize that these provisions add to the complexity of carbon-charge systems, but we argue that the public-health benefits are too large to ignore and that any system that does not take co-benefits into account is likely to encounter resistance among important constituencies of the public.

The study is organized as follows. Chapter 2 discusses why co-benefits matter for policy design. Chapter 3 discusses conceptual and practical issues in the measurement of co-pollutant intensity that arise from the existence of different co-pollutants, different associated damages in different locations, and the limits of currently available data. Chapter 4 examines variations in co-pollutant intensity, drawing upon national data from the United States on industrial point-source emissions. Chapter 5 considers spatial variations in co-pollutants, geographic clustering, and environmental justice impacts.

Chapter 6 presents options for the incorporation of co-benefits into climate policy. Chapter 7 summarizes our findings and offers concluding remarks.

2. WHY DO CO-BENEFITS MATTER?

To illustrate the real-world relevance of co-benefits, consider two substantial emitters of CO_2 in California. The first is the La Paloma power plant, a natural gas–fired electricity-generation facility that is sited about 40 miles west of Bakersfield, with fewer than 600 residents living within a 6-mile radius and no other major industrial facilities in the immediate vicinity. The second is the ExxonMobil petroleum refinery in Torrance, with about 800,000 residents living within a 6-mile radius and a number of other significant pollution sources located nearby (making residents of the area subject to what public-health researchers call "cumulative exposure"). According to 2008 data from the California Air Resources Board, both facilities emit roughly the same amount of CO_2: 2.5 to 3 million tons per year (t/yr). But the Torrance refinery emits about 350 t/yr of particulate matter (PM), whereas the La Paloma plant emits about 50 t/yr.[1]

With regard to the simple mass of PM emissions, the co-pollutant intensity of the Torrance refinery is roughly seven times higher than that of the La Paloma power plant. If in measuring co-pollutant impacts, we consider the number of people living within a 6-mile radius, the co-pollutant intensity of the Torrance plant relative to that of the La Paloma plant increases further by a factor of roughly 1,300. Multiplying these together, and assuming that the strategies to reduce CO_2 emissions result in a proportional reduction in PM emissions, the co-benefits from a ton of CO_2 emissions reductions are nearly 10,000 times greater at the Torrance refinery than at the La Paloma power plant. If we also take into account differences in vulnerability arising from cumulative impacts of multiple pollution sources, the relative co-benefits of CO_2 emissions reductions at the Torrance refinery would increase still further.

2.1 Efficiency Implications of Co-Benefits

This comparison makes the point that co-benefits can matter greatly across GHG pollution sources. Why, then, have they not been a greater part of the climate-policy debate? Partly this is because CO_2 is a global "public bad," affecting the earth's climate equally regardless of where it is emitted, and so policy makers typically assume that it doesn't matter where emissions reductions are achieved: the marginal abatement benefit (MAB), or "social cost of carbon," will be the same. Because marginal abatement cost (MAC), the cost of an additional unit of emissions reductions, is known to vary across sources, economists often advocate policy instruments such as a carbon tax or carbon permits that put a uniform price (or "carbon charge") on emissions. Faced with this price, polluters will choose to

1 These examples are taken from Pastor et al. (2010a).

reduce their emissions up to the point where their MAC equals the charge. By equalizing MAC across different pollution sources, price-based policies are intended to yield any given level of emissions reductions at the lowest total cost. The efficient level of emissions reductions is achieved by a charge that equates MAC to MAB (see Figure 1).

In the presence of co-benefits, there is an efficiency case for increasing the carbon charge (by increasing the carbon tax or equivalently by reducing the number of carbon permits) to achieve greater emissions reductions than would be justified on the basis of the social cost of carbon alone. In other words, policy should aim to internalize both externalities—those from carbon emissions and those from co-pollutant emissions—rather than only one of them.

In theory, if co-pollutant intensity were uniform across pollution sources, the total marginal benefit of abatement likewise would be uniform, and it would be efficient to apply the same increased charge to all polluters (see Figure 2).[2] But in reality, as illustrated by the comparison between the two California facilities, co-pollutant intensity varies across sources and so the benefit of abatement varies too. The efficient level of CO_2 emissions reductions thus can vary across sources not only due to differences in MAC, but also due to differences in MAB once co-benefits are included.

This scenario of varying benefits is depicted in Figure 3 (where for simplicity we assume that both firms have the same MAC). As we will see, evidence on variations in co-pollutant intensity across industrial facilities in the United States suggests that this scenario is common. If so, this may call for more sophisticated policies to "price in" co-pollutants, particularly if the efficiency case for doing so is coupled with the criterion of fairness. First, however, we consider evidence on whether the co-pollutant problem is of a size that merits the serious attention of researchers and policy makers.

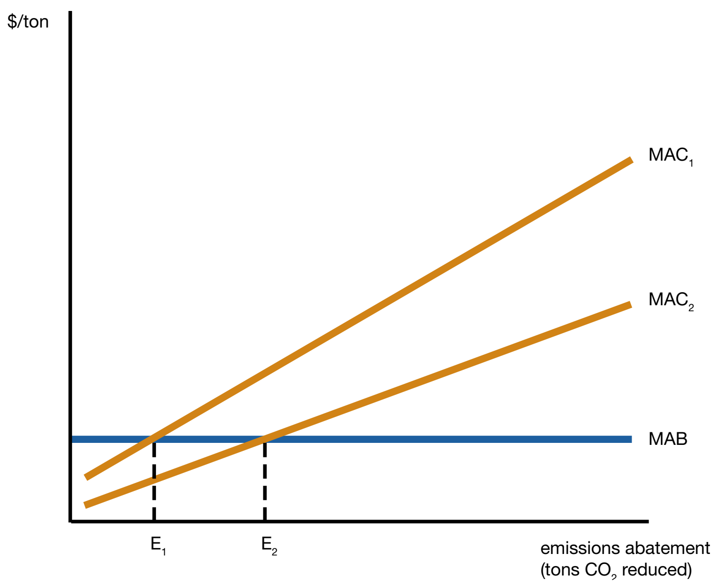


Figure 1: Efficient Abatement with Variable Costs

Faced with price-based incentives (such as emission taxes or purchased permits) for pollution abatement, firms choose to reduce emissions up to the point where their marginal abatement cost (MAC) equals the marginal abatement benefit (MAB). MAC varies across firms; MAB, to which the policy maker aims to peg the tax or permit price, conventionally is assumed to be uniform. When price =MAB, firm 2, which has a lower MAC, reduces emissions more than firm 1 (E2 > E1). The price-based incentive policy is designed to yield efficient (i.e., lowest total cost) emissions reductions.

[2] As Nemet et al. (2010) note, an alternative way to depict this result is to deduct co-benefits from the marginal abatement cost. The redefinition of "abatement cost" as net of co-benefits may be useful if the policy issue is framed as minimizing the cost of attaining a previously chosen abatement level rather than choosing the optimum level of abatement.

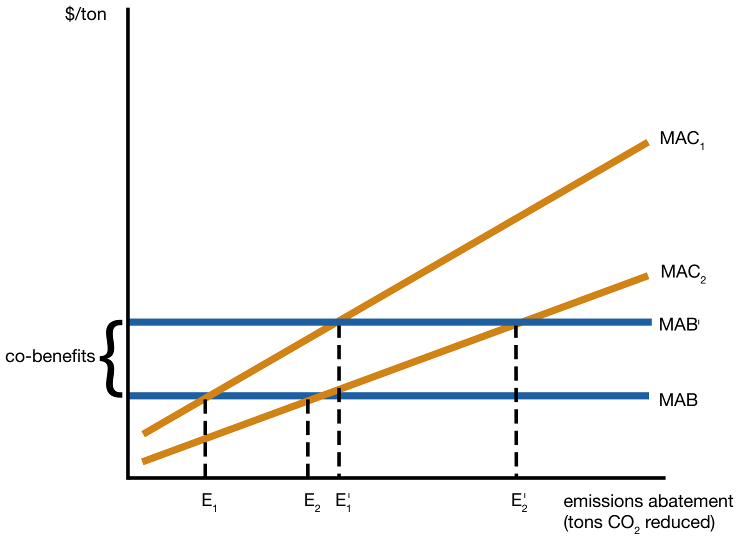


Figure 2: Efficient Abatement with Co-Benefits and Variable Costs

Abatement of CO_2 emissions generates co-benefits via abatement of co-pollutant emissions. Total marginal abatement benefit (MAB′) is greater than the marginal benefit from CO_2 abatement alone (MAB). Efficient levels of abatement for each firm rise accordingly.

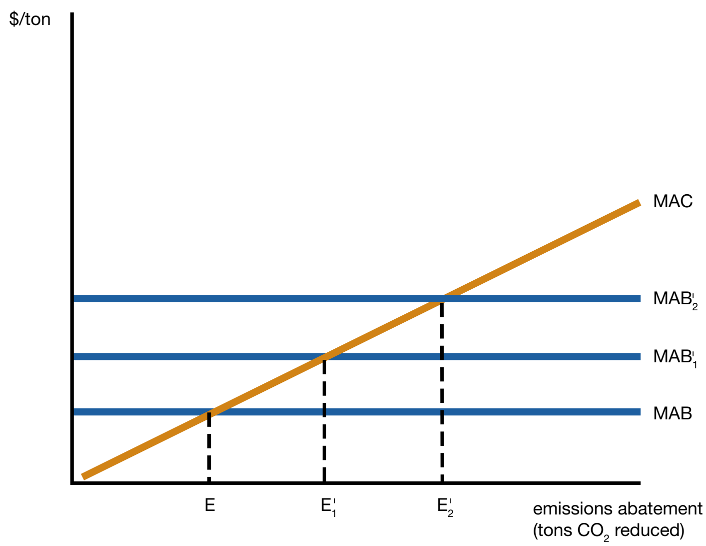


Figure 3: Efficient Abatement with Variable Co-Benefits and Uniform Costs

If co-benefits from co-pollutant abatement vary across firms, so do the efficient levels of abatement vary, even if firms face the same marginal abatement costs (as depicted here for simplicity). When co-benefits are higher for firm 2 than for firm 1, the efficient level of abatement is higher, too ($E_2 > E_1$).

2.2 Magnitude of 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 large. In a review of more than a dozen studies of multiple locations, including industrialized and developing countries, Bell et al. (2008) conclude that there is strong evidence that the health co-benefits of policies to reduce 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), with a range of \$2–\$196/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, with a range of \$5–\$65/tCO_2.[3]

In a study of the co-benefits of carbon emissions reductions in the European Union conducted for the Netherlands Environmental Assessment Agency, Berk et al. (2006) find 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). A subsequent study by Bollen et al. (2009) modeled the health benefits from reduced emissions of particulate matter and concluded that "the discounted benefits of local air-pollution reduction significantly outweigh those of global climate-change mitigation, at least by a factor of two."

Groosman et al. (2011) analyzed health co-benefits of reduced emissions in the transport and electric power sectors that would result in the United States from adoption of a climate policy akin to the Warner-Lieberman bill that was proposed in the U.S. Senate in 2008. They conclude that "climate-change policy in the U.S. will generate significant returns to society in excess of the benefits due to climate stabilization." The majority of co-benefits in their analysis come from reduced emissions of SO_2 from coal-fired power plants. They estimate that the present value of co-benefits from the climate policy over a 25-year time horizon would total \$90–725 billion. The marginal co-benefit range is \$2–14/tCO_2 (in 2006 dollars). This is the same order of magnitude as the policy's \$9/tCO_2 marginal abatement cost estimated by the U.S. Environmental Protection Agency (USEPA), which can be taken as an indicator of the bill's implicit measure of the social cost of carbon.

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. This monetary estimate includes only the mortality impacts of criteria air pollutants and does not include costs of morbidity (such as chronic respiratory ailments), health impacts from other co-pollutants, or health impacts from climate change. Muller et al. (2011) estimate that air pollution damages from coal-fired electricity generation in the United States exceed the industry's value added—not including damages from carbon dioxide emissions.

Holland (2010) notes that in addition to "output effects" from reduced use of fossil fuels, spillovers from climate policy may include "substitution effects" that could either increase or decrease emissions of co-pollutants. For example, he states that an increase in combustion temperatures at natural gas–fired electricity-generation units "reduces CO_2 emissions, but increases NO_x emissions." Testing for climate-policy spillovers using data from electricity generation in California, however, he finds that substitution effects are negligible compared to co-benefits arising from output effects.

3 For discussion and a critique of this estimate, see Ackerman and Stanton (2010).

Of course, the magnitude of co-benefits depends in part on the regulatory regime for co-pollutants. If co-pollutant emissions are themselves constrained by a cap, as in the case of SO_2 emissions from U.S. power plants, and the cap is binding tightly so that climate policy will not generate additional reductions, then there may be no immediate health benefit to be gained from explicitly incorporating these specific co-pollutants. However, even in the presence of a binding cap on emissions of a co-pollutant, climate policy can generate additional economic benefits via avoided future investments in co-pollutant abatement (Burtraw et al. 2003). Moreover, in the case of most co-pollutants, regulation relies on emissions standards and mandated technologies, not caps, in which case climate policy will generate further emissions reductions and the attendant health benefits.

Air-quality co-benefits are more immediate than the longer-term benefits of climate-change mitigation, so their value is less sensitive to the choice of a discount rate (a factor by which future benefits are weighted relative to current benefits). At higher discount rates—that is, where actors value the present more highly—the value of co-benefits relative to the benefits of climate-change mitigation increases (Bollen et al. 2009). Moreover, because the air-quality benefits arrive more quickly than reductions in global warming, they tend to be "seen" by constituencies that can then become a political force for reducing GHG emissions. The political salience of co-benefits was recently demonstrated in the battle over Proposition 23 in California, an unsuccessful effort backed by oil companies to roll back the state's tough new standards on GHG emissions (Lerza 2012). Although environmental advocates may wish that consciousness of the dangers of climate change and concern for the future of the planet were sufficient to motivate policy, the California experience suggests that there are compelling political (as well as economic) reasons for incorporating co-pollutants into climate policy.

In an analysis of co-pollutant emissions from electricity-generation units (EGUs) and light-duty vehicles in the United States, Muller (2012) analyzes not only aggregate co-pollutant damages, but also variations across sources and locations. He estimates the average damage from co-pollutant emissions at EGUs is \$62 per ton CO_2, and concludes that "the effective damage from GHG emissions from these source types would increase dramatically if co-pollutant impacts are counted"; a policy reflecting these impacts hence would likely be "significantly more stringent" than one focused on GHG impacts in isolation. Disaggregating across sources, Muller finds substantial variations in co-pollutant damages. Per ton of CO_2 emissions, the average co-pollutant damage from bituminous coal-burning EGUs, for example, is 34 times higher than from natural gas–fired units. Moreover, a small number of facilities (what statisticians would call "outliers") 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 percent of the pollution sources.

In sum, the co-pollutant benefits of climate policy are potentially quite large. Designing climate policy that seeks to capture those gains is desirable from the standpoints of both economic efficiency and public support for the policy. The case for integrating co-benefits into climate policy is further reinforced when we consider distributional impacts and the value of fairness.

2.3 How Important Are Co-pollutants From Industrial Sources?

In a recent analysis of the role of industrial sources in cumulative cancer risks from air toxics, Adelman (2012) suggests that, in most U.S. locations, mobile sources and small sources (such as gas stations and dry cleaners) account for a considerably larger share of cancer risks. Focusing on census tracts with the highest cumulative cancer risks (about 3,100 tracts nationwide), he finds that "for about three-quarters of the tracts, industrial emissions accounted for less than 3 percent of the cumulative cancer risk" (Adelman 2012, p. 33). Although he notes that industrial sources may predominate in some locations with high pollution levels, he suggests that the overall pattern is one where industrial co-pollutant issues might not loom large in climate-policy design, and that because of the concentration of important sources in a few locations, implementing limited remedial policies such as enhanced monitoring and zonal restrictions on permit trading in these cases would not be difficult, a point to which we return in Chapter 6.

To investigate further the relative importance of industrial sources in overall air-pollution burdens, we first examined data from the 2005 National-Scale Air Toxics Assessment (NATA), the main data source used by Adelman (2012). NATA models air toxics from multiple sources to derive census-tract-level estimates of cancer risks and potential noncancer health effects, including respiratory risk and neurological risk.[4] Using these data, we calculated the contribution of each source type to estimated cancer, respiratory, and neurological risk for (a) the average census tract, (b) the census tract of the average person (i.e., the population-weighted average), (c) the census tracts containing the top 10 percent of the population with regard to risk, and (d) the census tracts containing the top 1 percent of the population with regard to risk.[5] The results of the analysis are presented in Table 3. Consistent with Adelman's analysis, the table shows that for air toxics, industrial point sources are not the major contributors to cancer risk.[6]

Focusing solely on NATA's assessment of the cancer risk from air toxics, however, may be misleading if our aim is to assess the broader question of the importance of co-pollutants from industrial sources. As shown in Table 3, industrial sources are major contributors to neurological risk, particularly in those census tracts with the highest neurological risk overall. More important, NATA was designed specifically to model air toxics for which the USEPA has no active monitoring—and thus it leaves out, by design,

[4] Sources include point, nonpoint, on-road mobile, nonroad mobile, background, and secondary formation and decay (referred to as "secondary" in Table 2.1 below). For more information, see: http://www.epa.gov/ttn/atw/nata2005/05pdf/nata_tmd.pdf. In our past use of NATA (e.g. Pastor, Morello-Frosch, and Sadd, 2005), we have used alternate risk estimates generated by applying chemical-specific potency estimates from the California Office of Environmental Health Hazard Assessment (OEHHA) when they differed from those used by USEPA (or were not included in the USEPA's NATA risk estimation procedure altogether). The biggest difference between the chemical potency estimates from the OEHHA and those employed by USEPA is that OEHHA includes diesel particulates in their estimation of cancer risk and USEPA (and hence Adelman) does not. For this exercise, however, we used the "off-the-shelf" NATA risk estimates so that our results can be compared more directly with those of Adelman (2012).

[5] To be consistent with the geography of the NATA data–2000 census tracts – population information was taken from the 2000 5-year American Community Survey (ACS) summary file.

[6] It is also possible to calculate respiratory risk from the NATA, but the pattern there is similar to cancer risk, and Adelman (2012) focused only on cancer risk; we also look at neurological risk, because the pattern of the contributing role of point sources is quite different there.

the health effects of criteria air pollutants for which there is monitoring. Stationary sources are responsible for a large share of harmful criteria pollutants. Nationwide, nearly 95 percent of SO_2 emissions and over 40 percent of NO_x emissions come from stationary sources (see Table 6).

We can also calculate the relative industrial contribution of $PM_{2.5}$, a pollutant associated with premature death, decreased lung function, and asthma, using data from the USEPA's National Emissions Inventory for 2008. We do so in Table 4. The main contributors to $PM_{2.5}$ are dust, agriculture, and fires. However, none of these sources would come into a cap-and-trade or other carbon-pricing policy (although guarding against climate change will reduce fire probability). What would come under any pricing strategy are industrial point sources, including power plants, as well as mobile sources and residential fuel combustion. Industrial sources altogether account for more than 14 percent of the nation's $PM_{2.5}$, compared to about 9 percent from mobile sources and 6 percent from residential fuel.

Table 3: Distribution of Risk from the 2005 National-Scale Air Toxics Assessment by Source and Risk Type

	Source Type						
	Point	Area	Onroad	Nonroad	Background	Secondary	Total
Cancer Risk							
For the average U.S. census tract							
Simple average	3%	12%	12%	5%	24%	45%	100%
Population-Weighted Average	2%	11%	12%	5%	24%	46%	100%
For highest-risk U.S. census tracts							
Tracts containing 10% of pop. at highest risk	4%	19%	25%	10%	14%	29%	100%
Tracts containing 1% of pop. at highest risk	3%	22%	28%	13%	9%	24%	100%
Respiratory Risk							
For the average U.S. census tract							
Simple average	2%	21%	19%	10%	1%	47%	100%
Population-Weighted Average	2%	22%	19%	10%	1%	47%	100%
For highest-risk U.S. census tracts							
Tracts containing 10% of pop. at highest risk	1%	29%	25%	12%	0%	32%	100%
Tracts containing 1% of pop. at highest risk	0%	46%	15%	14%	0%	25%	100%
Neurological Risk							
For the average U.S. census tract							
Simple average	11%	26%	10%	7%	47%	0%	100%
Population-Weighted Average	10%	26%	10%	7%	47%	0%	100%
For highest-risk U.S. census tracts							
Tracts containing 10% of pop. at highest risk	23%	40%	14%	7%	16%	0%	100%
Tracts containing 1% of pop. at highest risk	42%	44%	4%	4%	6%	0%	100%

Table 4: Tons of $PM_{2.5}$ by Sector in 2008

	Tons	%
Dust	1,312,722	21.6%
Agriculture	930,989	15.3%
Fires	1,766,309	29.1%
Fuel Combustion		
Comm/Indus/Elec Generation	448,973	7.4%
Residential	355,546	5.9%
Industrial Processes	412,719	6.8%
Mobile	540,350	8.9%
Solvent	3,796	0.1%
Miscellaneous	294,681	4.9%
Total	6,066,086	100%
Source: National Emmissions Inventory (2008), http://www.epa.gov/ttn/chief/net/2008inventory.html.		

Finally, although Adelman's study addresses an interesting question—how important are industrial co-pollutants to reducing cancer risk from air toxics?—for our purposes this is not the relevant policy issue. Instead, the issue is this: if we are to have a cap-and-trade or other carbon-pricing system, are there important health co-benefits potentially left unattained? Given the significance of industrial sources in the neurological risks from air toxics and in emissions of the criteria air pollutants that would be impacted by carbon pricing, we think the answer to this question is a resounding yes.

In this study we focus on industrial sources not only because these account for an important share of air-pollution burdens in some highly polluted locations (including places with concentrations of GHG emitters, as we will see in Chapter 5), but also because industrial facilities have been of major concern in the environmental justice literature, and because relatively rich databases are available to document variations in co-pollutant intensity among industrial sources. Our aim here is not to produce a comprehensive mapping of co-pollutant burdens from all sources, but to illustrate the possibilities and data requirements for integrating co-pollutants into climate policy. The collection and analysis of comparable data on emissions from mobile sources and smaller point sources will also be vital for policy design.

2.4 Distributional Incidence of Co-Benefits

In the past three decades, a substantial body of literature has examined the relationship between pollution burdens and socioeconomic status in the United States. The bulk of these studies have found that people of color and low-income communities tend to bear disproportionate burdens, even when controlling for the other factors that may help to explain the pattern of facility location or levels of ambient air pollution, such as industrial land use or population density.[7] Although there remain many

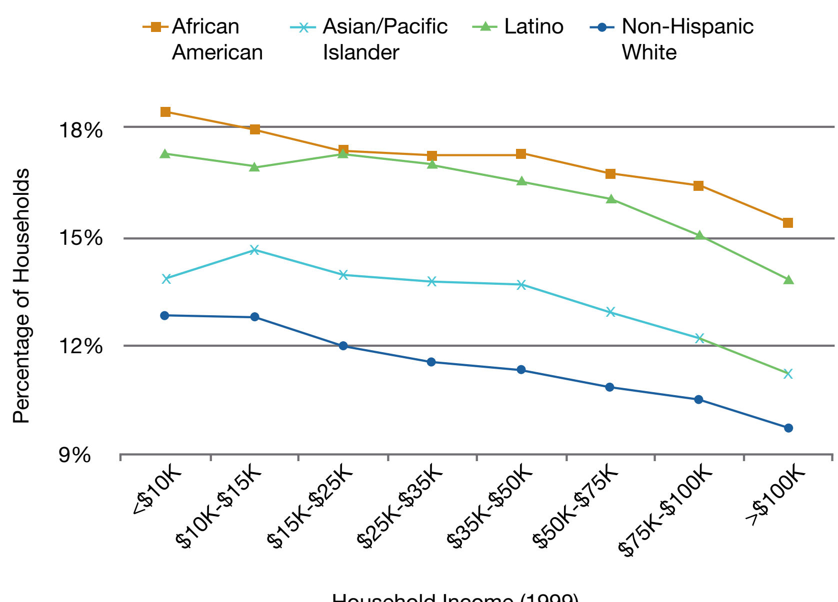
7 For literature reviews, see Szasz and Meuser (1997), Pastor (2003), Boyce (2007), Mohai (2008), and Morello-Frosch et al. (2011). In a metastudy of various research efforts, Ringquist (2005) concludes that the disparities are more consistent and statistically significant with regard to race than to income.

outstanding methodological issues in this work—including the degree of acceptable risk, the measurement of socioeconomic status, the longitudinal nature of siting versus move-in, and the role of spatial clustering in multivariate analysis—the conclusion is clear: disparities do exist.

A mandate to do something about them also exists. Even as research on environmental disparities was in its early phase, President Bill Clinton issued an Executive Order in 1994 mandating each federal agency to take steps to identify and rectify "disproportionately high and adverse human health or environmental effects of its programs, policies, and activities on minority populations and low-income populations." Many state agencies also operate under state environmental justice mandates, as in the case of the California Environmental Protection Agency (Cal/EPA); others have been motivated to consider environmental justice dimensions of their policies by Title VI of the Civil Rights Act, which prohibits state agencies that receive federal funds from discriminating on the basis of race.

Does this pattern of racial and income disparities exist in the case of co-pollutant exposures? Using California Air Resources Board data on emissions of PM_{10} (particulate matter 10 micrometers or less in diameter) and NO_X from 146 facilities in three important CO_2-emitting industrial sectors—power plants, petroleum refineries, and cement plants—Pastor et al. (2010a, 2010c) analyze the demographic correlates of co-pollution exposure. Their findings suggest that people of color are more likely than non-Hispanic whites to reside in close proximity to these facilities—even controlling for household income (see Figure 4). Analyzing emissions data, people of color also are more likely to reside near facilities posing greater co-pollutant burdens. Using a simple index of facility-level co-pollutant burden (tons of PM_{10} emissions multiplied by the number of people living within a given distance of a facility), they find that a relatively small number of facilities are responsible for a lion's share of the racial and ethnic disparities, with refineries topping the list.

Figure 4: Percentage of Households Living Within 2.5 Miles of any Facility by Income and Race/Ethnicity in California



Source: Pastor et al. (2010b, p. 6).

As we show below, the national pattern is broadly consistent with the California finding that refineries pose the most disproportionate co-pollutant burden by race/ethnicity. We also find that a relatively small number of facilities contribute a large share of the overall risk. This suggests that co-pollutant abatement (or lack thereof) among a small number of facilities and/or sectors could largely determine any progress (or regress) in the realm of environmental justice associated with climate-policy implementation. It also implies that big gains on the equity side could be ensured with relatively modest sectoral and/or facility-level adjustments to any carbon-pricing program.

Even without targeted interventions, the uneven incidence of co-pollutant burdens implies that the co-benefits of climate policy could advance the cause of environmental justice. If emissions reductions were constant (as a percentage of current emissions) across facilities, communities that bear larger burdens would gain larger co-benefits. The explicit integration of co-benefits into policy design could further advance this goal as well as build popular support for GHG reductions overall. In contrast, the neglect of co-benefits in policy design not only would fail to take full advantage of these opportunities, but also could exacerbate disparities if the pattern of emissions reductions turns out to be biased against overburdened communities.

The importance of environmental equity is heightened by the fact that low-income communities and people of color also face a "climate gap" in that they bear a disproportionate share of the likely burdens of climate change (Shonkoff et al. 2011). One simple measure: communities of color in California are far more likely to live in or near "heat islands," places where the effects of warming are exacerbated by lack of trees and shade. At the same time they are less able to afford air-conditioning. With the planet likely to warm even as we begin to tackle our GHG emissions, the weight of this disproportionate burden will grow.

2.5 Could Climate Policy Lead to Increases in Co-Pollutant Burdens?

The magnitude of co-pollutant damages and their uneven distribution imply that integrating co-benefits into climate policy could have payoffs in the areas of both efficiency and equity. A policy that ignores co-benefits would be inefficient in two respects: it would choose suboptimal emissions reduction targets overall, and it would in effect leave health-care dollars lying on the floor by failing to take account of differences in total abatement benefits across emission sources. On the equity side, it could also widen the preexisting disparities in co-pollutant exposures across communities, including disparities correlated with race and income, if the resulting emissions reductions are smaller in more polluted locations.

The distributional risk would be even more troubling if climate policy leads to increased co-pollutant burdens in certain places, rather than simply to uneven reductions across our landscapes. There are several ways in which some locations could see absolute as well as relative increases in emissions as a result of climate policies that do not take the co-pollutants into account:

1. **Intra-facility technological change:** Changes in how fossil fuels are burned may reduce CO_2 emissions, but increase emissions of co-pollutants. For example, as noted above, increasing combustion temperatures at natural gas–fired electricity-generation units can reduce CO_2 emissions, but increase NO_X emissions. And if carbon capture and sequestration (CCS) technologies were to become feasible, their additional energy requirements could generate increased co-pollutant emissions even as CO_2 emissions decline.

2. **Interfacility shifts:** One potential result of climate policy is a shift from coal to natural gas for electricity generation over the medium term, since the latter produces roughly twice as much electricity per ton of CO_2 emissions (IEA 2011, p. 39).[8] This could happen sooner rather than later given a recent rule proposal by the Obama administration that would set a limit on GHG emissions per megawatt-hour from new power plants (Barringer 2012). Assuming that the shift from coal to natural gas also entails a shift in the site of power generation, the result will be an increase in co-pollutant burdens in the specific places where gas-fired generation facilities are located.

3. **Transportation fuel shifts:** Similar displacement occurs in transportation fuels. In the United Kingdom, the introduction of vehicle taxes based on CO_2 emission rates has spurred greater use of diesel-powered vehicles that emit less CO_2 per mile (about 70 percent as much as gasoline-powered vehicles), but more particulate matter.[9] Mazzi and Dowlatabadi (2007) estimate that the shift to diesel-fueled cars in the United Kingdom will cause 20–300 additional deaths annually in the two decades from 2001 to 2020.[10]

[8] For this reason, natural gas is often touted as a "bridge fuel" in climate policy. However, methane leakages in natural-gas production and consumption could more than negate its advantages with regard to CO_2 emissions. For discussion, see Wigley (2011).

[9] Walsh (2008) notes that the higher co-pollutant intensity of diesel-powered vehicles in Europe is attributable to weaker diesel car emission standards than those in the United States and Japan, and that low-sulfur fuels coupled with high-efficiency exhaust-gas filters can reduce PM emissions to levels similar to or even below those of gasoline-powered vehicles.

[10] Two additional ways that climate policy could trigger increased pollution in some locations deserve mention, although these fall outside the scope of the present study. First, the policy could lead to the relocation of production to sites outside the regulated jurisdiction (i.e., to other states or countries with no regulation or weaker regulation), a phenomenon sometimes called "carbon leakage." Co-pollutant emissions would be relocated along with carbon emissions, and both would rise in the less regulated area. One policy remedy to leakage is "carbon cost leveling," border price adjustments that reduce or eliminate incentives for leakage (Economic and Allocation Advisory Committee 2010; Grubb 2011). Second, production activities associated with investments in energy efficiency (such as insulation) and renewable energy (such as windmills and photovoltaics) generate pollution too, as does the production of equipment for the extraction and combustion of fossil fuels.

These possibilities imply that some "hot spots" could see increased emissions. Fowlie et al. (2011) show that some absolute increases did occur in the NO_X trading program in southern California. In the case of climate policy, absolute increases may be less likely than uneven reductions in co-pollutants across locations. Some might argue that as long as the policy does not lead to absolute increases in co-pollutant emissions anywhere, it results in the pollution equivalent of a Pareto improvement: some may benefit more than others, and relative inequality in pollution burdens may widen, but in absolute terms no one would be worse off. We believe that pure equity considerations are important, however, and that consideration of equity issues can build support among constituencies that matter for policy change. We also believe, as mentioned, that it would be inefficient to leave potential health gains to one side if they can be achieved with modest shifts in carbon-pricing policies.

2.6 Interactions Between Climate Policy and Existing Regulations

Some proponents of a cap-and-trade approach to GHG reduction argue that co-pollutants are already regulated, and that bringing them into a climate regime would unnecessarily complicate the construction of a new market. We acknowledge that the existing regulatory regime may affect the extent of co-benefits, but that does not mean that co-benefits from climate policy are inconsequential or irrelevant to climate-policy design. Three regulatory regime settings can be distinguished:

1. **Unregulated co-pollutants:** Many air toxics are 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.[11] In cases where regulatory standards do exist, moreover, 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 example, coal-fired power plants that together account for 37 percent of U.S. capacity still had no emissions control equipment in place as of 2010 (Credit Suisse 2010, p. 20). These exclusions will not apply to USEPA regulation of GHG emissions from power plants now being formulated under the Clean Air Act (CAA), and as a result they will "reach some existing sources that have thus far escaped direct federal regulation" (Kaswan 2012, p. 55).[12]

2. **Regulated co-pollutants:** Even the strongest of our current environmental regulations—federal regulations applied to "new sources" in nonattainment areas, and state regulations in California—do not completely eliminate co-pollutant emissions from fossil-fuel combustion.[13] Even if existing regulations were economically efficient in the sense of equating the marginal cost of abatement to a well-defined monetary measure of marginal benefits, the carbon emissions re-

[11] See USEPA, Chemical Emergency Preparedness and Prevention Office, "List of Lists: Consolidated List of Chemicals Subject to the Emergency Planning and Community Right-to-Know Act (EPCRA) and Section 112® of the Clean Air Act" (http://www.epa.gov/ceppo/pubs/title3.pdf).

[12] In addition, the USEPA's recent mercury rule and cross-state pollution rule are likely to curtail these exclusions.

[13] The elimination of lead from vehicle emissions was a notable exception, made possible by the fact that lead was a gasoline additive.

ductions achieved via energy efficiency or conversion to alternative energy sources would bring about additional co-pollutant reductions.[14] Given that the stringency of regulation is in part a response to the severity of the pollution problem the regulations address, the co-benefits from reduced emissions of regulated pollutants may be just as significant as those from reduced emissions of currently unregulated pollutants.

3. **Co-pollutants subject to a cap:** There is one special case in which carbon emissions reductions could fail to reduce emissions of co-pollutants. If co-pollutant emissions themselves are already regulated by a cap-and-permit system and two additional conditions are met, there could be no further reduction in co-pollutants from carbon emissions reductions. The two additional conditions are that, first, the existing cap must act as a binding constraint on current emissions of the co-pollutant (which is not invariably true) and, second, there must be technological possibilities for reducing carbon emissions without simultaneously reducing co-pollutant emissions (thereby raising co-pollutant intensity per ton of carbon). In the case of SO_2 emissions from power plants, for example, reduced burning of coal in response to climate policy would ease pressure on the existing SO_2 cap, and firms could respond by shifting to coal with a higher sulfur content.

The fact that climate policy will generate co-benefits via reduced emissions of co-pollutants, notwithstanding current regulations on the latter, is underscored by the conclusion of the National Academy of Sciences (2009), cited above, that co-pollutants currently result in roughly 20,000 premature deaths annually in the United States. These fatalities occur under our actually existing regulatory regime.[15] With reduced burning of fossil fuels brought about by climate policy, some of these deaths would be

3. MEASURING CO-POLLUTANT INTENSITY

The review of evidence in the previous chapter suggests that the air-quality co-benefits from climate policy will be quantitatively significant. It also points to the importance of advancing our understanding of variations across sources and locations in co-pollutant intensity—that is, co-pollutant damages per ton of CO_2 emissions—for the design of efficient and equitable climate policies. This requires us to grapple with the conceptual and practical challenges of measuring co-pollutant intensity. We now turn to these issues.

[14] Putting a monetary value on these benefits, including reduced morbidity and mortality, is controversial. For discussions of how this is done, see Dorman (1996) and Viscusi and Aldy (2003). For contrasting views on the use of cost-benefit analysis for this purpose, see Ackerman and Heinzerling (2005) and Revesz and Livermore (2008).

[15] In part, this reflects the fact that air-quality standards are not always met. In southern California and the San Joaquin Valley, for example, it is estimated that failure to meet federal clean-air standards leads to nearly 4,000 excess deaths and more than $28 billion in costs annually (Hall et al. 2008).

3.1 Narrow Versus Broad Definitions

Definitions of co-pollutant intensity can differ in the breadth of processes linked to fossil-fuel combustion. At the narrow end of the spectrum, co-pollutants refer solely to emissions generated by the burning of the fossil fuels themselves. A slightly broader definition also includes emissions generated during combustion by virtue of fuel additives, such as lead or the lead substitute methylcyclopentadienyl manganese tricarbonyl (MMT) in gasoline.[16]

A still broader definition includes other emissions attributable to fossil-fuel production or fossil-fueled economic activities. In the case of natural gas, for example, it could include the air emissions generated by wells developed by means of hydraulic fracturing ("fracking"). In the case of automobiles, it could include pollution resulting from the degradation of tires and brake linings. At the broadest end of the spectrum, in an economy that is heavily reliant on fossil fuels it could include a large fraction of total pollution nationwide.

The magnitude of the difference between the narrow and broad measures depends on the specific co-pollutant and its sources. Data from the USEPA (2011b) for emissions of NO_X, $PM_{2.5}$, and SO_2 from industrial facilities nationwide indicate that the share of fossil-fuel combustion in total emissions ranges from 40.4 percent in the case of $PM_{2.5}$ to 87.6 percent in the case of SO_2 (see Table 5). A matrix showing how these ratios vary by sector as well as by co-pollutant would be a useful input into policy analysis. In the electrical power sector, for example, the vast majority of co-pollutants are generated by fuel combustion itself, and so the difference between co-pollutant intensities measured by the narrow and broad definitions is minor. But the same is not true for cement manufacturing, where substantial PM emissions result from physical attrition of raw materials such as limestone, clay, and sand. We do not know of any data source that systematically disaggregates sector-level or facility-level emissions of co-pollutants into emissions from fossil-fuel combustion and nonfuel emissions.

Table 5: Shares of Industrial Stationary-Source Emissions from Fuel Combustion and Other Processes, National Level, 2011

	NO_X	$PM_{2.5}$	SO_2
Fuel combustion	75.7%	40.4%	87.6%
Other processes	24.3%	59.6%	12.4%

Source: Calculater from USEPA (2011b).

[16] More than 95 percent of gasoline sold worldwide is now lead-free, but Walsh (2008) reports that leaded gasoline continues to be sold in a number of North African, central European, and Asian countries.

For some policy purposes the narrow definition, perhaps expanded to include co-pollutants resulting from fuel additives, may be most useful. If climate policy were simply to reduce fossil-fuel combustion, without at the same time reducing the scale of associated activities such as cement manufacturing or car driving and therefore without reducing nonfuel emissions that result from these activities, the narrow definition would be the appropriate measure. But one way that climate policy is likely to curb carbon emissions is precisely by reducing the extent of carbon-intensive activities, like driving cars. Insofar as reductions of this type occur, it is not a great conceptual stretch to define the resulting decrease in nonfuel emissions as a co-benefit of climate policy too and to include these in the definition of co-pollutant intensity.

In principle, therefore, choosing the appropriately narrow or broad definition of co-pollutant intensity hinges on degree to which reduced carbon emissions are achieved via energy efficiency and/or conversion to nonfossil energy sources (so that the same economic activities continue with their resulting emissions) or via reductions in the activities that are fossil-fueled. Both are plausible, and the mix between them is likely to vary from case to case. If so, the narrow definition can be considered to be a lower-bound measure of the co-benefits from reduced emissions of co-pollutants, while the broad definition provides an upper bound.

In practice, data availability may drive the choice of measures. As noted, the available data on co-pollutant emissions from U.S. industrial facilities, the focus of this study, generally refer to total emissions rather than emissions from fossil-fuel combustion alone. Hence we report broad measures of co-pollutant intensity.

3.2 Choice of Numerator

Choosing the numerator for co-pollutant intensity measurement requires decisions on which co-pollutant(s) to analyze, the method of aggregation across pollutants when more than one are combined in a single measure, and the units in which co-pollutant impacts are expressed.

The choice of which co-pollutants to measure is driven by policy relevance and data availability. The simplest measures refer to a single co-pollutant, like particulate matter (PM) or nitrogen oxides (NO_X), enumerated in terms of sheer mass (e.g., tons). We report data for several such measures in the following chapters.

The simplest method to aggregate co-pollutants is to add up their mass, as is sometimes done for emissions of hundreds of different chemicals from industrial facilities that are reported annually in the USEPA's Toxics Release Inventory (TRI). A shortcoming of this method is that, pound for pound, some chemicals are far more hazardous than others. To address this problem, one can assign weights to different chemicals based on their relative toxicity. Another extension—and one with important implications for public health—is to take into account the number of people impacted and their vulnerability to adverse health effects.

A fundamental question is whether to incorporate population density into impact measurement. As illustrated by the comparison between the La Paloma power plant and the Torrance refinery in California, with which we opened Chapter 2, the number of people impacted by emissions can vary greatly. If we want to assess statistical risk to individuals, regardless of whether they happen to reside in a densely or sparsely populated location, then population density is not relevant. However, if we want to assess total human health effects—such as the predicted number of premature deaths—population density is clearly relevant.

There is no "right" or "wrong" answer to the question of which measure is more appropriate, as it poses an ethical dilemma resulting from the tension between two distinct normative criteria. On the one hand, few would argue that health risks to individuals are more acceptable if the individuals happen to live in rural areas rather than in cities. On the other hand, most would agree that the release of a given amount of pollution in Manhattan is more consequential than the same release in the middle of a desert, a logic that presumably influenced the U.S. government's decision to conduct atomic weapons tests in the deserts of the Southwest rather than in the metropolitan areas of the Northeast.

The ethical dilemma is illustrated by the following comparison: using the criterion of total human health impacts, a one-in-a-million increase in cancer risk in Manhattan (with a population density of 67,000/mi^2) would count more than a one-in-a-hundred increase in Wyoming (with a population density of 6/mi^2)—whereas using the criterion of risk to individuals, the latter is worse by four orders of magnitude. Rather than relying entirely upon one measure or the other as the appropriate basis for policy making, a more nuanced response to this dilemma would be to incorporate both in a multicriteria decision analysis, acknowledging potential tradeoffs between them.

In the case of point-source emissions from industrial facilities, one simple way to incorporate population density into impact measures is to count up census data on the number people living within a given radius of the facility. Since different radii may generate different rankings, this requires an answer to the question, "How near is 'near'?" Moreover, individuals living within the same radius may be very differently affected depending on wind patterns and other factors that determine where emissions are inhaled.

Fate-and-transport models address both problems by estimating population exposures taking into account such factors as stack heights, prevailing winds, and chemical decay rates. These models require considerably more information and computational sophistication. As a result, researchers often simply draw buffers of different lengths and consider the population living within the relevant radii (see, e.g., Pastor et al. 2010a, 2010c). This has been the standard approach in much of the environmental justice literature (Mohai and Saha 2006).

The USEPA's Risk-Screening Environmental Indicators model, discussed in more detail in the next section, incorporates fate-and-transport modeling for point-source releases of air toxics, using 1-km^2 grid cells as the spatial unit for measurement of exposure and impacted populations. This level of resolution facilitates quite fine-grained analysis of environmental justice as well as aggregate pollution impacts, as we illustrate below. Similarly, Muller and Mendelsohn (2009) use the Air Pollution Emission Experiments and Policy model, in which counties are the receptor spatial unit, to generate source-specific measures of damages from criteria air pollutants. They find that population density and (in urban areas) stack height have strong effects on damage estimates.

A final numerator issue arises from the fact that not everyone is equally vulnerable to pollution. Children, for instance, generally are more vulnerable than adults, and in some cases there are differences between men and women. Apart from biological factors, moreover, there are social and environmental ones. People who are unable to afford pollution-avoidance strategies, like air conditioners, are more vulnerable. People who have less access to medical care are more vulnerable. And people who live in communities with greater cumulative impacts from multiple pollution sources and already suffer from ill health as a result may be more vulnerable to any additional pollution load. These considerations can be added in a social vulnerability layer of the analysis. In this study we do not attempt to include them, confining our analysis to the other issues in choosing a co-pollution numerator.

3.3 Measuring Intensity

Co-pollutant intensity can be measured at multiple scales, ranging from the micro level (e.g., an individual facility) to the macro level (aggregating all sources in a given spatial unit, such as a state or a nation). In policy analysis, meso levels that aggregate subsets of sources (e.g., all facilities within a given industrial sector) also can be of interest.

Data availability limits the measures of co-pollutant intensity that can be constructed at any given level. One recommendation that emerges from this study is that data availability ought to be expanded to aid researchers, concerned communities, and policy makers, among other ways through the standardization of facility codes, so that different sorts of emissions can be linked in order to analyze the overall impact of a facility. Thus far, the collection of data on CO_2 emissions has been largely independent of the 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.

Here we describe the most important data sources on emissions of CO_2, criteria air pollutants, and air toxics from U.S. industrial facilities. In the next chapter, we describe the procedure by which we have linked together these disparate datasets.

CO_2 Emissions

In January 2012 the USEPA's Greenhouse Gas Reporting Program released the first inventory of GHG emissions from large industrial facilities in the United States. The data cover emissions of CO_2 and five other GHGs from 6,157 facilities in the year 2010. They are based on annual reports that major emitters are required to submit under the EPA's Mandatory Reporting of Greenhouse Gases Rule, which was issued in response to a provision in the 2008 Consolidated Appropriations Act (H.R. 2764; Public Law 110–161), an Act intended to collect accurate and timely GHG data to inform future policy decisions. According to the USEPA, the dataset accounts for more than half of total U.S. emissions of GHGs and provides "nearly complete" coverage for major emitting sectors such as power plants and refineries.[17]

[17] The dataset can be downloaded at http://epa.gov/climatechange/emissions/ghgdata/index.html. For details on the GHG reporting program, see http://epa.gov/climatechange/emissions/ghgrulemaking.html. For brief descriptions of this and other USEPA data sources on GHG emissions, see http://epa.gov/climatechange/emissions/ghgdata/datasets.html.

Prior to January 2012, more fragmentary data on CO_2 emissions from industrial facilities were reported in the 2008 edition of the National Emissions Inventory (NEI). The NEI is compiled every three years by the USEPA, relying primarily on data provided by state and local authorities on emissions of criteria air pollutants and hazardous air pollutants. Historically, the NEI has not included information on CO_2, but in 2008 a number of state agencies voluntarily reported data on CO_2 emissions levels for a subset of facilities, and the NEI in turn reported these data for a total of 3,806 facilities. In addition to the smaller number of facilities and more sporadic coverage of states and sectors, a drawback of these data is that they were not collected under a uniform set of reporting rules. An advantage, however, is that they can be can be easily matched with NEI data on other air pollutants for the same year (and, less easily, with other facility-level emissions data for that year).

Criteria Air Pollutant Emissions

The Clean Air Act (CAA), passed in 1970 and last amended in 1990, 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 developed ambient standards for particulate matter (PM), ground-level ozone, carbon monoxide (CO), sulfur dioxide (SO_2), nitrogen oxides (NOx), and lead. These six pollutants are called "criteria air pollutants," because the USEPA uses health-based and environmentally based criteria to set permissible ambient levels.

The NEI provides facility-level data on emissions of PM, CO, SO_2, NO_X, and volatile organic compounds (VOCs), which react with NO_X in the presence of sunlight to form ground-level ozone. One limitation of these data is that they are produced only at three-year intervals, beginning in 1999. The 2008 NEI is the most recent data available at the time of this writing.

Air Toxics

Air toxics, also known as hazardous air pollutants, are chemicals not subject to NAAQS that are known or suspected to cause serious health effects or environmental effects, some but not all of which are subject to federal or state emissions controls. The USEPA currently has regulations on emissions of 188 air toxics.[19]

The principal source of data on air-toxics emissions in the United States is the Toxics Release Inventory (TRI). Created in response to Congressional legislation in the wake of the 1984 Bhopal disaster, TRI requires industrial facilities to report annual emissions of hundreds of toxic chemicals into air, water, and land. The number of industrial sectors and chemicals covered by TRI has expanded since the first report was issued in 1987. The inventory now includes data on emissions of 593 individually listed chemicals plus 30 chemical compound groups from more than 20,000 facilities nationwide.[20]

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[18] See http://www.epa.gov/ttn/chief/net/2008inventory.html.

[19] The list is available at http://www.epa.gov/ttn/atw/188polls.html. The term "hazardous air pollutants" is sometimes used more restrictively to refer to the regulated subset of air toxics.

[20] For details on TRI, see http://www.epa.gov/tri/tridata/index.html. The other main source of national data on air toxics is the National Air Toxics Assessment (NATA), which provides exposure estimates of air pollution for 33 air toxics at the census-tract level at three-year intervals beginning in 1996, the latest NATA being for 2005. However, NATA does not provide data on emissions from individual facilities. For details, see http://www.epa.gov/ttn/atw/natamain.

Building on TRI data, the USEPA created the Risk-Screening Environmental Indicators (RSEI) to assist regulators in prioritizing facilities and chemicals for attention based on estimated human health impacts. To this end, RSEI incorporates three additional types of information:

1. the relative toxicity of TRI chemicals, which in the case of inhalation toxicity can vary by up to nine orders of magnitude;[21]

2. fate-and-transport modeling of chemical releases, taking into account stack heights, exit-gas velocities, prevailing wind patterns, and so on; and

3. population densities in the localities impacted by the resulting exposures.[22]

The RSEI data for 2007, which we use in this study, include inhalation toxicity weights for 417 TRI chemicals and chemical compound groups. Using these data, we can compare three different aggregation methods for 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).

The RSEI model for 2007 estimates air-pollution exposures for 10,201 1-km^2 grid cells (a 101-km^2) around each facility. The facility-level data, which are publicly available, aggregate impacts across these localities. The RSEI geographic microdata (RSEI-GM), which are used to generate the facility-level RSEI scores, are not publicly available, but the USEPA has made these data available to selected researchers, including the two coauthors of this study. By mapping RSEI-GM data to census information on income, race, and ethnicity, it is possible to calculate facility-level measures of the environmental justice impact of emissions as well as total human health impact (Ash et al. 2009; Ash and Boyce 2011).

4. SOURCE-WISE VARIATIONS IN CO-POLLUTANT INTENSITY

This chapter explains how we merged the available facility-level data on CO_2 emissions and co-pollutant emissions in order to investigate how co-pollutant intensity varies across sources. In the next chapter, we use the same data to examine disparities in emissions burdens between communities with varying demographic characteristics.

To place the data in the context of overall CO_2 emissions, Table 6 reports nationwide stationary-source and mobile-source emissions of CO_2 and also of three criteria air pollutants, SO_2, NO_X, and CO, for the year 2009.[23] Stationary sources accounted for 3.4 billion metric tons (mt) of CO_2 emissions, about two-

[21] For example, 1 kg of benzidine is equivalent, in terms of inhalation toxicity, to 3.4 billion kg of chlorodifluoromethane (HCFC-22).

[22] For details on RSEI, see http://www.epa.gov/opptintr/rsei/pubs/basic_information.html.

[23] The data in Table 4.1 refer to emissions from fossil fuel combustion only. For these three co-pollutants, this accounts for the vast majority of total emissions; hence there is little difference between the narrow and broad measures of co-pollutant intensity.

thirds of the U.S. total. As noted earlier, stationary sources accounted for 94 percent of total SO_2 emissions; hence the simple SO_2 co-pollutant intensity ratio for stationary sources is eight times higher than the ratio for mobile sources. In the cases of NO_X and CO, mobile sources accounted for the majority of emissions; the NO_X and CO co-pollutant intensity ratios for mobile sources are three and nineteen times higher, respectively. This illustrates the sensitivity of co-pollutant intensity rankings to the choice

Table 6: Emissions from Fossil Fuel Combustion: Stationary versus Mobile Sources, 2009

Source	CO_2 Emmissions (million mt)	Co-Pollutant Emmisions (thousand mt)			Co-Pollutant Intensity (kg/mt CO_2)		
		SO_2	NO_X	CO	SO_2	NO_X	CO
Stationary	3447.6	7.167	4.159	4.543	2.08	1.21	1.32
Mobile	1719.7	0.455	6.206	43.355	0.26	3.61	25.21
Total	5167.3	7.622	10.365	47.898	1.48	2.01	9.27

Source: USEPA (2011a, Tables ES-3, ES-10).

4.1 Constructing the Dataset

Empirically analyzing co-pollution intensity at the sector and facility levels requires data on both CO_2 emissions and emissions of other pollutants for the same set of facilities. At present, no single data source combines high-quality information on these two classes of emissions. To measure co-pollutant intensity, we therefore combined data from three distinct sources.[24]

For CO_2, we use 2010 data from the USEPA's Greenhouse Gas Reporting Program (GHGRP), which reports emissions from 6,173 facilities as described above (see Section 3.3).[25] To calculate co-pollutant intensity ratios using a variety of numerators, we matched as many of these facilities as possible with those reported in two other USEPA databases: the National Emissions Inventory (NEI) and the Risk-Screening Environmental Indicators (RSEI). The NEI data provide us with the mass of NO_X, $PM_{2.5}$ and SO_2 emissions.[26] The RSEI data allow us to incorporate hundreds of additional co-pollutants and to make use of the toxicity weights and fate-and-dispersion modeling available within this database.

[24] USEPA's eGRID database provides information on several co-pollutants (NO_X, SO_2, and CH4) as well as CO_2 emissions for electrical generating units, but for our purposes the NEI has two important advantages: it includes $PM_{2.5}$, a co-pollutant of major policy interest; and it covers other important GHG-emitting sectors in addition to electrical power plants.

[25] While the GHGRP data include GHG emissions other than CO_2, such as methane, nitrous oxide and fluorinated gases, CO_2 accounts for 95 percent of all GHGs emitted from the facilities included in the dataset. Here we use only the CO_2 data.

[26] The NEI data include both $PM_{2.5}$ (particulate matter with a diameter of 2.5 micrometers or less, also known as "fine particles") and PM_{10} (which also includes "coarse particles" in the 2.5-10 micrometer range). In the following analysis, we use $PM_{2.5}$ because it has larger mortality effects (per unit mass) and is federally regulated on the basis of annual mean levels (whereas PM_{10} is regulated on the basis of 24-hour exceedances). The two measures are strongly correlated, and analysis of inter-sectoral and inter-facility variations in PM_{10} intensity yield similar results.

The benefit of the fate-and-dispersion modeling in the RSEI data is that it allows us to account for the size of the population affected by the reported air toxics. However, as noted earlier, there are also important health risks from each of the three NEI-listed co-pollutants (NO_X, $PM_{2.5}$ and SO_2). In these cases (for which we lack fate-and-dispersion modeling), we calculated simple population-weighted indices of facility-level co-pollutant burdens by multiplying tons of emissions by the number of people residing within 2.5 miles of each facility. To do this, we included the population counts for census block groups whose centroids were located within 2.5 miles of the facilities with data drawn from the 5-year American Community Survey, 2009. The effects of this population weighting on our results were similar across the three co-pollutants: to conserve space, here we report only the results for population-weighted $PM_{2.5}$, since this pollutant has been the focus of other research, including our own prior California-based analysis.

Matching facility-level data across all three datasets was a rather complicated task, despite the fact that all three come from the USEPA. First, although facilities in the RSEI and GHGRP databases are identified using Federal Registration System Identifiers (FRSIDs), those in the NEI are identified using Emissions Inventory System Identifiers (EISIDs). Overcoming this challenge was relatively easy: USEPA officials were able to provide a crosswalk between EISIDs and FRSIDs. Second, despite the fact that the RSEI and GHGRP databases both included FRSIDs, in many cases the FRSID reported for the same facility in the two datasets did not match.[27]

We began by simply matching facilities across the GHGRP and RSEI databases by their reported FRSIDs. This yielded 1,232 matches. The relatively low number of matches suggested that the FRSIDs often did not coincide between the two files, and this was confirmed by identifying several facilities with the same name and address in both files but with different FRSIDs.

We next made a number of additional matches using zip-code information, first by identifying all zip codes in which there was exactly one facility in both the GHGRP and RSEI databases. This yielded 494 additional facilities. We then examined the name and North American Industrial Classification System (NAICS) code for these facilities and determined that 278 of them were accurate matches.

We then used a Geographic Information System (GIS)–based approach to identify additional matches. This required geocoding by latitude-longitude coordinates all RSEI facilities that we had not yet matched to GHGRP facilities and then geocoding all GHGRP facilities that we had not matched to a RSEI facility, yielding two separate GIS shapefiles. To simplify this laborious task, only GHGRP facilities reporting emissions of at least 10,000 metric tons of CO_2 were geocoded—a restriction that is also applied to facilities included in our analysis below. We then carried out an iterative matching process. In each iteration, we first ran a spatial join between the two shapefiles, merging each GHGRP facility with its nearest RSEI neighbor. We then inspected identifying information from both datasets for each

[27] Information obtained from correspondence with USEPA staff suggests that these identifier discrepancies can occur for at least two reasons. First, when facilities registered to report to the GHGRP, they had the option of using their existing FRSID or registering as a new entity, in which case they were assigned a new FRSID. Second, different USEPA reporting systems may use different definitions for what constitutes an individual "facility."

of the merged facilities (zip code, name, and NAICS code) and selected facilities for which the facilities were determined to be an accurate match. After removing these facilities from the two shapefiles, the process was repeated. In the first iteration, almost 500 additional facilities were determined to match; by the fifth iteration, there were only 9 new matches, at which point the process was stopped. In total, 557 more facilities were matched between the GHGRP and RSEI data using this GIS-based approach.

Adding together the facilities matched by all three techniques described above, we obtained 2,067 facilities with both GHGRP and RSEI data. We then matched these to the NEI data using the USEPA-provided crosswalk.[28] In a few cases, there was more than one EISID (NEI facility) associated with a single FRSID (RSEI/GHGRP facility)—apparently an artifact of different USEPA programs using different definitions for what constitutes a "facility." In these cases, the NEI emissions data associated with those EISIDs was summed by FRSID to achieve a one-to-one correspondence between FRSIDs and EISIDs—thereby creating a facility-level file using the broader of the two definitions of a "facility."

After this step, we were left with 1,629 facilities with information across all three datasets. The relatively modest number of matches (1,629 out of 6,173 GHGRP facilities) is mostly attributable to differences in industry groups covered and reporting requirements. For example, the GHGRP database includes landfills and facilities involved in oil and gas extraction, pipeline transportation, and educational services, but these sectors are not included in the RSEI database. Among the industry groups that are included in all three databases, a smaller fraction of facilities are found in RSEI due to TRI reporting requirements that generally target the larger emitters of toxic pollutants. For example, although there are more than 1,000 facilities from the utilities industry (which includes all power plants) in the GHGRP dataset and more than 4,000 in the NEI dataset, the RSEI dataset includes fewer than 600 such facilities.

The USEPA-provided crosswalk, on which we relied to make matches between the merged GHGRP/RSEI data and the NEI data, could also be the cause of some missed matches. In the entire set of 6,173 GHGRP facilities, only 1,752 can be matched to the NEI data using the crosswalk. This fact, along with the mismatched FRSIDs for some facilities included in both the GHGRP and RSEI data, underscores the importance of one of the recommendations emerging from our work: that the USEPA improve consistency in how facilities are defined and tracked across its various reporting programs.

Notwithstanding these challenges, our final set of 1,629 facilities provides, we believe, a solid basis for the following analysis. Although these are only about one-fourth of the facilities in GHGRP, together they account for more than two-thirds of the total CO_2 emissions. Furthermore, examination of the data suggests that we have a reasonably representative sample of the major industry groups covered in the GHGRP, although facilities with larger GHG emissions are somewhat overrepresented (reflecting the fact that larger facilities are more likely to report to TRI).[29]

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[28] Given that many (835) of the 2,067 facilities with both GHGRP and RSEI data had two different FRSIDs (one from the GHGRP file and one from the RSEI file) we matched NEI facilities first by the FRSID from the RSEI file, then by the FRSID from the GHGRP file. The vast majority of matches were made in initial match using the FRSID from the RSEI file.

[29] For a list of industry groups that reported to the 2010 GHGRP, see http://epa.gov/climatechange/emissions/ghgdata/faq.html#q4.

There is a minor temporal mismatch in our databases: the RSEI data refer to the year 2007 and the NEI data refer to the year 2008, while the GHGRP data refer to the year 2010. The need to combine datasets from different years is not unusual in environmental justice analyses. Year-to-year variations in CO_2 emissions at the facility level generally are small, so we believe that the mismatch introduces only a small amount of noise into our analysis.[30]

Since the primary aim of co-pollutant intensity analysis is to consider policy options with regard to major CO_2 emitters, we focus in what follows on industrial sectors where the GHGRP data indicate that the average facility emits at least 50,000 tons of CO_2 annually. Classifying industries by three-digit NAICS codes, we identified 22 qualifying industries represented in our matched GHGRP-NEI-RSEI dataset. We then excluded individual facilities with very low levels of CO_2 emissions, which we defined as less than 50,000 tons annually for power plants and less than 10,000 tons annually for other industries. These exclusions trimmed our final working sample from 1,629 to 1,542 facilities, which together account for 66 percent of the total CO_2 emissions reported in the GHGRP database.[31]

The left-hand panel of Table 7 reports the number of facilities in each industry in our final sample, while the right-hand panel reports the number of facilities in each state.[32] As can be seen, the facilities are concentrated in certain industries and states. To avoid small-sample biases, we restrict our comparative analysis of co-pollutant intensity variations across industrial sectors and states, respectively, to the eight sectors or 21 states that have at least 30 facilities each.

[30] For example, for the 520 facilities in our sample for which we have data on CO_2 emissions from both the 2008 NEI and the 2010 GHGRP, the correlation between CO_2 emissions in these two years is 0.97.

[31] Focusing on the 22 major CO_2 emitting industries excluded 56 facilities from the sample; an additional 31 were dropped by virtue of the exclusion of individual facilities with low CO_2 emissions.

[32] In Table 7 and subsequent reporting by industry, the labels "power plants" and "petroleum refineries" refer to the 3-digit NAICS industries "utilities" and "petroleum and coal products manufacturers," respectively. Nearly all of the facilities in the "utilities" 3-digit NAICS category are in the "fossil-fuel electric-power generation" subsector, while 85 percent of the facilities in the "petroleum and coal products manufacturers" category are in the "petroleum refineries" subsector.

Table 7: Distribution of GHGRP-NEI-RSEI Data by Industry and State

By Industry		By State			
Industry	**Count**	**State**	**Count**	**State**	**Count**
Power Plants	431	TX	138	WA	24
Chemical Manufacturers	279	PA	101	OK	23
Nonmetallic Mineral Product Manuracturers	180	OH	88	MS	22
Primary Metal Manufacturers	175	LA	75	MD	18
Paper Mills	146	IL	70	NJ	18
Petroleum Refineries	117	IN	67	NE	17
Food Manufacturers	89	AL	58	MA	16
Transportation Equipment Manufacturers	52	MI	54	WY	16
National Security and International Affairs	25	CA	53	UT	14
Beverage and Tobacco Product Manufacturers	10	FL	46	AZ	13
Fabricated Metal Product Manufacturers	7	KY	45	ME	12
Mining (except Oil and Gas)	6	VA	45	OR	11
Textile Mills	5	WI	45	MT	9
Professional, Scientific, and Technical Services	5	NC	43	CT	8
Electrical Equipment, Appliance and Component Manufacturers	4	TN	41	ID	7
Miscellaneous Manufacturers	4	IA	40	DE	6
Oil and Gas Extraction	2	NY	40	NV	5
Support Activities for Mining	1	SC	39	NM	4
Merchant Wholesalers, Durable Goods	1	MO	38	NH	3
Support Activities for Transportation	1	MN	33	HI	2
Administrative and Support Services	1	GA	31	AK	1
Waste Management and Remediation Services	1	KS	26	DC	1
		WV	26	SD	1
		AR	24	VT	1
		CO	24		
	Total 1,542				Total 1,542

4.2 Pollutant Shares by Industrial Sector

With the data in place, our first question is straightforward and helps to motivate the rest of the study: what is the share of total emissions in this sample attributable to various industries? Table 8 presents the data for all of the emissions measures we analyze. The first three co-pollutants are criteria air pollutants with well-established adverse health effects: nitrogen oxides (NO_X), particulate matter ($PM_{2.5}$), and sulfur dioxide (SO_2). Data on these co-pollutants come from the NEI. The RSEI data allow us to calculate three aggregative co-pollutant measures: the total pounds of air toxics released; total pounds weighted by the relative toxicities of the different chemicals; and the facility's "RSEI score," a measure of total human health impact based on the modeled dispersion of the chemicals in the air, the toxicity of the resulting exposures, and the population densities of impacted localities. In the final column, we report a population-weighted measure of health impacts from $PM_{2.5}$ emissions—calculated, as noted earlier, by multiplying total $PM_{2.5}$ emissions from a facility by the number of people living with 2.5 miles of the facility.

Table 8: Share of Total Emissions by Industry in Sample (n = 1,542 facilities)

Industry	CO_2	NO_X	$PM_{2.5}$	SO_2	RSEI Chemicals	RSEI Toxicity-Weighted	RSEI Full-Model Score	Population-Weighted $PM_{2.5}$
Power Plants	79%	78%	65%	88%	63%	11%	9%	52%
Petroleum Refineries	7%	3%	6%	2%	3%	9%	8%	13%
Primary Metal Manufacturers	3%	2%	9%	2%	3%	12%	13%	15%
Chemical Manufacturers	5%	4%	6%	3%	12%	60%	24%	6%
Nonmetallic Miner Product Manufacturers	3%	6%	5%	1%	2%	1%	4%	6%
Paper Mills	2%	5%	7%	3%	12%	5%	2%	4%
Food Manufacturers	1%	1%	1%	1%	3%	1%	1%	3%
Transportation Equipment Manufacturers	0%	0%	0%	0%	1%	0%	26%	2%
All Other Industries	1%	1%	1%	1%	1%	1%	13%	1%
Note: Columns may not add to 100% due to rounding.								

As can be seen in Table 8, power plants are responsible for nearly 80 percent of the CO_2 emissions in our sample, but for a somewhat lesser share of $PM_{2.5}$ (whether weighted by population or not) and of RSEI chemicals aggregated by simple mass—and for a markedly lower share of the toxicity-weighted RSEI emissions and their human health impacts as indicated by the RSEI full-model score. Petroleum refineries have less than one-tenth the carbon footprint of the power plants, yet they have roughly the same public health impact from air toxics as measured by the RSEI score by virtue of the toxicity of the chemicals they emit and the location of the facilities.

These findings nicely illustrate the co-pollutant issue: although it will be important to consider variations within sectors too, one can see immediately that any carbon-charge system in which refineries en masse buy their way out of cleanup and let most of the carbon reduction come instead from power plants would forego significant health co-benefits.

Comparing shares of $PM_{2.5}$ with and without population weights, we find that the share of refineries in population-weighted $PM_{2.5}$ doubles relative to its share of the simple mass of $PM_{2.5}$ emissions. The share of power plants declines somewhat when population weights are used, implying that their emissions tend to impact areas with relatively low population densities. Another noteworthy result is that the share of primary metal manufacturers in population-weighted $PM_{2.5}$ emissions is five times higher than these facilities' share of CO_2 emissions.

4.3 Sectoral Co-Pollutant Intensities

Co-pollutant intensities—defined as the ratio of co-pollutant damages to CO_2 emissions—vary widely, as can be surmised from the sector-level analysis of pollution shares.[33] Table 9 reports average co-pollutant intensities for the eight industrial sectors.[34]

Table 9: Co-Pollutant Intensity Ratios by Industry

Industry	Mean CO_2 output (thousand tons)	NO_x (pounds/ tCO_2)	$PM_{2.5}$ (pounds/ tCO_2)	SO_2 (pounds/ tCO_2)	RSEI Chemicals (pounds/tCO_2)	RSEI Toxicity-Weighted (pounds/tCO_2)	RSEI Full-Model Score (per tCO_2)	Population-Weighted $PM_{2.5}$ (per tCO_2)
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 Miner 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

[33] "Damages" here refer to adverse impacts—as proxied, for example, by the mass of co-pollutant emissions, the toxicity-weighted sum of emissions of the mass of multiple co-pollutants, or the human-health impacts—without necessarily expressing the value of the impacts in monetary terms.

[34] Both here and in the state-level analysis in Chapter 5, we construct the co-pollutant intensity ratios by calculating the CO_2-weighted average of each ratio across all facilities in the same industry (or state). We consider this more appropriate than a simple (unweighted) average, because facilities with greater CO_2 emissions are the ones from which, all else equal, one would expect to see greater CO_2 reductions (and thus greater co-benefits for any given co-pollutant intensity ratio).

Several observations can be made:

1. First, and perhaps most important for climate-policy analysis, power plants and petroleum refineries have the highest mean CO_2 output. Recall that these also are the two sectors most responsible for CO_2 emissions overall, with power plants the largest by far. The mean output measure—emissions at the facility level—again suggests that these will be key sectors for co-pollutant policy attention in any regulatory program.

2. Second, co-pollutant intensities vary considerably across sectors, implying corresponding variations in the co-benefits of CO_2 emissions reductions. For example, the co-pollutant ratio for the RSEI score ratio is ten times higher for the refinery sector than for the power plant sector. Similarly, the population-weighted PM co-pollutant ratio is three times higher for refineries than for power plants. All else being equal, it would be desirable to achieve greater CO_2 emissions reductions in sectors with higher co-pollutant intensities.

3. Third, the pattern of inter-sectoral variations in co-pollutant intensity is not uniform across co-pollutants. While there is some consistency (for example, paper mills, nonmetallic mineral manufacturers and transportation equipment manufacturers generally rank higher, while petroleum refineries, power plants, and chemical manufacturers generally rank lower, with primary metal and manufacturers and food manufacturers in between) there are important differences. For example, primary metal manufacturers rank fairly low in terms of NO_X but fairly high in terms of $PM_{2.5}$, while for power plants the reverse is true.

4. Fourth, in the case of the RSEI air toxics (and $PM_{2.5}$), the pattern again varies depending on the method of aggregation across chemicals. Chemical manufacturers top the list by a wide margin in terms of toxicity-weighted pounds, while transportation equipment manufacturers top the list in terms of both the simple mass measure (just ahead of paper mills) and the full-model score (by a very large margin). Recall that the latter takes into account fate-and-transport and population densities, and what is driving this finding appears to be the location of auto plants in proximity to the populace. This can also be seen in the sector's exceptionally high population-weighted $PM_{2.5}$ intensity ratio.[35]

[35] Although this finding is important from a public-health perspective, note that the CO_2 emissions associated with transportation manufacturers are relatively low compared to those of the other sectors in our sample, implying that this sector may play a less important role in the co-pollutant issues associated with carbon pricing.

An important caveat to recall here is that our co-pollutant data refer to all emissions from the facility, not only to those from fossil fuel combustion. If, in response to climate policy, a 10 percent reduction in fossil fuel use were to be accomplished via a 10 percent cut in the facility's output, this would be likely to lead to commensurate reductions in emissions from all industrial processes. If, however, the same 10 percent reduction in fossil fuel use were to be accomplished entirely via energy efficiency improvements, with no cut in overall output, then co-pollutant emissions would be reduced only insofar as these come directly from fossil fuel combustion.[36]

4.4 Correlations Across Co-Pollutant Intensity Ratios

To shed further light on these variations in co-pollutant intensity, Table 10 reports the correlations among our seven measures of co-pollutant intensity corresponding to the seven co-pollutant categories reported in Table 8 for all 1,542 facilities in our sample. The correlations among the measures are generally positive, but in some cases they are close to zero. The imperfect correlations indicate that facilities that rank high by one co-pollutant intensity measure may not rank high by another.

The correlations are stronger among the criteria pollutant intensity ratios (NO_X, $PM_{2.5}$, and SO_2) and the simple-mass variant of the air-toxics ratio, all of which are calculated using mass of co-pollutant emissions as the numerator. The correlations are considerably weaker for the toxicity-weighted and exposure-weighted air-toxics ratios and for population-weighted $PM_{2.5}$.

Table 10: Interfacility Correlations Across Co-Pollutant Intensity Ratios

	NO_X (pounds/ tCO_2)	$PM_{2.5}$ (pounds/ tCO_2)	SO_2 (pounds/ tCO_2)	RSEI Chemicals (pounds/tCO_2)	RSEI Toxicity-Weighted (pounds/tCO_2)	RSEI Full-Model score (per tCO_2)	Population-Weighted $PM_{2.5}$
NO_X ratio (NEI)	1.00						
$PM_{2.5}$ ratio (NEI)	0.59	1.00					
SO_2 ratio (NEI)	0.50	0.72	1.00				
Simple volume ratio (RSEI)	0.33	0.26	0.18	1.00			
Weighted volume ratio (RSEI)	0.00	0.00	0.00	0.03	1.00		
Full-model score (RSEI)	0.02	0.00	-0.01	0.04	0.18	1.00	
Population-weighted $PM_{2.5}$ ratio (NEI)	0.17	0.30	0.02	0.05	-0.01	0.03	1.00

The low correlations among the three RSEI-based measures and between the simple mass measure of $PM_{2.5}$ and its population-weighted counterpart indicate that rankings for co-pollutant intensity measures can be quite sensitive to methodological choices in the definition of the numerator—simple mass, the use of toxicity weights for aggregation across multiple pollutants, the estimation of total human health impacts by means of exposure modeling (as in the RSEI score), or simpler measures of proximate population density (as shown here in the case of $PM_{2.5}$). This suggests that there can be substantial policy payoffs from implementing the more sophisticated exposure modeling and aggregation methods.

[36] As noted above in Section 3.1, USEPA data indicate that fossil-fuel combustion accounts for approximately 75 percent of NO_X emissions, 40 percent of $PM_{2.5}$ emissions, and 88 percent of SO_2 emissions from stationary sources nationwide, but these shares are likely to vary across industrial sectors.

Even for a single pollutant such as $PM_{2.5}$, although a measure based on sheer tonnage poses lower information requirements, its use for policy purposes is not likely to result in co-benefits comparable to those that could be obtained by simply factoring in population, not to mention more sophisticated measures, like the RSEI score, that better capture the resulting health outcomes.

Though not reported here to conserve space, we also examined correlations among co-pollutant intensity measures across facilities within the specific industrial sectors. We found consistently higher correlations for power plants (and also for paper mills) than for the other sectors.[37] Under a policy framework in which power plants are regulated separately from other sectors, such as the Regional Greenhouse Gas Initiative in the northeastern states, this suggests that focusing on a single co-pollutant intensity measure may yield co-benefits per unit of CO_2 reduction similar in scope to those that could be realized by using a more complex procedure that considers a variety of co-pollutant intensity measures.

4.5 Intra-sectoral Variations in Co-Pollutant Intensity

The analysis above reveals important differences in co-pollutant intensity *between* industrial sectors. In this section we examine differences in co-pollutant intensity among facilities *within* sectors. If co-pollutant intensity is fairly constant for any given measure 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.

To assess the extent of this variation, we use a simple measure called the coefficient of variation. It is calculated by dividing the standard deviation by the mean of a variable across a given set of observations. The normalization makes it useful for assessing relative variation across variables that are measured in different units and with different distributions. In Table 11, for each of our seven measures of co-pollutant intensity we report the coefficient of variation across facilities within each industrial sector as well as across all facilities combined.

[37] Again, the correlations are higher among the simple mass-based measures than between these and the other measures. There is also a strong correlation (Pearson's r = 0.73 in the power plant sector) between the RSEI full-model score and population-weighted $PM_{2.5}$.

Table 11: Coefficient of Variation for Co-Pollutant Intensity by Industrial Sector

	NO_x (pounds/ tCO_2)	$PM_{2.5}$ (pounds/ tCO_2)	SO_2 (pounds/ tCO_2)	RSEI Chemicals (pounds/tCO_2)	RSEI Toxicity-Weighted (pounds/tCO_2)	RSEI Full-Model Score (per tCO_2)	Population-Weighted $PM_{2.5}$ (per tCO_2)
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

Again several observations can be made:

1. There is generally less variation in the co-pollutant ratios within industrial sectors than across all facilities. This is not surprising, since variation in the latter is partly driven by interindustry differences. However, there is a high degree of variation within many industries too, with the coefficients only slightly lower (and sometimes higher) than those for all facilities combined.

2. Power plants are notable for their high variation in all co-pollutant measures other than RSEI toxicity-weighted pounds and full-model score, with the results often indicating even more variation than for all facilities combined.[38] Refineries, primary metal manufacturers, and mineral product manufacturers also show relatively high degree variation in the RSEI simple volume.

3. Consistent with our earlier findings, there are notable differences in the results depending on the co-pollutant considered, with variation tending to increase for the more complex aggregation methods (particularly for the RSEI full-model score, but also for toxicity-weighted RSEI and population-weighted $PM_{2.5}$ to some extent). This implies that it may be useful to develop policies that target specific facilities within industries; it also suggests that trades of carbon permits between firms, even within the same industry, can yield very different health impacts.

4. On the other hand, the sector with the lowest coefficient of variation with regard to the RSEI full-model score and the population-weighted $PM_{2.5}$ measure is the refinery sector. This suggests that a sector-wide approach to carbon pricing in that sector (such as regulating trades into and out of the sector under a cap-and-trade system) may be reasonably efficient.

[38] Different fuel sources (e.g., coal versus natural gas) are one reason for relatively high variations in the power plant sector, as the NEI data do not disaggregate power plant emissions by fuel type.

4.6 Disproportionality in Co-Pollutant Emissions

With the possible exception of the refinery sector, the facility-level differences in co-pollutant intensity described above suggest a rationale for designing climate policy to steer GHG reductions toward particularly high-intensity facilities. For this purpose, it is also useful to examine facility-level differences in total co-pollutant emissions, that is, in the absolute size of the co-pollutant intensity numerator. Facilities can vary dramatically in their total emissions of CO_2 as well as co-pollutants, so even if a facility has a high co-pollutant intensity, if it also has low overall emissions, then there is little room for reductions and associated co-benefits.

One indicator of the interfacility distribution of co-pollutant emissions is the share of total emissions that is attributable to the top 1 percent of facilities. This can be calculated for individual sectors as well as for all sectors combined. Another indicator that considers all facilities (rather than just the top 1 percent) is the Gini index. Though most often applied to measure economic inequality—for example, the concentration of income across households—it can be used to measure the concentration of co-pollutant emissions across facilities. The Gini index ranges from 0 to 1, with 0 in this case indicating that the quantity of emissions for each facility is the same and 1 indicating that all co-pollutants are emitted from a single facility.

Table 12 reports both of these indicators for all 1,542 facilities in our sample. The share of the top 1 percent thus refers to share of the top fifteen facilities for each co-pollutant measure. We also provide a visual representation of facility-level co-pollutant concentration in Figure 5 by plotting the Lorenz curve for emissions of each co-pollutant. The cumulative percentage of emissions, with facilities ranked from highest to lowest emissions, is plotted on the vertical axis and the cumulative percentage of facilities on the horizontal axis.

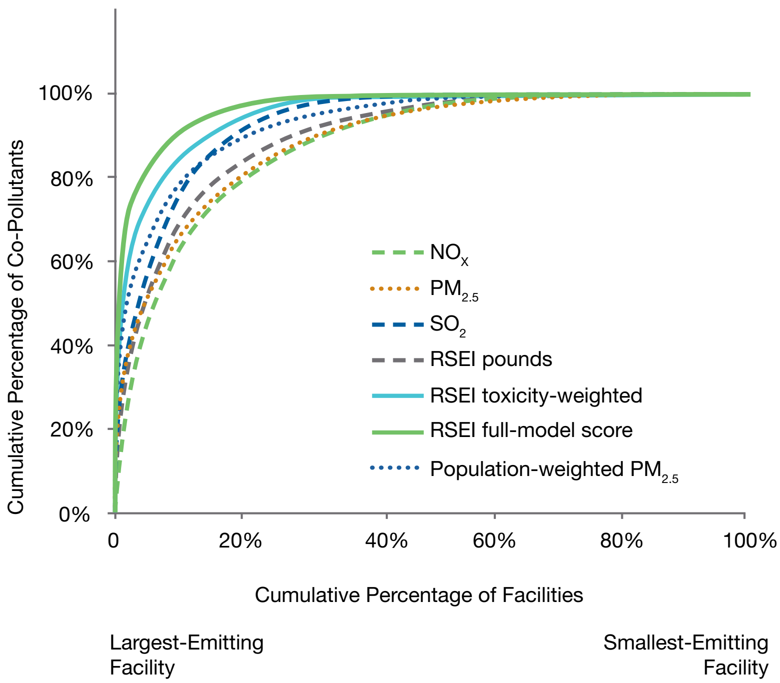
This analysis reveals that among the different measures of co-pollutants, the three that incorporate more information on potential health impacts—the RSEI toxicity-weighted pounds, the full-model score, and the population-weighted $PM_{2.5}$—are the most concentrated. In the case of air toxics, the top 1 percent of facilities accounted for roughly half of the total impacts by both measures. In the case of population-weighted $PM_{2.5}$, the top 1 percent accounted for 35 percent of the total impact.

These results clearly exhibit a high degree of "disproportionality," defined by Berry (2008, p. 239) as "a positively skewed distribution, where a small number of resource users create far more environmental harm than 'typical' group members." This is an important finding, because it suggests that specific policy attention to a small number of "bad actors"—bad in the sense of high co-pollutant impacts by virtue of the quantity and toxicity of their emissions and their proximity to vulnerable populations—likewise could bring about disproportionately positive results.

Table 12: Measures of Facility-Level Co-Pollutant Concentration

Co-Pollutant (absolute quantity)	Share for Top 1% of Facilities	Gini Index
NO_X	13%	0.76
$PM_{2.5}$	26%	0.78
SO_2	23%	0.86
RSEI pounds	22%	0.80
RSEI toxicity-weighted	44%	0.90
RSEI full-model score	59%	0.93
Population-weighted $PM_{2.5}$	35%	0.84

Figure 5 : Facility-Level Co-Pollutant Concentration



To shed further light on interfacility variations, we calculated the share of total co-pollutant emissions that is attributable to top-emitting facilities for each co-pollutant in each of the industrial sectors. For this purpose, we selected the top 5 percent rather than top 1 percent of facilities, given that some of the industrial sectors in our sample have a relatively small number of facilities. To examine the extent of overlap among the top 5 percent across our seven different co-pollutant measures, we also calculated the percentage of facilities in each sector that rank in the top 5 percent for at least one co-pollutant measure. The results are reported in Table 13.

Table 13: Facility-Level Co-Pollutant Concentration by Industry

	Share of Total Co-Pollutants Accounted for by the Top 5% of Facilities							
Industry	**NO_x**	**$PM_{2.5}$**	**SO_2**	**RSEI Chemicals**	**RSEI Toxicity-Weighted**	**RSEI Full-Model Score**	**Population-Weighted $PM_{2.5}$**	**Share of Facilities in Top 5% for *Any* Co-pollutant**
All Facilities	44%	52%	56%	51%	73%	82%	64%	20%
Power Plants	22%	46%	32%	42%	46%	69%	65%	18%
Petroleum Refineries	20%	25%	44%	23%	31%	49%	47%	16%
Primary Metal Manufacturers	47%	41%	63%	62%	76%	62%	54%	19%
Chemical Manufacturers	44%	39%	59%	37%	62%	70%	59%	21%
Nonmetallic Mineral Product Manufacturers	21%	24%	44%	53%	90%	86%	38%	22%
Paper Mills	15%	22%	41%	22%	30%	66%	40%	23%
Food Manufacturers	32%	33%	38%	29%	61%	78%	43%	16%
Transportation Equipment Manufacturers	33%	50%	44%	18%	24%	94%	79%	17%

Similar to the results for all facilities (shown in Table 12 and Figure 5), when we examine individual industrial sectors we again find that the RSEI toxicity-weighted pounds, the full-model score measures, and the population-weighted $PM_{2.5}$ are most concentrated. The top 5 percent of facilities account for 49 to 94 percent of the total RSEI full-model score, depending on the sector; the range for the population-weighted $PM_{2.5}$ is 38 to 79 percent. The industrial sector that exhibits greatest concentration depends on the co-pollutant considered. For example, transportation equipment manufacturers show highest concentration for the RSEI full-model score and population-weighted $PM_{2.5}$, yet lowest concentration for total pounds of RSEI chemicals (both unweighted and toxicity-weighted), again pointing to the importance of population proximity in that sector.

The final column in this table reports the percentage of facilities that rank in the top 5 percent for at least one of our seven co-pollutant measures. If there were perfect overlap—if the same facilities ranked in the top stratum by all measures—this would equal 5 percent; if there were no overlap at all, it would equal 35 percent (5 percent times seven measures). The results are roughly midway between these extremes, ranging from 16 to 23 percent. Only three facilities (all power plants) were in the top 5 percent for all co-pollutant measures. This again suggests that one needs to pay attention to selecting the most salient co-pollutant metric, but the broader point is clear: a large share of co-pollutant emissions, by any measure, are generated by a fairly small percentage of industrial facilities.

To examine further the interfacility variations in co-pollutant emissions, Table 14 reports coefficients of variation by industrial sector. Not surprisingly, the results are similar to those in Table 13, since concentration and variation are closely related. More intriguing is the observation that the results in Table 14 show some consistency with the facility-level variation in co-pollutant intensity (rather than total emissions) reported in Table 11. Sectors with higher variation in one dimension also tend to have higher variation by the other.

Table 14: Coefficient of Variation for Total Co-Pollutants by Industry

	NO_x	$PM_{2.5}$	SO_2	RSEI Chemicals	RSEI Toxicity-Weighted	RSEI Full-Model Score	Population-Weighted $PM_{2.5}$
All Facilities	2.10	3.69	2.98	2.75	5.82	10.88	5.37
Power Plants	1.17	2.85	1.58	2.08	2.34	4.62	5.00
Petroleum Refineries	1.09	1.35	2.26	1.27	1.71	2.67	2.82
Primary Metal Manufacturers	2.66	2.10	3.71	3.58	6.49	380	2.61
Chemical Manufacturers	2.31	2.29	3.11	2.02	3.68	3.97	4.28
Nonmetallic Mineral Product Manufacturers	1.11	1.22	2.28	3.20	4.93	5.92	1.92
Paper Mills	0.88	1.23	2.17	1.21	1.91	3.94	2.09
Food Manufacturers	1.67	1.74	2.12	1.60	3.79	4.14	2.58
Transportation Equipment Manufacturers	1.88	2.97	2.77	1.20	1.45	6.56	5.48

To explore the relationship between the two, we also calculated the correlation between co-pollutant intensity and co-pollutant emissions at the facility level. The results, for each co-pollutant and industrial sector, are reported in Table 15.

Table 15: Correlations Between Total Co-Pollutant Emissions and Co-Pollutant Intensity

Industry	NO_x	$PM_{2.5}$	SO_2	RSEI Chemicals	RSEI Toxicity-Weighted	RSEI Full-Model score	Population-Weighted $PM_{2.5}$
All Facilities	0.04	0.17	0.16	0.11	0.70	0.96	0.29
Power Plants	0.03	0.18	0.11	0.14	0.18	0.86	0.89
Petroleum Refineries	0.18	0.24	0.56	0.37	0.62	0.60	0.32
Primary Metal Manufacturers	0.13	0.25	0.79	0.29	0.90	0.93	0.27
Chemical Manufacturers	0.19	0.49	0.46	0.25	0.72	0.70	0.36
Nonmetallic Mineral Product Manufacturers	0.21	0.46	0.42	0.82	0.73	0.97	0.78
Paper Mills	0.22	0.61	0.53	0.51	0.73	0.85	0.59
Food Manufacturers	0.19	0.18	0.33	0.29	0.92	0.74	0.43
Transportation Equipment Manufacturers	0.93	0.99	0.95	0.90	0.93	1.00	1.00

The correlations are invariably positive, which is to be expected, since total emissions are the numerator in the co-pollutant intensity ratio. More surprising, perhaps, is the finding that in many cases the correlation is rather low: in the first four columns, where co-pollutants are measured in simple mass of emissions, about half of the correlations are less that 0.3. As we move to RSEI-based measures that rely on more sophisticated aggregation techniques—toxicity-weighted pounds and full-model score—and population-weighted $PM_{2.5}$, we find that the correlations generally rise. For the latter measures, the general finding is that big polluters also tend to have higher co-pollutant intensity. Put differently, it suggests that the same facilities that are likely to yield the greatest co-benefits per unit of CO_2 abated also tend to also have the greatest potential for co-pollutant reductions in absolute terms.

5. SPATIAL VARIATIONS IN CO-POLLUTANT INTENSITY

This chapter examines spatial variations in co-pollutant intensity across states, metropolitan areas, and localities, enabling us to explore the environmental justice dimensions of co-pollutant impacts. Working with the 1,542-facility sample described above, we once again find significant variations, this time across geography instead of across sectors and facilities. But we also can discern an important pattern: the sectors that emit the most CO_2 (power plants, refineries, and chemical manufacturers), and hence are likely to have the most significant co-pollutant impacts under any climate-regulatory system, generally have facilities that are more concentrated in low-income communities and communities of color.

5.1 State-wise Variations

Table 16 reports average co-pollutant intensities for industrial facilities for all states with at least 30 facilities in our sample. Again we observe considerable heterogeneity. In the case of population-weighted $PM_{2.5}$, for example, the impact per ton of CO_2 is about 50 times greater in New York than in Wisconsin. Note that interstate differences can arise in two ways: (a) due to the varying industrial composition across different states; and (b) due to interstate differences in co-pollutant intensities for any given industrial sector, arising, for example, from different state regulatory regimes or different plant vintages.

Table 16: Co-Pollutant Intensities by State

State	Count	NO_x (pounds/ tCO_2)	$PM_{2.5}$ (pounds/ tCO_2)	SO_2 (pounds/ tCO_2)	RSEI Chemicals (pounds/tCO_2)	RSEI Toxicity-Weighted (pounds/tCO_2)	RSEI Full-Model Score (pounds/tCO_2)	Population-Weighted $PM_{2.5}$ (pounds/tCO_2)
TX	138	1.66	.017	4.41	0.18	1,776	0.064	1,553
PA	101	3.47	0.22	14.51	0.55	495	0.055	4,830
OH	88	3.81	0.83	14.64	0.83	620	0.048	9,981
LA	75	2.02	0.44	3.70	0.40	2,352	0.041	4,344
IL	70	2.40	0.22	5.92	0.22	885	0.021	5,536
IN	67	3.40	1.34	9.52	0.38	415	0.038	10,400
AL	58	3.71	0.49	10.02	0.45	392	0.020	4,400
MI	54	3.40	0.19	9.79	0.58	299	0.014	5,624
CA	53	1.07	0.22	0.37	0.10	75	0.043	9,577
FL	46	3.23	0.37	5.94	0.48	167	0.007	3,909
KY	45	3.47	0.89	8.11	0.56	1,514	0.016	3,589
VA	45	4.33	0.22	9.21	0.76	4,953	0.087	2,017
WI	45	3.14	0.03	8.05	0.30	206	0.023	419
NC	43	2.10	0.65	7.29	0.97	347	0.017	3,249
TN	41	4.21	0.34	9.72	0.75	482	0.037	1,914
IA	40	2.26	0.31	4.95	0.24	216	0.009	2,560
NY	40	2.96	0.28	6.17	0.23	251	0.088	21,430
SC	39	3.13	0.87	10.15	0.86	799	0.016	2,708
MO	38	2.60	0.17	8.27	0.13	176	0.008	721
MN	33	3.20	0.23	3.87	0.16	250	0.029	3,135
GA	31	3.12	0.19	13.97	0.75	291	0.005	1,191
All Other States	326	3.37	0.27	5.14	0.33	600	0.102	4,715
Average (unweighted)	-	3.00	0.41	7.90	0.46	798	0.036	4,900

5.2 Environmental Justice Results

What is the pattern of burdens by race and income? To answer this question, we calculate how much of each facility's RSEI-score impacts are borne by African Americans, Latinos, all racial and ethnic minorities combined (including Asian Pacific Islanders and Native Americans) and by the poor (households with incomes below the federal poverty line). To do this, we use the RSEI-GM (geographic microdata) to track each facility's air toxics to the specific neighborhoods impacted and then examine census data on the demographics of those neighborhoods.[39] We also calculate a comparable measure for population-weighted $PM_{2.5}$, measuring the share of each ethnic and income group in the population living within 2.5 miles of the facilities.

Table 17 reports the results for the RSEI air toxics, and Table 18 reports the results for population-weighted $PM_{2.5}$. If co-pollutant exposure was evenly distributed across all racial, ethnic, and economic groups, their impact shares would correspond to their respective shares in the national population. For comparison, the latter is reported in the final row of each table.[40]

The RSEI score results show disproportionate exposures for African Americans in all sectors except for nonmetallic mineral product manufacturing and paper mills, with particularly high shares of exposure in the food manufacturing and petroleum refining sectors. Latinos are disproportionately burdened in four of the eight sectors, with chemical manufacturing and petroleum refining topping the list and power plants close behind. Overall, petroleum refineries pose the most disparate burden on people of color. They also pose the most disparate burden on the poor.

Table 17: 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%

[39] For details on our methodology, see Ash et al. (2009) and Ash and Boyce (2011).

[40] Because the USEPA uses 2000 Census data to calculate RSEI scores, we report national demographics for 2000 in Table 17. For population-weighted $PM_{2.5}$, for which we used census-tract information from the 5-year pooled American Community Survey (2005–2009), we report the corresponding national demographics.

The population-weighted $PM_{2.5}$ results are similar for race and ethnicity (here the baseline demographic comparison is to a somewhat later time period in which the percentage of the nation's population that is nonwhite has risen). What is different is that the share of impact borne by the poor is substantially higher (even compared to the higher poverty rate in the later period). Primary metal manufacturers and refineries pose a particularly high burden on African Americans, whose share of their emissions impacts is nearly three times their share in the population. Only two sectors, paper mills and food manufacturers, do not disproportionately burden minorities, and all sectors disproportionately burden the poor.

Table 18: 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%

Comparing these rankings to the sectoral sources of carbon emissions, we find that the three industrial sectors for which carbon reduction may be the most important—power plants, refineries and chemical manufacturing, which together account for more than 90 percent of industrial CO_2 emissions in our sample—also have the most disproportionate impacts on minorities as shown by the RSEI measure and rank among the top five as shown by population-weighted $PM_{2.5}$. Any regulatory program that reduces co-pollutant emissions along with carbon emissions in these sectors, therefore, is likely to reduce environmental disparities and advance environmental justice objectives. By the same logic, any regulatory program that fails to incorporate air-quality co-benefits into its design will not only forgo public health benefits, but also is likely to violate the official federal directives to consider environmental equity in rule and decision making.

To examine geographic variations in environmental justice disparities, Tables 19 and 20 present the same data disaggregated by state. The first four columns of each table report the average share of co-pollutant burdens borne by different demographic groups. The next four columns report the state population share for each group, and the last four columns report the difference between the two shares. This final measure, termed the "discrepancy" by Ash et al. (2009), is the difference between the share of co-pollutant exposure actually borne by each demographic group and the share that would be expected if exposure were evenly disturbed across demographic groups in the state.

Table 19: Environmental Justice by State: Air Toxics

State	Black Share	Hispanic Share	Minority Share	Poor Share	Population Percent Black	Population Percent Hispanic	Population Percent Minority	Population Percent Poor	Black Discrepancy	Hispanic Discrepancy	Minority Discrepancy	Poor Discrepancy
TX	13.8%	41.3%	58.6%	15.1%	11.5%	32.0%	47.6%	15.4%	2.2%	9.3%	11.1%	-0.2%
PA	13.5%	12.6%	29.1%	13.9%	10.0%	3.2%	15.9%	11.0%	3.5%	9.4%	13.2%	2.9%
OH	10.0%	1.8%	14.4%	10.8%	11.5%	1.9%	16.0%	10.6%	-1.5%	-0.1%	-1.6%	0.2%
LA	46.8%	3.6%	52.6%	19.6%	32.5%	2.4%	37.5%	19.6%	14.4%	1.2%	15.1%	0.0%
IL	19.1%	24.8%	47.1%	13.9%	15.1%	12.3%	32.2%	10.7%	4.0%	12.5%	14.9%	3.3%
IN	12.9%	4.3%	20.1%	10.4%	8.4%	3.5%	14.2%	9.5%	4.5%	0.8%	5.9%	0.9%
AL	43.6%	1.9%	47.1%	17.7%	26.0%	1.7%	29.7%	16.1%	17.6%	0.2%	17.4%	1.6%
MI	26.0%	8.6%	39.5%	15.7%	14.2%	3.3%	21.4%	10.5%	11.7%	5.3%	18.1%	5.2%
CA	10.8%	48.9%	74.3%	16.3%	6.7%	32.4%	53.3%	14.2%	4.1%	16.5%	21.0%	2.1%
FL	13.0%	10.7%	27.2%	10.6%	14.6%	16.8%	34.6%	12.5%	-1.6%	-6.1%	-7.4%	-1.9%
KY	6.9%	1.1%	10.0%	15.1%	7.3%	1.5%	10.7%	15.8%	-0.4%	-0.4%	-0.7%	-0.7%
VA	8.7%	1.5%	13.8%	16.8%	19.6%	4.7%	29.8%	9.6%	-11.0%	-3.1%	-16.1%	7.2%
WI	2.2%	10.2%	21.5%	11.2%	5.7%	3.6%	12.7%	8.7%	-3.5%	6.6%	8.8%	2.6%
NC	29.2%	3.2%	34.7%	13.2%	21.6%	4.7%	29.8%	12.3%	7.6%	-1.5%	4.9%	0.9%
TN	16.1%	1.5%	19.5%	14.7%	16.4%	2.2%	20.8%	13.5%	-0.3%	-0.6%	-1.3%	1.2%
IA	5.4%	5.0%	13.9%	8.6%	2.1%	2.8%	7.4%	9.1%	3.3%	2.2%	6.5%	-0.5%
NY	18.4%	10.4%	33.2%	14.6%	15.9%	15.1%	38.0%	14.6%	2.6%	-4.7%	-4.8%	0.0%
SC	35.6%	2.8%	41.8%	13.2%	29.5%	2.4%	33.9%	14.1%	6.0%	0.4%	7.9%	-0.9%
MO	27.5%	2.5%	32.5%	13.1%	11.2%	2.1%	16.2%	11.7%	16.3%	0.3%	16.2%	1.3%
MN	4.5%	4.4%	15.6%	6.1%	3.5%	2.9%	11.8%	7.9%	1.1%	1.5%	3.7%	-1.8%
GA	34.9%	4.7%	42.5%	13.4%	28.7%	5.3%	37.4%	13.0%	6.2%	-0.6%	5.2%	0.4%
All Other States	14.2%	11.3%	34.8%	12.9%	8.9%	9.7%	26.0%	11.3%	5.4%	1.6%	8.8%	1.6%
All States (2000)	15.7%	14.4%	36.3%	13.6%	12.3%	12.5%	30.9%	12.4%	3.4%	1.8%	5.5%	1.2%

Table 20: Environmental Justice by State: Population-Weighted $PM_{2.5}$

State	Black Share	Hispanic Share	Minority Share	Poor Share	Population Percent Black	Population Percent Hispanic	Population Percent Minority	Population Percent Poor	Black Discrepancy	Hispanic Discrepancy	Minority Discrepancy	Poor Discrepancy
TX	12.4%	49.5%	65.2%	21.0%	11.3%	35.9%	52.2%	16.8%	1.1%	13.6%	13.0%	4.2%
PA	23.8%	4.9%	35.2%	19.1%	10.1%	4.7%	18.5%	12.1%	13.6%	0.3%	16.7%	7.0%
OH	11.9%	4.6%	19.4%	18.6%	11.6%	2.6%	17.5%	13.6%	0.3%	1.9%	1.9%	5.0%
LA	62.5%	2.0%	66.2%	29.0%	31.7%	3.3%	38.1%	18.4%	30.8%	-1.3%	28.1%	10.5%
IL	17.1%	28.7%	50.8%	17.9%	14.5%	14.6%	34.8%	12.4%	2.6%	14.1%	16.0%	5.5%
IN	24.3%	9.2%	36.4%	20.3%	8.6%	5.1%	16.8%	13.2%	15.8%	4.1%	19.6%	7.1%
AL	69.7%	2.6%	73.9%	26.3%	26.0%	2.8%	31.5%	16.8%	43.7%	-0.2%	42.4%	9.5%
MI	13.8%	17.6%	34.8%	23.9%	13.9%	4.0%	22.5%	14.5%	0.0%	13.6%	12.3%	9.4%
CA	8.7%	48.9%	72.7%	15.8%	6.0%	36.1%	57.5%	13.2%	2.7%	12.8%	15.3%	2.6%
FL	13.4%	16.8%	34.3%	13.6%	14.8%	20.6%	39.5%	13.2%	-1.4%	-3.7%	-5.2%	0.4%
KY	4.0%	1.2%	7.6%	21.2%	7.4%	2.4%	12.3%	17.4%	-3.4%	-1.2%	-4.7%	3.8%
VA	31.6%	6.0%	42.2%	14.9%	19.3%	6.7%	33.0%	10.1%	12.3%	-0.6%	9.2%	4.9%
WI	4.5%	10.0%	20.8%	13.2%	5.9%	4.9%	14.9%	11.1%	-1.3%	5.1%	5.9%	2.1%
NC	17.3%	6.7%	28.2%	12.3%	20.9%	7.0%	32.6%	15.1%	-3.6%	-0.3%	-4.3%	-2.7%
TN	16.9%	3.0%	22.0%	17.2%	16.4%	3.7%	23.0%	16.1%	0.5%	-0.7%	-1.1%	1.1%
IA	3.6%	4.5%	12.1%	14.0%	2.4%	4.1%	9.6%	11.4%	1.2%	0.5%	2.5%	2.7%
NY	14.9%	26.2%	51.6%	18.9%	14.6%	16.3%	39.7%	13.8%	0.2%	9.9%	11.9%	5.0%
SC	34.8%	3.9%	40.8%	18.6%	28.1%	4.0%	34.9%	15.8%	6.8%	-0.2%	5.8%	2.8%
MO	12.5%	2.6%	17.8%	15.0%	11.1%	3.1%	17.9%	13.7%	1.4%	-0.5%	-0.1%	1.3%
MN	4.5%	4.3%	14.2%	9.2%	4.3%	4.0%	14.6%	10.0%	0.3%	0.3%	-0.4%	-0.8
GA	31.3%	6.0%	40.5%	18.2%	29.4%	7.77%	41.6%	15.0%	1.9%	-1.7%	-1.1%	3.2%
All Other States	14.5%	24.5%	50.4%	15.7%	8.4%	11.8%	28.0%	12.3%	6.2%	12.8%	22.4%	3.5%
All States (2005-2009)	20.1%	16.4%	42.6%	18.6%	12.1%	15.1%	34.2%	13.5%	8.0%	1.3%	8.4%	5.2%

The RSEI-based results show especially large discrepancies for African Americans. In four states—Alabama, Missouri, Louisiana, and Michigan—their share of the air-toxics co-pollutant burden exceeds their share of the state's population by more than 10 percent. In Missouri, for example, the share of African Americans in air-toxics impacts is 27.5 percent, whereas their share in the state's population is only 11.2 percent. Comparably large discrepancies for Latinos are found in California and Illinois. Another notable state is Virginia, which shows the largest poor discrepancy (7.2 percent) despite a large negative minority discrepancy, implying that poor white Virginians are most likely to bear disparate burdens from co-pollutant emissions in the case of air toxics.

The results are somewhat different for population-weighted $PM_{2.5}$. Table 20 again shows generally larger disparities overall, as was the case when we looked at differences by industry. African Americans bear a starkly higher share of the $PM_{2.5}$ burden (as compared to the RSEI air-toxics burden) in Alabama and Virginia and a substantially higher share in Louisiana, Indiana and Pennsylvania. The same can be seen for Latinos in New York and Michigan and for the poor in Louisiana and Alabama.

5.3 Cumulative Impacts and Geographic Disparity

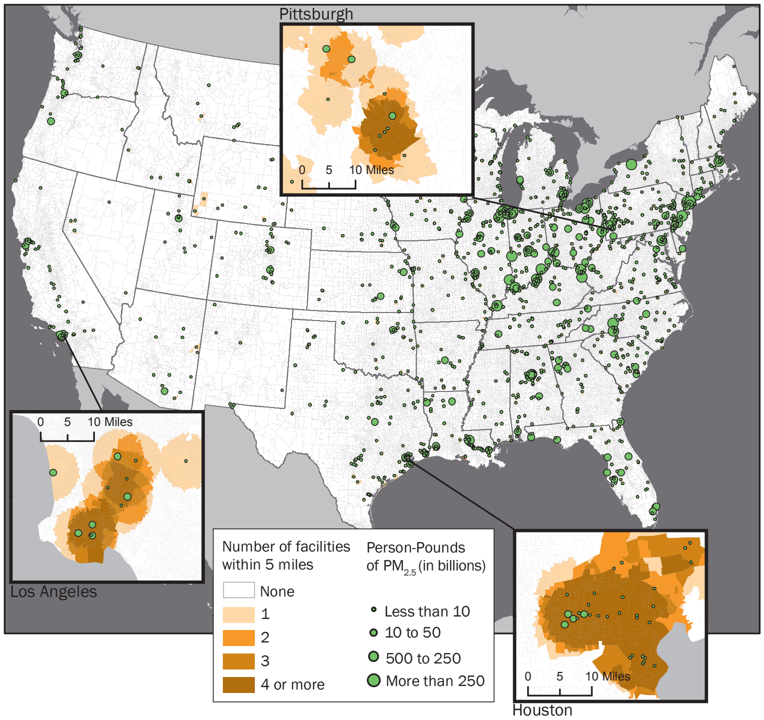
Contemporary environmental justice analysis has moved beyond the distribution of impacts from individual pollution sources to also consider issues of cumulative impacts. Such impacts are defined as "exposures, public health or environmental effects from the combined emissions and discharges, in a geographic area, including environmental pollution from all sources, whether single or multi-media, routinely, accidentally, or otherwise released" (Alexeeff et al. 2010, p. vii). The reason for this shift is that many neighborhoods are affected by multiple hazards—and the argument is that in considering health effects, one should not only consider sources in isolation, but also the proximity of one source to another, since it is the sum total of pollution from multiple sources that generates health impacts.

Why is this important in the context of climate policy and co-pollutants? The clustering of emitters is not consequential in the case of CO_2, since wherever you reduce a certain amount of carbon emissions, whether from a single industrial facility or from a group of facilities, the effect on climate change is the same. On the co-pollutant side, however, clustering matters: if a cluster of facilities is persuaded to reduce its emissions rather than, say, buy emission allowances or offset credits, then the surrounding neighborhood could find its overall air quality substantially improved. This sort of geographic externality is a reason why some have talked about "no trading" zones in the context of cap-and-trade, a point to which we return in the next chapter.

Is geographic clustering a feature of our data on industrial emissions of co-pollutants? In an earlier study on California's electric utility, petroleum refinery, and cement industries, we showed that clustering does occur in the state and results in significant cumulative PM impacts in certain areas (Pastor et al. 2010a, 2010c). To explore this issue on a national scale, we geocoded and mapped the facilities included in the analysis above and ran a cluster routine to identify metropolitan areas where there is a set of facilities in close proximity to one another (see Figure 6). To illustrate the existence of clusters, we focus on Los Angeles, Houston, and Pittsburgh. The first two have been particularly important sites of struggle over environmental justice issues, and Pittsburgh has seen community concerns as well.

Figure 6 provides a visual sense of clustering in $PM_{2.5}$ emissions in our sample. The size of the facility icons varies with the population-weighted pollution load (again using the number of people residing within 2.5 miles of the facility). As can be seen, although most areas of the country have no facilities, there is noticeable clustering of facilities in many places shown on the map, and those areas often have the bigger generators of population-weighted $PM_{2.5}$. In the three highlighted locations, we also calculated the number of facilities located within 5 miles of each 2000 census block group, with darker shades in the inset maps indicating block groups with more proximate facilities.

Figure 6: Clustering of Population-Weighted $PM_{2.5}$ Emissions



To examine whether industrial sources—which include the large CO_2 emitters in our analysis—are important contributors to overall air quality–related health risks in the vicinity of such clusters, we again use 2005 air-toxics data from NATA, which provides census-tract-level coverage for the United States.[41] Recall from our earlier analysis that the NATA data suggest that point sources account for a large share of neurological risk in areas with high neurological risk overall, but that this is not the case with respect to cancer or respiratory risk from air toxics. The question is whether clustered CO_2-emitting facilities in our analysis are located in areas where point sources are a major factor in health risks from air toxics.

The maps in Figures 7 and 8 depict the point-source share of total cancer risk and neurological risk, respectively, in the 2005 NATA data. There is a noticeable visual correlation between the clusters of facilities and the point-source share of risk under both measures. Among the three localities examined, the share of point sources in cancer risk in Pittsburgh and neurological risk in Houston are particularly notable.

Finally, Table 21 presents data on the relationship between the point-source share of cancer and neurological risks from air toxics in relation to proximity to all 1,542 facilities in our sample. Again we see that when facilities cluster, the share of risk from point sources rises. Considering the number of facilities within 2.5 miles of a census tract as a measure of such clustering, the point-source shares of cancer risk and neurological risk rise monotonically with each additional facility inside that distance band. The share is larger for neurological risk (roughly double or more at each step), surpassing 30 percent of risk when there are three facilities within 2.5 miles.

[41] We can't conduct a comparable analysis for $PM_{2.5}$ because we do not have comprehensive data on PM emissions from all sources nationwide.

A similar pattern is found with regard to variations in proximity to any facility: the closer one's census tract is to any facility, the higher the point-source share of cancer risk and neurological risk. These patterns suggest that the large CO_2-emitting facilities analyzed in this study are likely to be important contributors to health risks of their residential neighbors. We believe that this sort of spatial analysis should be incorporated into policy design for carbon reduction.

Figure 7: Point-Source Share of Air-Toxics Cancer Risk, 2000 Census Tracts

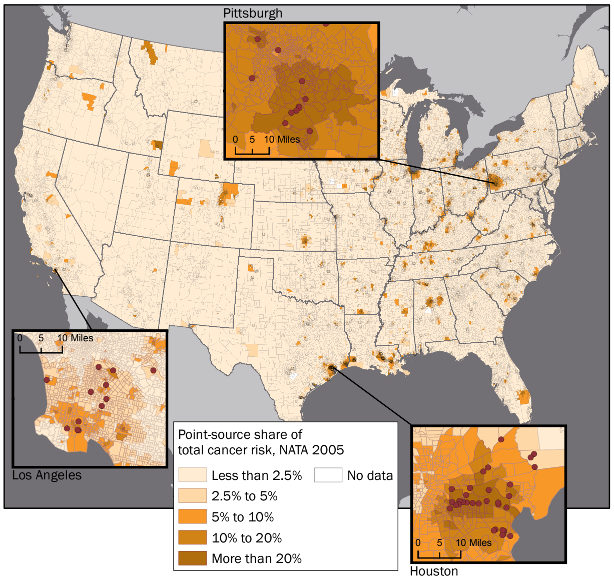


Figure 8: Point-Source Share of Air Toxics Neurological Risk, 2000 Census Tracts

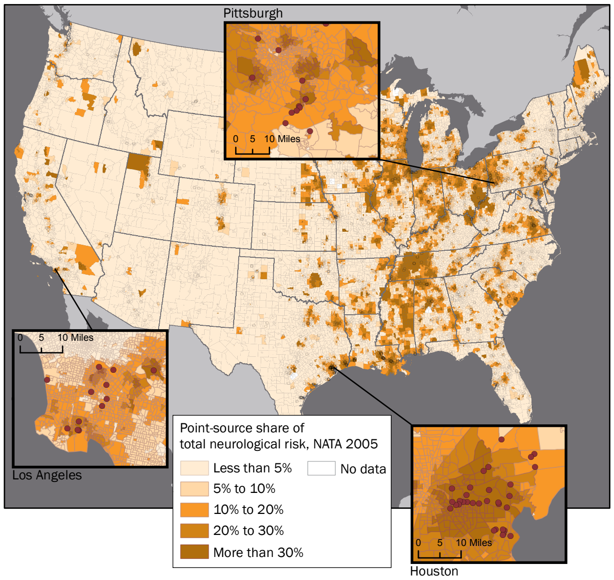


Table 21: Point-Source Share of Air Toxics Risks in Relation to Proximity to Large CO_2-Emitting Facilities

	Source Type						
	Point	Area	Onroad	Nonroad	Background	Secondary	Total
Cancer Risk							
By number of GHG-emitting facilities with 2.5 miles							
0	2%	11%	12%	5%	24%	46%	100%
1	5%	14%	16%	6%	21%	37%	100%
2	6%	16%	17%	8%	19%	34%	100%
3	11%	11%	18%	6%	20%	33%	100%
4	16%	11%	18%	5%	17%	33%	100%
5	16%	10%	19%	6%	17%	31%	100%
6	24%	11%	9%	4%	18%	32%	100%
7	32%	10%	4%	9%	16%	29%	100%
By *any* GHG-emitting facilities within given distance							
.5 mile	8%	14%	17%	6%	20%	35%	100%
1 mile	7%	15%	16%	6%	20%	36%	100%
2.5 miles	6%	14%	17%	7%	20%	36%	100%
5 miles	5%	14%	17%	6%	21%	38%	100%
Neurological Risk							
By number of GHG-emitting facilities with 2.5 miles							
0	10%	26%	9%	7%	48%	0%	100%
1	20%	31%	10%	7%	31%	0%	100%
2	23%	34%	11%	7%	26%	0%	100%
3	31%	24%	10%	7%	27%	0%	100%
4	33%	20%	14%	7%	27%	0%	100%
5	34%	25%	11%	6%	24%	0%	100%
6	43%	23%	5%	4%	25%	0%	100%
7	63%	17%	2%	4%	15%	0%	100%
By *any* GHG-emitting facilities within given distance							
.5 mile	27%	29%	9%	6%	28%	0%	100%
1 mile	26%	30%	9%	6%	29%	0%	100%
2.5 miles	21%	32%	10%	7%	30%	0%	100%
5 miles	17%	32%	11%	7%	33%	0%	100%

6. POLICY OPTIONS

This chapter presents options for integrating co-pollutants into climate policy. Because the relevance of these options may depend on the policies that are used to reduce carbon emissions themselves, we first sketch briefly the major climate policies that have been under consideration in the United States. We then discuss arguments for and against integrating co-pollutants into climate policy. Finally, we consider a variety of options for doing so.

6.1 Carbon Policy Context

Policies to reduce carbon emissions from fossil-fuel combustion fall into two broad types: quantitative controls, such as emission standards or mandated technologies, and price-based policies, such as marketed permits or a carbon tax.[42] The USEPA is now moving to initiate quantitative controls on CO_2 emissions, following the 2007 U.S. Supreme Court decision that CO_2 meets the Clean Air Act (CAA) definition of a "pollutant" and the failure of Congress to pass climate legislation.

Quantitative controls and price-based policies are not mutually exclusive. Federal regulation of SO_2 emissions from power plants, for example, has combined conventional quantitative controls with a cap-and-trade program. Similarly, California will implement its Global Warming Solutions Act of 2006 (AB 32) by means of both quantitative controls, such as a renewable portfolio standard for electricity and low-carbon fuel standard for transportation fuels, and a cap-and-trade program for CO_2 emissions.

The case for price-based policies to supplement (or possibly replace) quantitative controls typically is framed in terms of cost-minimization and flexibility: by allowing polluters to choose their level and methods of emission reduction, based on the marginal abatement costs compared to the price of emissions, the policy can incentivize a wide range of measures to reduce emissions to achieve the overall objective at the lowest total cost. This is the static efficiency logic depicted in Figure 1.[43]

There are two other attractions of price-based policies. First, price-based policies can provide incentives for technological innovation above and beyond those provided by conventional regulations, a phenomenon sometimes called "dynamic efficiency."[44] Second, price-based policies offer an opportunity to implement the principle that the natural environment—in this case, the environment's capacity to serve as a sink for the disposal of carbon emissions—belongs to the people. If permits are auc-

[42] We use the term "price-based" to refer to both carbon taxes and cap-and-permit systems, since in both cases polluters change their behavior in response to price signals.

[43] There are differences, however, between how price-based incentives work in theory and how they have worked in practice. In an analysis of how coal-fired power plants have responded to a trading program for NO_x emissions in the eastern United States, Fowlie and Muller (2011) find that firms often did not, in fact, choose cost-minimizing compliance strategies. The incentive structure under rate-of-return regulation of utilities (which sets a rate of return on investment regardless of whether it is cost-minimizing) may help to explain this inefficiency.

[44] In an analysis of the SO_2-trading program for electric utilities, introduced under the 1990 CAA amendments, Carlson et al. (2000) find that static efficiency gains were small (at least as of 1995 and 1996), but that technological change (possibly spurred by the incentives created by the trading program) generated large cost savings. See also Burtraw (1996) on the role of the SO_2 program in accelerating cost-saving technological change.

tioned (or, equivalently, pollution is taxed), rather than given away free of charge, the polluter pays and the money goes to the government on behalf of the public. If the revenue from permit auctions (or pollution taxes) is then returned directly to the people as "dividends" and community benefit funds, the principle of ownership by the people would be further strengthened.[45]

Price-based policies often have encountered opposition, however, from environmental justice advocates on the grounds that they will allow co-pollutant "hot spots" to persist in their communities and perhaps even worsen—exactly the sort of concerns we have raised in this study. In California, environmental justice advocates filed a lawsuit that attempted to block the cap-and-trade program under AB 32.[46]

Because price-based instruments are part of the policy mix in California, and because they have figured prominently in efforts to pass climate legislation at the federal level, it is particularly important to consider how air-quality co-benefits and the equity concerns raised by environmental justice advocates can be integrated into pricing strategies.

6.2 Should Co-Pollutants Be Integrated into Climate Policy?

Let us start first with the "do nothing" option, by which we here do not mean having no climate policy, but rather leaving co-pollutants out of climate policy. This is the recommendation of Schatzki and Stavins (2009), who argue that co-pollutants are best treated as a separate issue to be addressed independently of climate policy. This stance appeals theoretically to some economists in that it echoes the principle advanced by Tinbergen (1952) that consistent economic policy requires the number of policy instruments to equal the number of policy targets. For some environmental advocates it also has political appeal: they argue that it is difficult enough to craft a viable climate policy without bringing co-pollutants into the picture.

Against this option we can make several points. First, we cannot safely assume that co-pollutant impacts will be adequately addressed by other policies. The large health costs attributable to co-pollutants, discussed in Chapter 2, and the potentially large co-benefits of climate policy attest to this fact. Furthermore, recent developments favor an integrated approach to co-pollutants and climate policy: the past decade has seen growing interest in multipollutant as opposed to single-pollutant strategies for air-quality management.[47]

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[45] For discussion, see Boyce and Riddle (2007).

[46] For a summary of their concerns, see http://www.ejmatters.org/docs/Summary%20of%20AB32%20lawsuit%20[5pg].pdf. In response to the lawsuit, in March 2011 a California court put a hold on AB 32 implementation. The injunction was lifted in June 2011 pending a final ruling by the California Court of Appeals. In June 2012 the Appellate Court upheld the lifting of the injunction and dismissed the case (Farber 2012, pp. 43–45).

[47] A shift to multipollutant strategies was recommended by the National Academy of Sciences (2004, p. 12). For discussion, see also McCarthy et al. (2010).

Second, even if we were to assume that co-pollutants are adequately regulated, co-pollutant damages would not be eliminated altogether. Assuming these damages continue to vary across polluters, the total marginal benefit from CO_2 abatement (including co-benefits) will be variable too, and a first-best climate policy would take this variation into account, following the logic depicted in Figure 3.

Third, the administrative costs of incorporating co-pollutants into pricing policy could be modest. As we have noted, price-based instruments and quantity-based instruments can function in tandem.[48] Moreover, if, as our analysis suggests, a relatively small number of sectors and facilities are most problematic on the co-pollutant side, selective targeting can effectively tackle the bulk of the issues.

Finally, the political virtues of simplicity must be weighed against the political costs of failing to address concerns about co-pollutants in climate policy. These costs were illustrated rather dramatically by the lawsuit brought by environmental justice groups in California. Bringing air-quality co-benefits explicitly into climate-policy design could widen and deepen the constituency for an active policy.

6.3 Co-Pollutant Policy Options

Our policy menu starts from the premise that something should be done to integrate co-pollutants into climate policy. We discuss six options for doing so:[49]

Monitor Impacts of Climate Policy on Co-Pollutants

Minimally, we believe there is a compelling case for monitoring co-pollutant emissions to evaluate the impacts of climate policy and to provide a basis for policy modifications, should these impacts prove to be unacceptable. The analysis presented above demonstrates that co-pollutant emissions vary widely, and that often they are concentrated in low-income and minority communities. The political and economic forces that account for this concentration may influence whatever changes in the distribution of co-pollutant burdens are brought about by climate policy, leading to differential impacts on local public health. There is a need to at least track the impacts of climate policy on co-pollutant emissions and assess whether there is real cause for concern. This is the approach taken by the California Air Resources Board (2011) in its "adaptive management plan" for the cap-and-trade policy under AB 32.

Tracking co-pollutant emissions to see if "hot spots" of increasing emissions develop or if there are sharply uneven reductions in co-pollutants is particularly important in the cases of the largest co-pollutant producers, the most co-pollutant producing industrial sectors, and the most impacted neighborhoods.

[48] For discussion, see also Kaswan (2011).

[49] This section draws on and extends the earlier analysis in Boyce (2009) and Pastor et al. (2010a, 2010c); see also Kaswan (2008, 2011).

Community Benefits Fund

Price-based climate policies will generate large monetary transfers from consumers (in proportion to their carbon footprints) to the recipients of what can be termed "carbon rent." In the case of cap-and-permit systems, this rent is called "allowance value." Potential recipients of carbon rent include governments, firms, and the public.[50]

Most climate-policy proposals envision that some fraction of the rent will be channeled into public investments in energy efficiency and renewable energy. Part of this money could be allocated to a community benefits fund to mitigate co-pollutant impacts and to protect public health in vulnerable communities. Such a policy was proposed in California Assembly Bill 1405, which was passed by the state legislature but vetoed by former governor Arnold Schwarzenegger in September 2010.[51] The bill would have mandated that at least 30 percent of the revenues generated under California's cap-and-trade program be deposited into a community benefits fund, which would channel these resources to vulnerable communities, defined as "those areas within each air basin with the highest 10 percent of air-pollution impacts, taking into account air-pollution exposures and socioeconomic indicators." The community benefits fund would have provided competitive grants for purposes such as reducing emissions of GHGs and co-pollutants, minimizing health impacts of global warming, and emergency preparedness for extreme weather events.

The proposal for a community benefits fund has been revived in California Senate Bill 535, with a smaller claim on the share of revenues. One challenge in implementing a community benefits fund policy would be how to determine which communities are eligible to receive benefits. A potential tool for this purpose is the Environmental Justice Screening Method developed by Sadd et al. (2011), which takes into account hazard proximity, local air quality, and measures of social vulnerability. A more recent iteration of this method incorporates climate vulnerability indicators as well, such as air-conditioning, tree canopy, transit access, flood risk, wildfire risk, and sea level.[52]

A Co-Pollutant Surcharge

A third policy option is to identify facilities, industrial sectors, and/or localities with high co-pollutant intensities and in these cases add a surcharge to the price of carbon permits. In effect, such a policy would try to price in the co-pollutant externality. By further increasing the price of fossil-fuel combustion, this would create an incentive for additional emissions reductions in these places.

[50] At the federal level, the Waxman-Markey bill that died in the U.S. Senate in 2010 included all three types of recipients (with an initial emphasis on firms, and dividends to the public to kick in only after 2025), while the CLEAR Act introduced by Senators Cantwell and Collins proposed to allocate 75 percent to dividends and 25 percent to the government for public investments in clean energy and transitional adjustment assistance.

[51] For discussion, see Prasad and Carmichael (2008) and Boyce (2009).

[52] Use of these criteria would strengthen the nexus between the source of revenues (permit auctions) and their use for a community benefits fund—an issue in California state law that has arisen in the context of AB 32 implementation (Horowitz et al. 2012).

Revenue from co-pollutant surcharges could be used for co-pollutant abatement, addressing cumulative impacts, or community benefits funds. If revenues from a location-based surcharge were recycled to the communities in which they are generated, this would be consistent with the principle that the "sink" functions of the air (i.e., the use of the air as a medium for disposal of wastes) belong to the people who breathe it.

Zonal Trading Systems

Zonal trading systems can be designed to ensure some minimum level of emissions reduction in designated zones, high-priority locations with the greatest potential benefits from emissions reduction. Polluters in high-priority (in this case, high co-benefit) zones are barred from "buying out" of emissions reduction by purchasing permits or offsets from other localities. The high-priority zones thus have their own zone-specific caps, with emissions reduction targets that either mirror or are more ambitious than those in the aggregate cap.

One precedent for zonal trading is California's Regional Clean Air Incentives Market (RECLAIM), which was launched in 1994 to reduce point-source emissions of NO_X and SO_2 in the Los Angeles basin. The South Coast Air Quality Management District established two zones under RECLAIM: zone 1, the coastal zone, where pollution is more severe and hence the benefits from pollution reduction are considered to be greater; and zone 2, the inland zone, where pollution is less severe. Facilities in zone 1 can buy permits only from other facilities in the same zone, whereas facilities in zone 2 can buy permits from either zone. One indicator of the impact of this zonal system is that average permit prices have been roughly eight times higher in zone 1 than in zone 2, creating a much stronger incentive to reduce emissions in zone 1 than would exist if polluters could purchase allowances from the less-polluted zone 2 (Gangadharan 2004). These restrictions may help to explain the finding by Fowlie et al. (2011) that the RECLAIM program did not exacerbate environmental injustice, as some had feared.[53]

Trading Ratios

Where pollution permits are tradable but damages per unit pollutant vary across pollution sources, the exchange rate at which permits are traded can be used as another policy instrument. If, for example, total (CO_2 plus co-pollutant) marginal damages per ton of CO_2 are twice as high in location A as in location B, owing to higher co-pollutant damages (and hence greater co-benefits from emissions reductions) in location A, the exchange rate ("trading ratio") would set one permit in location A to equal two permits in location B. This policy option has been widely discussed and in some cases implemented in water-pollution control (Farrow et al. 2005). In principle, location-based trading ratios can be applied to air pollution too (Tietenberg 1995; Muller and Mendelsohn 2009). Apart from the locational criteria, they also could be applied to trades between facilities distinguished on the basis of industrial sector or other facility-specific criteria related to differences in co-pollutant intensity.

[53] At least, this was their finding when using a multivariate regression model; their simple univariate models do show that RECLAIM led to more pollution reduction in whiter and wealthier neighborhoods.

Designate Priority Locations, Sectors or Facilities for Co-Pollutant Reductions

A final policy option is to designate priority locations, sectors, or facilities for co-pollutant reductions. This would respond to the heterogeneity in co-pollutant intensities that we have documented in this study. Recall that a small number of the facilities in our sample are responsible for a large share of the co-pollutant emissions—for example, the top 1 percent of the facilities is responsible for nearly 60 percent of the RSEI population-weighted health score and 35 percent of the population-weighted $PM_{2.5}$ measure. Policy makers could fairly easily accord special attention to these polluters. This could involve the use of conventional regulatory instruments, caps on emissions from the priority facilities, and/or restrictions on their ability to purchase permits from other polluters.

7. CONCLUSIONS

7.1 Summary

A strong case can be made for integrating co-pollutants into climate-policy design on both efficiency and equity grounds. From an efficiency standpoint, failure to account for variations in air-quality co-benefits across carbon-emission sources is tantamount to leaving health-care dollars lying on the floor. From an equity standpoint, co-pollutant burdens lie at the critical interface between climate policy and environmental justice. Numerous studies of the magnitude of air-quality co-benefits from climate policy have concluded that these 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 policy objective.

To make co-pollutants a tractable matter for policy design, we need to be able to measure co-pollutant intensity, here defined as the ratio of co-pollutant damages to CO_2 emissions. As this study has shown, this is by no means a straightforward matter. One conceptual issue is whether to use a broad measure that includes all emissions of pollutants from fossil-fueled activities or a narrow measure that is restricted to emissions from fossil-fuel combustion alone. In some cases, there is not much difference between the two, but in others the difference can be large. Climate policy can lead to co-pollutant emissions reductions not only from fossil-fuel combustion itself, but also from ancillary activities, and in principle the choice between narrow and broad measures should hinge on the likely mix between these responses. In practice, data availability may dictate the choice, as it did in our analysis of emissions from industrial facilities.

Given the variety of co-pollutants, a second issue is how to aggregate them to come up with an overall measure of co-pollutant intensity. Some co-pollutants are, pound for pound, far more hazardous than others, and toxicity weights can be used to account for this difference.

A third methodological issue is how to measure co-pollutant damages and, in particular, whether and how to account for differences in the number of people exposed to hazardous emissions and for differences in vulnerability across different population subgroups.

Our empirical analysis has demonstrated that these are not trivial problems. Rankings of co-pollutant intensity across industrial sectors and facilities can vary substantially depending on which co-pollutants are measured and, in the case of multiple co-pollutants, on how they are aggregated.

Using seven different measures of co-pollutant intensity—with the co-pollutant numerator defined variously as the mass of nitrogen oxides (NO_X); the mass of particulate matter ($PM_{2.5}$); the mass of sulfur dioxide (SO_2); the mass of air toxics; the toxicity-weighted mass of air toxics; the RSEI full-model score that also accounts for fate-and-transport of the air toxics and the number of people impacted by any given release; and a population-weighted $PM_{2.5}$ measure that we constructed—we demonstrated that co-pollutant intensity varies considerably across industrial sectors as well as across facilities. Any climate policy that is blind to these variations will lead to a pattern of carbon emissions reductions that is suboptimal from an efficiency standpoint.

We also demonstrated that there is wide spatial variation in co-pollutant emissions and intensity, and that people of color and low-income communities often bear a disproportionate share of the co-pollution burdens. Any climate policy that is blind to these variations is likely to lead to a pattern of carbon emissions reductions that is suboptimal from the standpoint of environmental justice too.

Finally, we discussed a number of ways in which co-pollutants can be integrated into climate policy, ranging from simply monitoring co-pollutant impacts—a policy option that we regard as a necessary minimum—to a variety of policies that would secure greater emissions reductions in high-priority facilities, sectors, and locations.

7.2 Recommendations to Improve Information for Policy Design

Our review of available evidence on co-pollutant intensity and its variations has highlighted a number of limitations in the information currently available for policy making. Measures that could improve the informational basis for policy design include the following:

- **Develop mechanisms for co-pollutant monitoring:** Climate-policy design should include provisions for monitoring policy impacts on emissions of co-pollutants, particularly at facilities and locations with relatively high emissions. This should include both source-based reporting (e.g., of co-pollutant emissions from industrial facilities) and ambient air-quality measurement. In the allocation of monitoring resources, special attention should be given to localities that are assigned priority by virtue of existing cumulative pollution impacts, vulnerability to further increases in pollution burdens, and sheer numbers of affected people. Annual reviews of monitoring results should be conducted, with a view to introducing remedial measures if the climate policy is found to widen the extent of disproportionate impacts of co-pollutants on minorities and low-income communities. Findings of absolute increases in co-pollutant burdens associated with climate-policy implementation should trigger immediate policy actions to ensure co-pollutant abatement in such locations.

- **Synchronize facility identification codes:** To facilitate research, effective monitoring, and policy making, we recommend that the USEPA synchronize the facility identification codes used in the agency's various databases. The data consolidation procedures that were required to conduct this study were not for the faint of heart—and although it appealed to the side of our personalities that loves going where no data analysts have gone before, we suggest that the next trip should have a better road map. In general, community-based accountability, particularly when entirely new regulatory systems are being devised, requires transparency and ease in understanding, using, and analyzing data. These are not features of the current data setup.

- **Develop aggregate measures of co-pollutant impacts:** To improve our picture of overall co-pollutant emissions and their human health impacts, data on emissions and ambient concentrations of criteria air pollutants (PM, NO_X, SO_2, CO, ozone, and lead) should be combined with comparable data on air toxics from the Toxics Release Inventory and geographic microdata from the Risk-Screening Environmental Indicators. This will require the application of fate-and-transport models to criteria air pollutants to estimate exposure levels, particularly in areas where actual monitoring is weak. Aggregate measures should be developed both with population weights (to assess total human health impacts) and without population weights (to assess statistical risk to individuals even in sparsely populated locations), since both are relevant to policy.

- **Develop environmental justice screening tools:** To improve the ability of policy makers to identify high-priority localities for policy attention, the Environmental Justice Screening Method recently advanced by researchers in California should be further developed by incorporating information on vulnerability to climate change itself. Comparable screening methods should be tested nationwide, with a view to making them a standard item in the environmental policy toolkit.

- **Extend data collection and analysis to nonindustrial pollution sources:** In many locations, mobile sources, such as automobiles and aircraft, and small-point sources, such as gas stations and dry cleaners, equal or surpass industrial facilities as sources of co-pollutant emissions. Spatial analysis of variations in co-pollutant burdens and co-pollutant intensity for these sources is needed to provide a basis for incorporating these co-pollutant emissions into climate policy. Several of the policy options sketched in the preceding chapter can be adapted to address emissions from these sources too.

7.3 Policy Recommendations

This study has demonstrated, we believe, that there is a compelling case for integrating air-quality co-benefits into the design of climate policy. To this end, we recommend the following measures:

- **Strengthen carbon emissions reduction targets:** A large body of evidence has established that the impacts of co-pollutants on public health are substantial. Air-quality co-benefits, therefore, should be included as standard practice in setting targets for carbon emissions reductions. The concept of the "social cost of carbon" should be expanded to include the social cost of co-pollutants. One result of incorporating this information into policy design will be more ambitious carbon emissions reduction targets.

- **Designate high-priority zones:** Climate-policy design should include the identification of high-priority zones where the co-benefits from reduced carbon emissions have the potential to be particularly large. In these zones, the policy design should ensure that emissions reductions will equal or exceed the average level of reductions achieved by the policy as a whole. Insofar as the climate policy relies on conventional "command-and-control" regulation, this can be accomplished by specifying more stringent standards for high-priority zones (akin to current policy differentiation under the Clean Air Act based on attainment area status). Insofar as the climate policy relies on price-based instruments, this can be accomplished by introducing specific caps for these zones that limit the number of permits to be auctioned or otherwise allocated to facilities in these zones and prevent the purchase of offsets or permits from outside the zone.[54]

- **Designate petroleum refineries and chemical manufacturers as high-priority sectors:** Among the three biggest carbon-emitting sectors analyzed in this study—power plants, petroleum refineries, and chemical manufacturers—the latter two were found to have higher co-pollutant intensities for toxicity-weighted and population-weighted emissions (see Table 9). They also have greater disproportionate impacts on minorities and the poor (see Tables 17 and 18). For these reasons, these should be designated as high-priority sectors for emissions reductions. In these sectors, climate policy should ensure that carbon emissions reductions will equal or exceed the average level of reductions achieved by the policy as a whole. Again, this can be achieved by conventional regulatory instruments or by sector-specific emission caps that specify the number of permits available to these sectors and prohibit purchases of permits from other sectors.

- **Designate high-priority facilities:** There is a high degree of disproportionality in co-pollutant emissions, as documented in Section 4.6 of this study: a small number of facilities account for a very large share of emissions. Facilities that rank in the top 5 percent in co-pollutant emissions (by one or more co-pollutant measures deemed to be most significant indicators of public-

[54] Note that the designation of high-priority zones is feasible only for price-based policies that include an emissions cap. This policy could not be applied to a simple carbon tax, though a higher tax rate (or surcharge) could be instituted in high-priority zones in an effort to obtain similar results.

health impacts) should be designated as high-priority facilities for carbon emissions reductions. In these facilities, climate policy should ensure that carbon emissions reductions will equal or exceed the average level of reductions achieved by the policy as a whole. Again, this can be achieved by differentiated standards or by facility-specific caps.

- **Allocate a share of carbon revenues to community benefit funds:** To ensure that disadvantaged communities that bear disproportionate pollution burdens obtain a fair share of the benefits from public investments in the clean energy transition, a fraction of the carbon rent generated by the use of price-based instruments in climate policy should be directed to community benefit funds to support environmental and public-health improvements in these localities. The screening methods used to identify high-priority zones could also be applied to identify localities eligible for the community benefits fund.

7.4 Concluding Remarks

In recent debates on climate policy, the air-quality co-benefits that could be obtained by curtailing the burning of fossil fuels have received too little attention. Yet a substantial body of peer-reviewed literature in the environmental and health sciences has concluded that the potential benefits of reduced emissions of hazardous co-pollutants could be quite large—certainly large enough to be of policy relevance.

What accounts, then, for the past neglect of co-pollutants in climate-policy debates? Part of the answer may lie in the fact that policy makers and climate-policy advocates feel overburdened by other issues: debates over conventional regulations versus price-based instruments, the social cost of carbon, and even the scientific basis for climate change itself. In our view, part of the answer also lies in the political and economic marginalization of the constituencies that are most heavily burdened by co-pollutants. This marginalization can be redressed only by ensuring that they have a place at the climate-policy table.

In this study, we have shown that it is feasible to bring empirical evidence to bear on the issue of whether and how to incorporate co-pollutants into climate-policy design. Focusing on emissions from industrial facilities in the United States, we have shown that it is possible to compute measures of co-pollutant intensity—co-pollutant damages per ton of carbon dioxide emissions—and that co-pollutant intensity varies widely across facilities and industrial sectors. We have also shown that minority and low-income communities bear disproportionate burdens from co-pollutant emissions in those industrial sectors that stand to be most strongly impacted by climate policy.

The conclusion, we believe, is clear: for reasons of both efficiency and equity, co-pollutants and air-quality co-benefits warrant inclusion in climate-policy design. Co-benefits should be integrated into the design of policies built on conventional regulatory instruments, such as those currently being formulated by the USEPA under the Clean Air Act. Co-benefits likewise should be integrated into price-based policies, such as the cap-and-trade system now being adopted under California's Global Warming Solutions Act. Any 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 benefits of reducing emissions of co-pollutants.

APPENDIX: INTERNATIONAL VARIATIONS IN CO-POLLUTANT INTENSITY

The focus of this study has been how to integrate air-quality co-benefits into climate policy in the United States. In this appendix we briefly consider evidence on how co-pollutant intensity varies internationally and discuss some potential policy implications of this variation.

All else being equal, co-pollutant intensity is likely to be higher where air-pollution regulations are weaker. This suggests that the co-benefits per unit of carbon emissions reductions may be highest in developing countries. In countries where carbon emissions remain very low, this may not be of much importance. But in newly industrializing countries, where fossil-fuel consumption is growing rapidly and where this growth is outpacing advances in air-quality management, the policy salience of co-pollutants may be particularly great.

Data on variations in co-pollutant intensity across countries are few and far between. The only source (or at least, the only one we know of) that provides fairly comprehensive international data, covering developing countries as well as industrial countries, on both CO_2 emissions and a measure of co-pollutant measure emissions is the World Bank's World Development Indicators (WDI).[55] The WDI data include a number of variables generated by the World Bank in order to develop "adjusted savings" measures for national income accounting that take environmental degradation (dissaving) into account. These adjustments include one for "particulate emissions damage" and one for "CO_2 damage." By computing the ratio between the two, we can get a broad measure of co-pollutant intensity variations for PM across nations.

The World Bank's particulate emissions damage estimate is calculated as "the willingness to pay to avoid mortality attributable to particulate emissions." Although details are not provided—the technical notes to the WDI reference an unpublished 2006 paper—willingness to pay varies with ability to pay, and hence the valuation of human life used in these calculations may have been lower in low-income countries than in high-income countries (much lower, if it was a linear function of per capita income). If so, an alternative measure of co-pollutant intensity in which the numerator was mortality attributable to particulate emissions (rather than a dollar valuation of that mortality) could substantially increase the co-pollutant intensity values in low-income countries relative to those in high-income countries. Another limitation of the World Bank data is that the measure of particulate emissions probably includes emissions from other sources in addition to fossil-fuel combustion (the technical notes do not discuss this question), in which case we can derive only a broad measure of co-pollutant intensity (including all particulate emissions in the numerator) as opposed to a narrow one (restricted to emissions from fossil-fuel combustion).

[55] Available online at http://databank.worldbank.org/ddp/home.do.

The World Bank's CO_2 damage estimate is based on an across-the-board valuation of $20 per ton of carbon. The same valuation is applied to all countries, so the damage measure is a straightforward multiple of carbon emissions; the choice of a different carbon price thus would not affect the pattern of relative co-pollutant intensities across countries.[56]

As a final caveat, we note that PM is not the only co-pollutant, and it is possible that intensity measures for other co-pollutants, such as SO_2 and air toxics, would show different patterns at the international level, as we find at the facility level within the United States. Insofar as differences in regulatory regimes explain the international variations, however, we might expect similar patterns for other co-pollutants. Moreover, PM itself is an important co-pollutant in its impacts on public health.

With these caveats in mind, we turn to the data. Table 21 presents co-pollutant intensity measures based on the World Bank data for the G-20 countries (there are actually 19 countries, since the 20th member of the G-20 is the European Union). Together these countries account for 76 percent of world income and 77 percent of total world carbon emissions from fossil-fuel combustion.[57] The data indicate a wide range of international variation in co-pollutant intensity (even if we omit the near-zero value for the United Kingdom as a likely data error). Among the G-20 countries with per capita incomes over $40,000, the United States has the highest co-pollutant intensity.

Although the correlation is by no means perfect, there is an apparent inverse relation between co-pollutant intensity and per capita income (Pearson's r = –0.40, excluding the U.K.). Fitting a quadratic curve to the data (see Figure 9), we find an inverted-U relationship akin the so-called environmental Kuznets curve, suggesting that co-pollutant intensity tends first to rise with per capita income and then to decline after reaching a turning point at roughly $17,000 (in 2008 dollars). This may be a consequence of the mortality valuation methodology used in the World Bank data, however; recall that a co-pollutant intensity measure based simply on mortality would show relatively higher values at the lower end of the income spectrum.

This exploratory exercise lends empirical support to the thesis that air-quality co-benefits could provide an important inducement for governments of major developing countries to participate in international agreements to curb carbon emissions.[58] To be sure, developing countries could reduce emissions of PM and other co-pollutants by adopting regulations that lower co-pollutant intensity rather than by reducing fossil-fuel combustion as part of an international climate agreement, and the marginal cost of PM reductions by the first route may be lower than by the second. For developing-

[56] The World Bank's CO2 damage data for the 19 countries are perfectly correlated with the U.S. Energy Information Administration's data on carbon dioxide emissions from the consumption of energy in 2008 (r = 0.999).

[57] Percentage of world income calculated from 2008 WDI data on gross national income; percentage of 2008 carbon emissions calculated from U.S. Energy Information Administration, "Total Carbon Dioxide Emissions from the Consumption of Energy," http://www.eia.gov/cfapps/ipdbproject/iedindex3.cfm?tid=90&pid=44&aid=8&cid=regions&syid=2005&eyid=2009&unit=MMTCD.

[58] Pittel and Rubbelke (2008) advance this thesis in a game-theoretic model in which "private" co-benefits to individual countries interact with the "public" benefits of climate-change mitigation. In their survey of co-benefit estimates, Nemet et al. (2010) report generally higher values in studies of developing countries and conclude that "the inclusion of co-benefits provides stronger incentives for cooperation from develop-ing countries than do climate benefits alone." See also Cienfuentes et al. (2001a, 2001b) and Halsnæs and Olhoff (2005).

country policy makers, however, the second route to air-quality improvement has the added attractions that come with participation in an international agreement to mitigate climate change. In this respect, co-benefits are a two-way street: just as air-quality improvements are a co-benefit of climate policy, climate mitigation can be a co-benefit of policies to curb air pollution.

International variations in co-pollutant intensity also have implications for "flexibility mechanisms" (sometimes shortened to "flexmex") in international climate agreements, such as Joint Implementation and the Clean Development Mechanism in the Kyoto accord. These create ways for countries that have accepted binding limits on their carbon emissions to finance equivalent emissions reductions (or in some cases carbon sequestration) elsewhere, an option that may be seen as easier or cheaper than achieving the same emissions reductions at home.

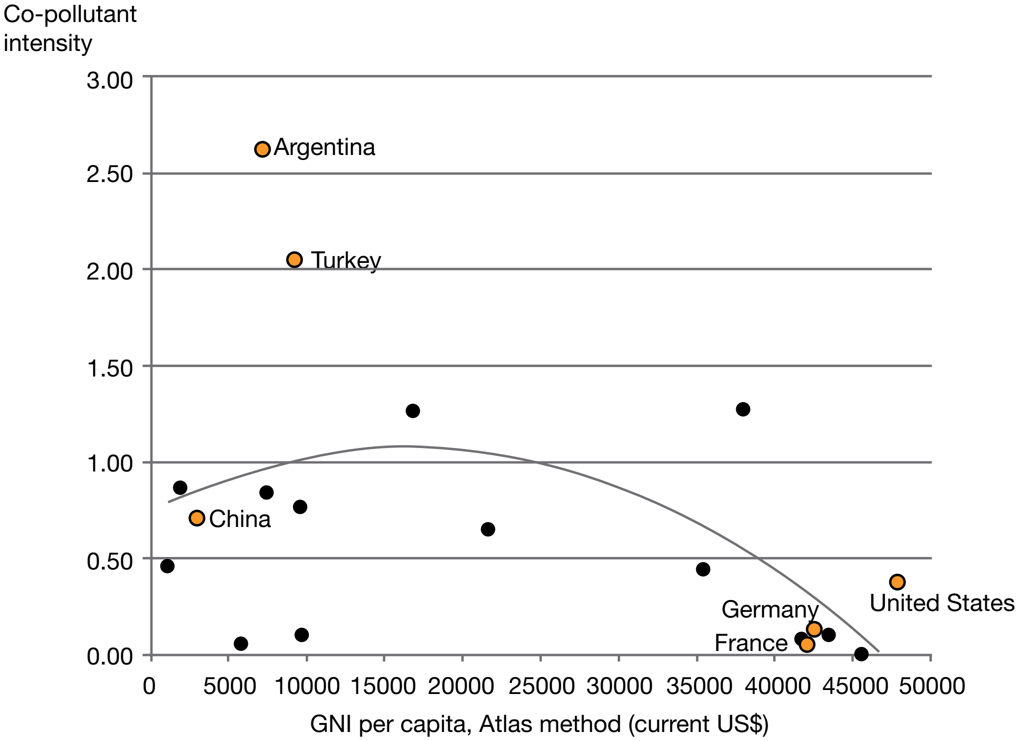
Relocating carbon emissions reductions means relocating their air-quality co-benefits. From the standpoint of the "buyer" country, this reduces the appeal of using such mechanisms instead of reducing emissions at home. From the standpoint of the "seller" country, where the actual emissions reductions take place, it enhances their appeal. Insofar as the buyers are high-income countries and the sellers are low-income countries, as in the Clean Development Mechanism, the air-quality results of such transactions may be distributionally progressive in that they relocate co-benefits to countries with lower per capita incomes. The merits of international carbon trading can be questioned on the grounds of additionality, verifiability, and distributional impacts within the participating countries, but the higher air-quality co-benefits that can be secured in developing countries would weigh on the positive side of the scales.

Table 22: Co-Pollutant Intensity in the G-20 Countries, 2008 (Ratio of particulate emissions damages to CO_2 emissions)

Country	GNI Per Capita[1]	Co-Pollutant Intensity
India	1,080	0.468
Indonesia	1,950	0.865
China	3,040	0.708
South Africa	5,860	0.061
Argentina	7,190	2.624
Brazil	7,480	0.843
Turkey	9,260	2.047
Mexico	9,640	0.764
Russian Federation	9,710	0.108
Saudi Arabia	16,790	1.261
Korea, Rep.	21,580	0.653
Italy	35,360	0.442
Japan	38,000	1.266
Australia	41,760	0.086
France	42,060	0.056
Germany	42,520	0.124
Canada	43,470	0.099
United Kingdom	45,610	0.002
United States	47,840	0.374

[1] Atlas method (current U.S. $)

Figure 9: Co-Pollutant Intensity in the G-20 Countries (Ratio of particulate emissions damages to CO_2 emissions, 2008)



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REFERENCES

Ackerman, Frank and Lisa Heinzerling (2005) *Priceless: On Knowing the Price of Everything and the Value of Nothing.* New York: New Press.

Adelman, David E. (2012) "The Collective Origins of Toxic Air Pollution: Implications for Greenhouse Gas Trading and Toxic Hotspots." Austin, TX: University of Texas School of Law, working paper. Online at http://papers.ssrn.com/sol3/papers.cfm?abstract_id=2015375.

Alexeeff, George, John Faust, Laura Meehan August, Carmen Milanes, Karen Randles, and Lauren Zeise (2010) *Cumulative Impacts: Building a Scientific Foundation.* Office of Environmental Health Hazard Assessment, State of California. Online at http://oehha.ca.gov/ej/pdf/CIReport123110.pdf.

Ackerman, Frank and Elizabeth A. Stanton (2010) "The Social Cost of Carbon." Report for the E3 Network: Economics for Equity and the Environment, April. Online at www.e3network.org/papers/SocialCostOfCarbon_SEI_20100401.pdf.

Ash, Michael *et al.* (2009) *Justice in the Air: Tracking Toxic Pollution from America's Industries to our States, Cites, and Neighborhoods.* Los Angeles: Program for Environmental and Regional Equity and Amherst, MA: Political Economy Research Institute. Online at http://www.peri.umass.edu/fileadmin/pdf/dpe/ctip/justice_in_the_air.pdf

Ash, Michael and James K. Boyce (2011) "Measuring Corporate Environmental Justice Performance." *Corporate Social Responsibility and Environmental Management* 18(2): 61-79.

Barringer, F. (2012) "For New Generation of Power Plants, a New Emission Rule From the E.P.A." *The New York Times*, 27 March.

Bell, M.L. *et al.* (2008) "Ancillary Human Health Benefits of Improved Air Quality Resulting from Climate Change Mitigation." *Environmental Health* 7:41.

Berk, M.M. *et al.* (2006) "Sustainable energy: Trade-offs and synergies between energy security, competitiveness, and environment." Bilthoven: Netherlands Environmental Assessment Agency (MNP).

Berry, Lisa M. (2008) "Inequality in the Creation of Environmental Harm: Looking for Answers from Within," in Robert C. Wilkinson and William R. Freudenburg, eds., *Equity and the Environment. Research in Social Problems and Public Policy* 15: 239-265.

Bollen, J., B. van der Zwaan, C. Brink C and H. Eerens (2009) "Local air pollution and global climate change: A combined cost-benefit analysis." *Resource and Energy Economics* 31(3): 161-181.

Boyce, James K. (2007) "Inequality and Environmental Protection," in Jean-Marie Baland, Pranab Bardhan, and Samuel Bowles, eds., *Inequality, Cooperation, and Environmental Sustainability*. Princeton: Princeton University Press, pp. 314-348.

Boyce, James K. (2009) "Investment in Disadvantaged Communities." Memorandum for the Economic and Allocation Advisory Committee, California Air Resources Board and California Environmental Protection Agency, December 30. Online at http://www.climatechange.ca.gov/eaac/documents/member_materials/Boyce_memo_on_investment_indisadvantaged_communities-revised-30_Dec_2009.pdf.

Boyce, James K. and Matthew E. Riddle (2007) "Cap and Dividend: How to Curb Global Warming While Protecting the Incomes of American Families." Amherst, MA: Political Economy Research Institute, Working Paper No. 150, November.

Burtraw, Dallas (1996) "The SO_2 Emissions Trading Program: Cost Savings without Allowance Trades." *Contemporary Economic Policy* 14: 79-94.

Burtraw, Dallas et al. (2003) "Ancillary Benefits of Reduced Air Pollution in the U.S. from Moderate Greenhouse Gas Mitigation Policies in the Electricity Sector." *Journal of Environmental Economics and Management* 45: 650-673.

California Air Resource Board (2011) "Adaptive Management Plan for the Cap-and-Trade Regulation." 10 October. Online at http://www.arb.ca.gov/cc/capandtrade/adaptive_management/plan.pdf.

Carlson, Curtis, Dallas Burtraw, Maureen Cropper and Karen L. Palmer (2000) "Sulfur Dioxide Control by Electric Utilities: What are the Gains from Trade?" *Journal of Political Economy* 108(6): 1292-1326.

Cienfuentes, L. V.H. Borja-Aburto, N. Gouveia, G. Thurston and D.L. Davis (2001a) "Assessing the Health Benefits of Urban Air Pollution Reductions Associated with Climate Change Mitigation (2000-2020): Santiago, Sao Paulo, Mexico City, and New York City." *Environmental Health Perspectives* 109(suppl 3): 419-425.

Cienfuentes, L., V.H. Borja-Aburto, N. Gouveia, G. Thurston and D.L. Davis (2001b) "Hidden Health Benefits of Greenhouse Gas Mitigation." *Science* 293(5533): 1257-9.

Credit Suisse (2010) "Growth from Subtraction: Impact of EPA Rules on Power Markets." 23 September. Online at http://op.bna.com/env.nsf/id/jstn-8actja/$File/suisse.pdf.

Dorman, Peter (1996) *Markets and Mortality: Economics, dangerous work, and the value of human life.* Cambridge: Cambridge University Press.

Economic and Allocation Advisory Committee, California Air Resources Board and California Environmental Protection Agency (2010) *Allocating Emissions Allowances under a California Cap-and-Trade Program: Recommendations to the California Air Resources Board and the California Environmental Protection Agency*. March.

Farber, Daniel A. (2012) "Pollution Markets and Social Equity: Analyzing the Fairness of Cap and Trade." *Ecology Law Quarterly* 39: 1-56.

Farrow, R. Scott, Martin T. Schultz, Pinar Celikkol and George L. Van Houtven (2005) "Pollution Trading in Water Quality Limited Areas: Use of Benefits Assessment and Cost-Effective Trading Ratios." *Land Economics* 81(2): 191-205.

Fowlie, Meredith and Nicholas Muller (2011) "Designing Markets for Pollution When Damages Vary Across Sources: What are the Gains from Differentiation?" Draft, 9 December.

Fowlie, Meredith, Stephen P. Holland, and Erin T. Mansur (2011) "What Do Emissions Markets Deliver and to Whom? Evidence from Southern California's NOx Trading Program." Draft March 28, forthcoming in *American Economic Review*.

Gangadharan, Lata (2004) "Analysis of prices in tradable emission markets: An empirical study of the Regional Clean Air Incentives Market in Los Angeles." *Applied Economics* 36: 1569-1582.

Groosman, Britt, Nicholas Z. Muller and Erin O'Neill-Toy (2009) "The ancillary benefits from climate policy in the United States." Middlebury College, Department of Economics, Discussion Paper No. 0920. November. Online at sandcat.middlebury.edu/econ/repec/mdl/ancoec/0920.pdf

Grubb, Michael (2011) "International Climate Finance from Border Carbon Cost Levelling." *Climate Policy* 11(3): 1050-1057.

Hall, Jane V., Victor Brajer and Frederick W. Lurmann (2008) *The Benefits of Meeting Federal Clean Air Standards in the South Coast and San Joaquin Valley Air Basins.* Fullerton, CA: Institute for Economic and Environmental Studies, California State University, Fullerton.

Halsnæs, K. and A. Olhoff (2005) "International markets for greenhouse gas emission reduction policies - possibilities for integrating developing countries." *Energy Policy* 33: 2313-2325.

Holland, Stephen P. (2010) "Spillovers from Climate Policy." NBER Working Paper No.16158. July. Online at http://www.nber.org/papers/w16158. Forthcoming in Don Fullerton and Catherine Wolfram, eds., *The Design and Implementation of U.S. Climate Policy*. Chicago: University of Chicago Press.

Horowitz, Cara, M. Rhead Enion, Sean B. Hecht and Ann Carlson (2012) "Spending California's Cap-and-Trade Revenue: Understanding the *Sinclair Paint* Spectrum." Los Angeles: UCLA School of Law, Emmett Center on Climate Change and the Environment, March.

Interagency Working Group on Social Cost of Carbon, U.S. Government (2010) "Technical Support Document: Social Cost of Carbon for Regulatory Impact Analysis Under Executive Order 12866." Washington, DC, February. Online at www.epa.gov/oms/climate/regulations/scc-tsd.pdf.

International Energy Agency (IEA) (2011) CO_2 *Emissions from Fuel Combustion: Highlights. 2011 Edition*. Paris: OECD/IEA.

Kaswan, Alice (2008) "Environmental Justice and Domestic Climate Change Policy." *Environmental Law Reporter* 38: 10287-10315.

Kaswan, Alice (2011) "Reconciling Justice and Efficiency: Integrating Environmental Justice into Domestic Cap-and-Trade Programs for Controlling Greenhouse Gases," in Denis G. Arnold, ed., *Ethics and Global Climate Change*. Cambridge and New York: Cambridge University Press, pp. 232-254.

Kaswan, Alice (2012) "Climate Change, the Clean Air Act, and Industrial Pollution." *UCLA Journal of Environmental Law & Policy* 30: 51-120.

Lerza, Catherine (2012) "California's New Environmental Movement." *Shelterforce*, spring. Available at http://www.shelterforce.org/article/2766/californias_new_environmental_movement/.

Mazzi, E.A. and H. Dowlatabadi (2007) "Air quality impacts of climate mitigation: UK policy and passenger vehicle choice." *Environmental Science & Technology* 41(2): 387-392.

McCarthy, James E., Larry Parker and Robert Meltz (2010) "After the CAIR Decision: Multipollutant Approaches to Controlling Powerplant Emissions." Washington, DC: Congressional Research Service, 4 March.

Mohai, Paul (2008) 'Equity and the Environmental Justice Debate,' in Robert C. Wilkinson and William R. Freudenberg, eds., *Equity and the Environment*, Research in Social Problems and Public Policy, Vol. 15. Amsterdam: Elsevier, pp. 21-50.

Mohai, Paul and Robin Saha (2006). "Reassessing Racial and Socioeconomic Disparities in Environmental Justice Research." *Demography* 43(2): 383-399.

Morello-Frosch, R., M. Zuk, M. Jerrett, B. Shamasunder and A.D. Kyle (2011) "Understanding the Cumulative Impacts of Inequalities in Environmental Health: Implications for Policy." *Health Affairs* 30(5): 879-887.

Muller, Nicholas Z. (2012) "Optimal Climate Policy with Air Pollution Co-Benefits." *Resource and Energy Economics*, forthcoming.

Muller, Nicholas Z. and Robert Mendelsohn (2009) "Efficient Pollution Regulation: Getting the Prices Right." *American Economic Review* 99(5): 1714-39.

Muller, Nicholas Z., Robert Mendelsohn, and William Nordhaus (2011) "Environmental Accounting for Pollution in the United States Economy." *American Economic Review* 101(5): 1649–75.

National Academy of Sciences (2004) *Air Quality Management in the United States*. Washington, DC: National Academies Press.

National Academy of Sciences (2009) *Hidden Costs of Energy: Unpriced Consequences of Energy Production and Use*. Washington, DC: National Academies Press.

Nemet, G.F., T. Holloway and P. Meier (2010) "Implications of Incorporating Air-quality Co-benefits into Climate Change Policymaking." *Environmental Research Letters* 5(1): 014007.

Pastor, Manuel (2003) "Building Social Capital to Protect Natural Capital: The Quest for Environmental Justice," in James K. Boyce and Barry G. Shelley, eds., *Natural Assets: Democratizing Environmental Ownership*. Washington, DC: Island Press, pp. 77-97.

Pastor, Manuel, Rachel Morello-Frosch, and Jim Sadd (2005) "The Air is Always Cleaner on the Other Side: Race, Space, and Air Toxics Exposures in California." *Journal of Urban Affairs* 27(2): 127-148

Pastor, Manuel, Rachel Morello-Frosch, James Sadd and Justin Scoggins (2010a) *Minding the Climate Gap: What's at Stake if California's Climate Law Isn't Done Right and Right Away*. Los Angeles, CA: USC Program for Environmental & Regional Equity.

Pastor, Manuel, Rachel Morello-Frosch, James Sadd, and Justin Scoggins (2010b) *Minding the Climate Gap, Executive Summary*. Los Angeles, CA: USC Program for Environmental & Regional Equity.

Pastor, Manuel, Rachel Morello-Frosch, James Sadd and Justin Scoggins (2010c) "Risky Business: Cap-and-Trade, Public Health, and Social Equity." Los Angeles: USC Program on Environmental and Regional Equity.

Pittel, Karen and Dirk T.G. Rubbelke (2008) "Climate Change and Ancillary Benefits: A survey and integration into the modeling of international negotiations on climate change." *Ecological Economics* 68: 210-220.

Prasad, Shankar and Tim Carmichael (2008) "AB 32 Community Benefit Fund to Reduce Cumulative and Disproportionate Impacts." Sacramento, CA: Coalition for Clean Air. March.

Revesz, Richard and Michael Livermore (2008) *Retaking Rationality: How Cost-Benefit Analysis Can Better Protect the Environment and Our Health*. New York: Oxford University Press.

Ringquist, Evan J. (2005). "Assessing evidence of environmental inequities: A meta-analysis." *Journal of Policy Analysis and Management* 24(2): 223-247.

Sadd, James L., Manuel Pastor , Rachel Morello-Frosch, Justin Scoggins and Bill Jesdale (2011) "Playing It Safe: Assessing Cumulative Impact and Social Vulnerability through an Environmental Justice Screening Method in the South Coast Air Basin, California." *International Journal of Environmental Research and Public Health* 8: 1441-1459.

Schatzki, Todd and Robert N. Stavins (2009) "Addressing Environmental Justice Concerns in the Design of California's Climate Policy." Online at http://www.climatechange.ca.gov/eaac/comments/2009-11-03_Schatzki_and_Stavins_attachment.pdf.

Shonkoff, Seth B., Rachel Morello-Frosch, et al. (2011) "The Climate Gap: Environmental Health and Equity Implications of Climate Change and Mitigation Policies in California—A review of the literature." *Climatic Change* 109(Suppl 1): S485-S503.

Szasz, Andrew and Michael Meuser (1997) "Environmental Inequalities: Literature Review and Proposals for New Directions in Research and Theory." *Current Sociology* 45(3), 99-120.

Tietenberg, Tom (1995) "Tradeable permits for pollution control when emission location matters: What have we learned?" *Environmental and Resource Economics* 5: 95-113.

Tinbergen, J. (1952) *On the Theory of Economic Policy*. Amsterdam: North-Holland.

United States Environmental Protection Agency (USEPA) (2011a) *Inventory of U.S. Greenhouse Gas Emissions and Sinks:* 1990-2009. EPA 430-R-11-005. April 15.

USEPA (2011b) *1970-2011 Average annual emissions, all criteria pollutants in MS Excel* – October. http://www.epa.gov/ttn/chief/trends/index.html. Accessed 5 July 2012.

Viscusi, W.K. and J.E. Aldy (2003) "The Value of a Statistical Life: A Critical Review of Market Estimates Throughout the World." *Journal of Risk and Uncertainty* 27: 5-76.

Walsh, Michael P. (2008) "Ancillary Benefits for Climate Change Mitigation and Air Pollution Control in the World's Motor Vehicle Fleet." *Annual Review of Public Health* 29: 1-9.

Wigley, Tom M.L. (2011) "Coal to Gas: The Influence of Methane Leakage." *Climatic Change* 108(3): 601-608.