

The Collective Origins of Toxic Air Pollution: Implications for Greenhouse Gas Trading and Toxic Hotspots

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This Article presents the first synthesis of geospatial data on toxic air pollution in the United States. Contrary to conventional views, the data show that vehicles and small stationary sources emit a majority of the air toxics nationally. Industrial sources, by contrast, rarely account for more than ten percent of cumulative cancer risks from all outdoor sources of air toxics. This pattern spans multiple spatial scales, ranging from census tracts to the nation as a whole. However, it is most pronounced in metropolitan areas, which have the lowest air quality and are home to eighty percent of the U.S. population.

The secondary status of industrial facilities as sources of air toxics has important implications for the current debate over cap-and-trade regulation—the policy instrument of choice for controlling greenhouse gas (GHG) emissions responsible for climate change. Environmental justice advocates have opposed GHG trading in significant part because it could exacerbate inequitable exposures to toxic co-pollutants, not GHGs themselves, in minority and low-income communities.

The likelihood of such disparities occurring has remained an open empirical question. The geospatial data reveal that, apart from a few readily identifiable census tracts, the potential for GHG trading to cause toxic hotspots is extremely low. Moreover, for the few jurisdictions in which disparities cannot be ruled out, targeted policies exist to prevent them without compromising market efficiency.

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INTRODUCTION

The potential for cap-and-trade programs¹ to cause localized hotspots of toxic air pollution has long been a flashpoint for opposition to this form of market-based environmental regulation.² The Regional Clean Air Incentives Market, a controversial pollution trading system in southern California, was almost upended due to concerns that environmental-justice advocates raised about toxic hotspots around major industrial facilities.³ Similarly, the George W. Bush Administration's plan to establish a cap-and-trade regime for regulating mercury emissions from electric utilities was opposed successfully on the grounds that it would allow

1. A pollution-trading regime establishes a numerical cap on aggregate emissions, allocates emissions credits among sources (for example, an auction) and allows credits to be traded in a market—hence the term “cap and trade.” Jonathan Remy Nash & Richard L. Revesz, *Markets and Geography: Designing Marketable Permit Schemes to Control Local and Regional Pollutants*, 28 *ECOLOGY L.Q.* 569, 575–76 (2001).

2. See, e.g., *id.* at 580 (noting that academics have “focused almost exclusively on the possibility that emissions trading will [cause hotspots of air pollutants]”); Lily N. Chinn, *Can the Market Be Fair and Efficient? An Environmental Justice Critique of Emissions Trading*, 26 *ECOLOGY L.Q.* 80, 108–09 (1999); Stephen M. Johnson, *Economics v. Equity: Do Market-Based Environmental Reforms Exacerbate Environmental Injustice?*, 56 *WASH. & LEE L. REV.* 111, 111–12 (1999); Alice Kaswan, *Environmental Justice and Domestic Climate Change Policy*, 38 *ENVTL. L. REP.* 10287, 10287–88 (2008).

3. See Richard Toshiyuki Drury, Michael E. Belliveau, J. Scott Kuhn & Shipra Bansal, *Pollution Trading and Environmental Injustice: Los Angeles' Failed Experiment in Air Quality Policy*, 9 *DUKE ENVTL. L. & POL'Y F.* 231, 268–69 (1999) (describing failures of the RECLAIM program and the deficiencies of market-based regulations generally); Lesley K. McAllister, *Beyond Playing “Banker”: The Role of the Regulatory Agency in Emissions Trading*, 59 *ADMIN. L. REV.* 269, 287–312 (2007).

mercury emissions to concentrate around power plants.⁴ Even the venerable SO₂ trading system for U.S. power plants has not been immune to fears that it could cause pollution hotspots.⁵

Despite this record of controversy, cap-and-trade regulation is the policy instrument of choice for controlling greenhouse gas (GHG) emissions responsible for climate change.⁶ Globally, the most prominent regulatory efforts have embraced it: the European Union's Emissions Trading System, the Regional Greenhouse Gas Initiative in the northeastern United States, and, most recently, the landmark Global Warming Solutions Act in California.⁷ Policymakers are drawn to the projected efficiencies of cap-and-trade regimes and to their political virtues—they avoid taxes and unpopular command-and-control regulations.⁸ Further, because GHGs are global pollutants that do not have direct localized impacts, regulatory experts have considered the risks of hotspots to be essentially zero.⁹

This benign view of GHG-trading regimes is being challenged by environmental-justice groups in California and elsewhere.¹⁰ In a recent legal action,

4. See Catherine A. O'Neill, *Mercury, Risk, and Justice*, 34 ENVTL. L. REP. 11070, 11070–71 (2004) (describing the controversy surrounding the proposed cap-and-trade regime for mercury emissions from power plants).

5. Jason Corburn, *Emissions Trading and Environmental Justice: Distributive Fairness and the USA's Acid Rain Programme*, 28 ENVTL. CONSERVATION 323, 323–24 (2001); Evan J. Ringquist, *Trading Equity for Efficiency in Environmental Protection?: Environmental Justice Effects from the SO₂ Allowance Trading Program*, 92 SOC. SCI. Q. 297, 300–01 (2011).

6. See Jonathan Baert Wiener, *Global Environmental Regulation: Instrument Choice in Legal Context*, 108 YALE L.J. 677, 712–13 (1999) (arguing that global environmental regulation, particularly for mitigating climate change, should favor pollution trading regimes); A. DENNY ELLERMAN, PAUL L. JOSKOW & DAVID HARRISON, JR., EMISSIONS TRADING IN THE U.S.: EXPERIENCE, LESSONS, AND CONSIDERATIONS FOR GREENHOUSE GASES 1–2, 34 (2003), available at <http://www.c2es.org/publications/emissions-trading-us-experience-lessons-and-considerations-greenhouse-gases>.

7. Kaswan, *supra* note 2, at 10291–92; Slobodan Perdan & Adisa Azapagic, *Carbon Trading: Current Schemes and Future Developments*, 39 ENERGY POL'Y 6040, 6040–41 (2011). There was the failed attempt by congressional Democrats to pass cap-and-trade regulations for GHGs in 2009. See William W. Buzbee, *Clean Air Act Dynamism and Disappointments: Lessons for Climate Legislation to Prompt Innovation and Discourage Inertia*, 32 WASH. U. J.L. & POL'Y 33, 62–64 (2010).

8. See ELLERMAN ET AL., *supra* note 6, at v, vi, 29, 34–35 (describing the efficiency of pollution trading systems); Byron Swift, *U.S. Emissions Trading: Myths, Realities, and Opportunities*, NAT. RESOURCES & ENV'T, Summer 2005, at 3, 3–4; Alan Murray, *Why Key Executives Are Warming To Legislation on Climate Change*, WALL ST. J., Feb. 7, 2007, at A10 (describing the support of the business community for cap-and-trade regulation of GHGs).

9. See ELLERMAN ET AL., *supra* note 6, at 40–41; MKT. ADVISORY COMM. TO THE CAL. AIR RES. BD., RECOMMENDATIONS FOR DESIGNING A GREENHOUSE GAS CAP-AND-TRADE SYSTEM FOR CALIFORNIA 12 (2007), available at <http://www.energy.ca.gov/2007publications/ARB-1000-2007-007/ARB-1000-2007-007.PDF> (stating that a trading program “[i]n no case” would cause an increase in emissions of toxic co-pollutants).

10. See Felicity Barringer, *California Judge Calls Time Out for Climate Change Law*,

a coalition of groups successfully delayed the decision of the California Air Resources Board (CARB) to establish a cap-and-trade system for regulating GHGs.¹¹ One of their primary concerns was the potential for the trading regime to exacerbate inequitable exposures to toxic co-pollutants, not GHGs themselves, in minority and low-income communities.¹² Simplifying their argument for the moment, they claimed that the flexibility inherent in a trading regime—the freedom to surpass regulatory limits by buying tradable credits—could concentrate emissions of toxic co-pollutants around industrial facilities located in minority communities.

The validity of this critique follows from the focus of market-based regimes on aggregate efficiency. Nothing in a pollution-trading regime forecloses inequities in exposures to toxic co-pollutants, but, as environmental-justice advocates acknowledge, the likelihood of this occurring is contingent on several factual predicates. The factors include the relative cost of reducing GHG emissions at industrial facilities, the geographic distribution of industrial facilities with significant GHG emissions, correlations between GHG and toxic emissions, and the relative contribution of major GHG sources to toxic emissions from all sources collectively.¹³

This Article will examine whether these factual predicates are likely to exist. I will focus particular attention on the geographic distribution of industrial facilities and their relative contributions to toxic emissions. The Environmental Protection Agency (EPA) maintains, and has been enhancing, several databases on toxic emissions and cumulative cancer risks that provide an unprecedented level of information (down to the census tract) and that have not received the attention they deserve.¹⁴ Although subject to important limitations, the databases provide new insights into the prevalence of hotspots nationally, the geographic concentration of industrial sources of toxic emissions regionally, and the impacts of diffuse point and mobile sources on air quality throughout the country.

N.Y. TIMES GREEN BLOG (Mar. 22, 2011), <http://green.blogs.nytimes.com/2011/03/22/california-judge-calls-time-out-for-climate-change-law/>; Ngoc Nguyen, *Climate-Change Law: Why CA Environmentalists Are Fighting Each Other*, NEW AM. MEDIA (Mar. 14, 2011), <http://newamericamedia.org/2011/03/cap-and-trade-story-here.php>. In a subsequent ruling, the California court allowed implementation of the state's GHG-trading system to go forward. Bob Egelko, *Cap and Trade Gets Green Light Under State Emissions Law*, S.F. CHRON., Dec. 8, 2011, at C2.

11. Egelko, *supra* note 10.

12. Alice Kaswan, *Reconciling Justice and Efficiency: Integrating Environmental Justice into Domestic Cap-and-Trade Programs for Controlling Greenhouse Gases*, in *THE ETHICS OF GLOBAL CLIMATE CHANGE* 232, 240–42 (Denis G. Arnold ed., 2011); Chinn, *supra* note 2, at 80–81; Johnson, *supra* note 2, at 111–12. Note that “co-pollutant” is typically defined to encompass all emissions of air toxics from a facility, regardless of whether they are co-emitted with GHGs. I will use this broad definition but will be careful to note where air toxics are not co-emitted.

13. See TODD SCHATZKI & ROBERT N. STAVINS, ADDRESSING ENVIRONMENTAL JUSTICE CONCERNS IN THE DESIGN OF CALIFORNIA'S CLIMATE POLICY 6–14 (2009), available at http://www.analysisgroup.com/uploadedFiles/Publishing/Articles/Environmental_Justice.pdf.

14. EPA maintains two databases on toxic emissions, the National Emissions Inventory and Toxics Release Inventory, and one on cancer risks, the National-Scale Air Toxics Assessment. See *infra* Part II for details.

A few statistics will demonstrate the significance of the findings. Perhaps the most striking result is that industrial facilities rarely account for more than ten percent of aggregate toxic emissions from outdoor sources at the county- or census-tract level.¹⁵ In terms of cancer risks, industrial emissions of air toxics were associated with at least twenty deaths per million (the national average for all sources was fifty per million) and accounted for more than thirty percent of the cumulative cancer risks in 240 census tracts.¹⁶ To put this in perspective, cancer risks from industrial sources exceeded five per million (ten percent of the national average) in 3792 census tracts out of 65,000 nationally, or six percent of all tracts.¹⁷ The data show that mobile sources (for example, cars and trucks) and small point sources (for example, dry cleaners, gas stations, and landfills) dominate toxic emissions and risks in most jurisdictions.

EPA data also reveal that industrial sources are clustered in about ten states. Texas, with its collection of oil refineries and chemical plants, leads this group with toxic emissions that were more than double those of any other state. The density of facilities along the Gulf Coast is particularly high—toxic emissions from industries in Houston alone exceeded those in all but three states. Yet, despite Houston's status as the largest *industrial* hotspot in the country, large stationary sources collectively accounted for just over a quarter of the toxic emissions and about ten percent on average of the excess cancer risks from all sources in 2005.

The EPA data expose a seemingly inconsistent pattern typical of air toxic emissions. Industrial sources of air toxins are geographically concentrated, but even where their emissions are the highest they rarely dominate. This apparent incongruity is driven by greater emissions from other sources—air toxins are largely a product of many small sources, which include stationary and mobile sources as well as personal and commercial activities. The data reveal that the common association of toxic air emissions with large industries is inaccurate for most jurisdictions,¹⁸ although important exceptions exist to this general rule.

15. Others, including Congress, have noted the importance of mobile and small stationary sources in the past. COMM. ON ENV'T & PUB. WORKS, CLEAN AIR ACT AMENDMENTS OF 1989, S. REP. NO. 101-228, at 187 (1989) (noting “that . . . as much as 75 percent[] of the cancer incidence is attributable to area sources”); *see also* U.S. GOV'T ACCOUNTABILITY OFFICE [GAO], CLEAN AIR ACT: EPA SHOULD IMPROVE THE MANAGEMENT OF ITS AIR TOXICS PROGRAM 21–22 (2006), *available at* <http://www.gao.gov/new.items/d06669.pdf> (finding that “small stationary and mobile sources in total have accounted for more emissions than major stationary sources in every emissions inventory completed since [1990]”).

16. This statistic covers emissions of Occupational Safety and Health Administration (OSHA) carcinogens (about 200 substances) and is limited to census tracts in which the cumulative excess cancer risk was greater than ten per million.

17. In terms of absolute emissions, there are roughly 2850 facilities nationally that emit more than 1000 pounds of carcinogens per year (about three pounds per day), and they are located in about 2250 census tracts.

18. *See, e.g.*, Thomas O. McGarity, *Hazardous Air Pollutants, Migrating Hot Spots, and the Prospect of Data-Driven Regulation of Complex Industrial Complexes*, 86 TEX. L. REV. 1445, 1445 (2008) (observing that “[u]nlike the criteria pollutants, which come from ‘numerous or diverse mobile or stationary sources,’ [air toxics] are associated with particular industrial activities, like chemical plants and metal smelters” (quoting Clean Air Act §

These findings appear to be quite robust despite the uncertainties and potential biases of EPA's emissions inventories and cancer-risk estimates for air toxics. A rough bounding analysis of the EPA data suggests that the uncertainties are significant for a small number of industries and jurisdictions. While this analysis cannot resolve all of the potential concerns about the EPA data, it provides benchmarks for the reliability of my conclusions and highlights the empirical findings for which potential errors cannot be ignored.

The secondary status of industrial facilities as sources of toxic emissions has particular relevance to concerns about GHG-trading regimes. A simple calculation illustrates this point: If industrial sources account for roughly ten percent of cancer risks from air toxics, as they do in many industrialized census tracts in Los Angeles,¹⁹ then a drop of twenty percent in toxic emissions from industrial sources would cause at most a two percent decline in cumulative cancer risks.²⁰ This ten-fold factor limits the potential for inequities to arise at the scale of a census tract or county. Other factors, both economic and technical, reinforce this limit on inequities originating from GHG trading by industrial facilities. These findings suggest that a tradeoff often presumed between efficiency and equity will rarely exist for GHG-trading regimes in the United States, and that, where inequities are a potential concern, targeted policies could be adopted to mitigate them without compromising market efficiency.

It is important to recognize that environmental-justice activists oppose GHG-trading programs for reasons beyond their potential to exacerbate disparities in exposures to toxic co-pollutants. Among other concerns, they believe that GHG-trading regimes undermine participatory justice,²¹ impede deployment of renewables, and are unreliable in practice.²² This Article addresses only one of the grounds for their opposition, albeit one that has figured prominently in their advocacy and has had salience with policymakers. Resolving questions about environmental disparities is thus a critical element of the current debate, but it will not neutralize activists' opposition to GHG trading. It could even reinforce their

108(a)(1)(B), 42 U.S.C. § 7408(a)(1)(B) (2000)).

19. See *infra* Part IV.A. The average for the Los Angeles air basin is much lower—a little over two percent of the cancer risk is attributable to point sources. S. COAST AIR QUALITY MGMT. DIST., MATES III: MULTIPLE AIR TOXICS EXPOSURE STUDY IN THE SOUTH COAST AIR BASIN 3–10 (2008) [hereinafter MATES III], available at <http://www.aqmd.gov/prdas/matesIII/MATESIIIFinalReportSept2008.html>.

20. The targets in California, which are the most aggressive nationally, require GHG emissions to be reduced to 1990 levels by 2020. CAL. HEALTH & SAFETY CODE § 38550 (West Supp. 2012). Additionally, California has a soft target (under an Executive Order) of an eighty percent reduction below 1990 levels by 2050. CAL. AIR RES. BD., FINAL SUPPLEMENT TO THE AB 32 SCOPING PLAN FUNCTIONAL EQUIVALENT DOCUMENT 6 (2011), available at http://www.arb.ca.gov/cc/scopingplan/document/final_supplement_to_sp_fed.pdf.

21. LUKE W. COLE & SHEILA R. FOSTER, FROM THE GROUND UP: ENVIRONMENTAL RACISMS AND THE RISE OF THE ENVIRONMENTAL JUSTICE MOVEMENT 13–15 (2001); Alice Kaswan, *Distributive Justice and the Environment*, 81 N.C. L. REV. 1031, 1045–47 (2003).

22. See, e.g., Eileen Gauna, *An Essay on Environmental Justice: The Past, The Present, and Back to the Future*, 42 NAT. RESOURCES J. 701, 708–09 (2002); The California Environmental Justice Movement's Declaration on Use of Carbon Trading Schemes to Address Climate Change (2008) [hereinafter EJ Declaration], available at <http://www.environmentalhealth.org/PDFs/CAEJDeclarationonClimateChange.pdf>.

support for a tax on emissions of GHGs,²³ which, like all market-based regulations, is susceptible to causing toxic hotspots.²⁴

The Article proceeds in four parts. Part I describes the current debate over market-based systems and the objections raised by environmental-justice advocates. To provide the empirical grounding for the debate, Part I then evaluates the literature on environmental inequities associated with outdoor sources of toxic air pollutants. Parts II and III present analyses of EPA data on toxic emissions and cancer risks, respectively, and evaluate geographic patterns at the county, city, and census-tract levels.²⁵ Part IV analyzes the implications of the EPA data for GHG-trading regimes in California and for the country as a whole.

I. THE CLASH OVER TOXIC HOTSPOTS AND POLLUTION MARKETS

Concerns about disparate impacts on low-income and minority communities lagged behind the emergence of the environmental movement in the 1960s and 1970s. The failure of environmental policies to address these distributional issues was first brought to light in the 1980s through a combination of high-profile grassroots community campaigns and academic studies.²⁶ These efforts grew into what is now known as the “environmental justice” movement, which promotes distributional equity and the participation of poor communities of color in regulatory processes that impact them.²⁷ The political salience of the movement grew through campaigns against the siting of new industrial facilities that were modeled on civil rights struggles²⁸ and empirical studies that revealed undesirable land uses (for example, hazardous waste facilities and industrial plants) were disproportionately located in poor communities of color.²⁹

One might presume that climate-change policies and environmental-justice principles would be aligned or, at worst, not overlap. Environmental-justice advocacy has focused on community-level disparities in environmental quality,

23. See Charles Komanoff, *National Environmental Justice Coalition Blasts Cap-and-Trade, Backs Carbon Tax*, GRIST (June 17, 2008, 3:33 AM), <http://grist.org/politics/no-justice-no-cap/>.

24. Johnson, *supra* note 2, at 138 (observing that a tax “on pollution discharges . . . without regard to the location of the discharge . . . could create toxic hot spots in the same manner as pollutant trading systems”).

25. Census tracts are small, statistical subdivisions of a county that usually have between 2500 and 8000 persons, but their spatial sizes vary widely depending on population density. Census tracts were established to enable statistical comparisons from census to census. *Tracts and Block Numbering Areas*, U.S. CENSUS BUREAU (Apr. 19, 2000), http://www.census.gov/geo/www/cen_tract.html.

26. See COLE & FOSTER, *supra* note 21, at 20–21; CHRISTOPHER H. FOREMAN JR., *THE PROMISE AND PERIL OF ENVIRONMENTAL JUSTICE* 13–18 (1998).

27. Sheila Foster, *Justice from the Ground Up: Distributive Inequities, Grassroots Resistance, and the Transformative Politics of the Environmental Justice Movement*, 86 CAL. L. REV. 775, 838–41 (1998).

28. COLE & FOSTER, *supra* note 21, at 20–21 (describing how the environmental justice movement applied and extended the methods used by civil rights activists).

29. ROBERT D. BULLARD, PAUL MOHAI, ROBIN SAHA & BEVERLY WRIGHT, *TOXIC WASTES AND RACE AT TWENTY, 1987-2007*, at 2–4 (2007); Gauna, *supra* note 22, at 702–04.

which appear distantly related to the global-scale phenomena driving climate change—it is the globally averaged concentration of GHGs that drives global warming. Yet, environmental-justice advocates have raised multiple objections to GHG-pollution-trading regulations,³⁰ ranging from systemic concerns about their implementation to moral objections about granting rights to pollute.³¹

This Article addresses just one of these objections—heightened inequities in exposures to toxic co-pollutants that a GHG-trading program could permit. This focus is motivated by the prominence of the issue in the public debate over GHG-trading programs, the divisions it has created within the environmental community, and the salience it has with policymakers.³² The potential for hotspots also represents a widely recognized shortcoming of market-based environmental regulations generally,³³ and thus the new findings presented in Parts II and III have implications beyond GHG-trading regimes.

The potential for inequities follows from the market structure of a cap-and-trade regime. Pollution trading has three basic elements: (1) a cap on aggregate GHG emissions, (2) a system for allocating emissions quotas to specific facilities, and (3) a trading market for GHG credits that allows firms to efficiently buy and sell them.³⁴ Under this regime, firms can choose to meet their emissions allowance (and not participate in the trading market), aggressively reduce their emissions and sell excess credits, or exceed their emissions quota and purchase additional credits to offset the shortfall.³⁵ If the costs of reducing GHG emissions are variable across different facilities or industries, a pollution-trading system will reduce the cost of meeting an aggregate-emissions target relative to a regime in which all sources must meet the same standard. This efficiency gain is achieved by allowing companies for which reducing GHG emissions is costly to purchase credits generated by companies with lower emissions-reduction costs.

The simplest, and most likely, scenario for a GHG-trading system to create inequities would involve major industrial facilities located in low-income and minority communities choosing to purchase permits over reducing GHG emissions.³⁶ If emissions of toxic co-pollutants were correlated with GHG emissions, the flexibility to purchase permits would have the secondary effect of allowing industrial facilities to emit toxic co-pollutants at their historic levels. At the same time, other sources would be collectively reducing their GHG emissions

30. See, e.g., J. ANDREW HOERNER & NIA ROBINSON, A CLIMATE OF CHANGE: AFRICAN AMERICANS, GLOBAL WARMING, AND A JUST CLIMATE POLICY FOR THE U.S. 11, 13 (2008), available at <http://urbanhabitat.org/files/climateofchange-2.pdf>; EJ Declaration, *supra* note 22, at 1–2; Kaswan, *supra* note 2, at 10287–88.

31. See *supra* note 30.

32. See Chinn, *supra* note 2, at 80–81; Kaswan, *supra* note 12, at 236, 240–42.

33. See *supra* note 32.

34. See Nash & Revesz, *supra* note 1, at 575–76.

35. *Id.* at 577.

36. Some commentators have suggested that GHG trading could allow emissions at a facility to increase. See, e.g., Kaswan, *supra* note 2, at 10299–300. As a practical matter this appears unlikely. A GHG-trading program would be applied on top of existing regulations and effectively increase the cost of emitting toxic co-pollutants. As discussed below, *infra* Part IV.B., this and other factors weigh against such increased emissions at a facility.

and, assuming a correlation with emissions of toxic co-pollutants, this would cause a concurrent reduction in their toxic emissions. A GHG-trading regime could therefore allow emissions of toxic pollutants to remain roughly the same in affected low-income and minority communities while levels of air toxics would fall elsewhere. The net effect would be to increase (or create) inequities in toxic pollution levels experienced by poor communities of color.

A GHG-trading program would not inevitably cause inequitable exposures to air toxics. The likelihood of this occurring is an empirical question, with the relevant factors including: (1) the geographic distribution of industrial facilities with large emissions of GHGs and toxic co-pollutants, (2) the relative cost of reducing GHG emissions at high-emitting facilities, (3) the correlation between GHG and toxic emissions at high-emitting facilities, and (4) the contribution of emissions from such industrial sources to aggregate toxic emissions and cancer risks.³⁷

Information now exists on emissions of air toxics to assess these factual predicates and to determine whether GHG-pollution-trading regimes could exacerbate environmental inequities. EPA collects extensive data on toxic emissions and their cumulative cancer risks by source category (for example, point sources, non-point sources, and mobile sources)³⁸ and at geographic scales ranging from the nation as a whole to individual census tracts. These databases are complemented by the EPA's Toxics Release Inventory, which is a compilation of data on toxic emissions from individual industrial sources.

Numerous environmental-justice studies have used the EPA data (or collected their own) to examine the geographic distribution of industrial facilities and their local impacts. Three basic types of studies exist: (1) those that focus exclusively on emissions from industrial sources, (2) those that evaluate emissions from all categories of outdoor sources (point, non-point, mobile), and (3) those that estimate cancer risks from all sources of toxic air pollutants, including indoor as well as outdoor sources.³⁹ Most environmental-justice studies have focused on emissions from industrial sources only—ignoring contributions from other source categories altogether.

37. See SCHATZKI & STAVINS, *supra* note 13, at 6–18.

38. Point sources include large industrial facilities and electric power plants but also increasingly include many smaller industrial and commercial facilities, such as dry cleaners and gas stations. Non-point sources (previously “area sources”) include all stationary sources not treated as “point sources” because their locations cannot be accurately measured at the facility level (for example, small manufacturers, fireplaces/wood stoves, and prescribed burns). Mobile sources include on-road vehicles (for example, cars, trucks, and buses) and nonroad sources (for example, trains, ships, construction equipment, and farm machinery). Background emissions include natural sources, persistent air toxics (for example, those originating from a previous year’s emissions), and long-range emissions (for example, those greater than fifty kilometers). ICF INT’L, AN OVERVIEW OF METHODS FOR EPA’S NATIONAL-SCALE AIR TOXICS ASSESSMENT 19 (2011) [hereinafter NATA OVERVIEW], available at http://www.epa.gov/ttn/atw/nata2005/05pdf/nata_tmd.pdf.

39. Indoor sources are critically important because “most people spend as much as 80 to 90% of their time indoors.” EPA, NATIONAL AIR TOXICS PROGRAM: THE INTEGRATED URBAN STRATEGY 2-4 (2000), available at <http://www.epa.gov/ttn/atw/urban/natprpt.pdf>.

Potential inequities of GHG-trading programs cannot be evaluated by focusing solely on industrial sources. The disparities center on elevated risks to public health,⁴⁰ which places a premium on more direct measures of risk (for example, emissions levels and cancer risk estimates) that implicate numerous sources. Otherwise, disparities that would be trivial relative to aggregate outdoor emissions of air toxics could appear significant when viewed in isolation. Thus, although environmental-justice advocates have focused on industrial sources, a full assessment requires that toxic emissions from all sources be considered and that emissions from industrial sources be evaluated in relative as well as absolute terms.

Another important shortcoming of existing studies is their failure to consider the spatial scale over which hotspots of air toxics occur. Typically, questions about spatial scale are either ignored or set implicitly by the available data, which rarely have a spatial resolution below the census-tract level.⁴¹ Because the spatial scales are set arbitrarily, the scale of the data for toxic emissions and excess cancer risks often will not match the spatial scale of the heightened impacts associated with toxic emissions from specific sources.

Recent studies suggest that the divergence could be substantial between localized hotspots and geospatial data on toxic emissions and risks. Elevated impacts of toxic emissions can be limited to a short distance from a point source but will vary dramatically from as little as several hundred meters to a few kilometers.⁴² The most reliable monitoring data are for mobile sources, and they show that hotspots occur within 0.09 to 0.35 miles of a roadway (a scale much smaller than a census tract).⁴³ Recognizing this variability, EPA's demographic studies default to evaluating impacts at two radii: five and fifty kilometers from a source.⁴⁴ EPA has cautioned that, while such an analysis "gives some indication of

40. See Chinn, *supra* note 2, at 86, 94–95; Drury et al., *supra* note 3, at 251; Johnson, *supra* note 2, at 130–31; Kaswan, *supra* note 2, at 10293–301.

41. In cities, census tracts typically cover areas of about two square miles; in rural areas, the spatial averaging of toxic emissions or risks can be over much larger spatial areas (for example, tens or hundreds of square miles). NATA OVERVIEW, *supra* note 38, at 27–28.

42. Telephone Interview with Ted Palma, Senior Physical Scientist, EPA (Nov. 2, 2011) [hereinafter Palma Interview]; see also Jawad S. Touma, Vlad Isakov, Jason Ching & Christian Seigneur, *Air Quality Modeling of Hazardous Pollutants: Current Status and Future Directions*, 56 J. AIR & WASTE MGMT. ASS'N 547, 548 (2006) (observing that "the maximum impact ranges from a few hundred meters to a few kilometers from [an individual] source").

43. See Alex A. Karner, Douglas S. Eisinger & Deb A. Niemeier, *Near-Roadway Air Quality: Synthesizing the Findings from Real-World Data*, 44 ENVTL. SCI. & TECH. 5334, 5341 (2010); Touma et al., *supra* note 42, at 550 (finding elevated levels of benzene from roadways at distances "between 200 and 400 m"); Clifford P. Weisel, *Benzene Exposure: An Overview of Monitoring Methods and Their Findings*, 184 CHEMICO-BIOLOGICAL INTERACTIONS 58, 61 (2010) (finding elevated levels of benzene in homes when located less than 200 meters from "busy roadways or gasoline stations"); see also Vlad Isakov, John S. Irwin & Jason Ching, *Using CMAQ for Exposure Modeling and Characterizing the Subgrid Variability for Exposure Estimates*, 46 J. APPLIED METEOROLOGY & CLIMATOLOGY 1354, 1362 (2007) (concluding that "modeling at 4 km, and perhaps even at finer grid sizes of the order of 1 km, cannot accurately characterize 'hot spots'").

44. National Emission Standards for Hazardous Air Pollutants from Coal- and Oil-

populations that may be exposed to levels of pollution that cause concern, it does NOT identify the demographic characteristics of the most highly affected individuals or communities.”⁴⁵

Resolving the appropriate scale of analysis for assessing the impacts of toxic hotspots is challenging in itself, but obtaining the relevant monitoring or modeling data poses an even greater impediment.

Few studies have attempted to do this systematically, and, to my knowledge, no studies of environmental disparities currently do so. Thus, while spatial scale is critically important to assessing inequities, the relationship between the distance from a source of toxic emissions and its impact on human health is complex and poorly characterized for most sources. The discussion below of current environmental-justice studies accepts them on their own terms—in effect, treating hotspots as defined by the spatial scale of the relevant data.⁴⁶

In the two subsections that follow, I review the empirical work assessing disproportionate impacts on low-income and minority communities. The first subsection analyzes studies on the siting of industrial sources in low-income and minority communities and the impacts of their toxic emissions. The second subsection examines studies of disproportionate cancer risks from all sources of air toxics. The studies cover different types of sources, including indoor sources, to put industrial emissions of air toxics in context.

A. Empirical Support for the Inequitable Impacts of Industrial Facilities

The empirical work initially focused on the siting of industrial facilities and hazardous waste sites in low-income and minority neighborhoods.⁴⁷ This work did not assess the magnitude of the risks to human health; instead, the authors viewed the existence of major plants as harmful without attempting to quantify the impacts on human health. The results, which began to be published in the late 1980s, garnered significant public and governmental attention and galvanized the environmental-justice movement into the 1990s.⁴⁸

The emergence of this empirical work sparked a great deal of controversy among social scientists and prompted numerous subsequent studies, which examined associations with other facilities (for example, chemical plants) and employed a variety of empirical methods.⁴⁹ The conflicting findings that resulted from this work provoked several reassessments of the methods and data. Points of disagreement centered on the geographic scale of analyses, the statistical methods

Fired Electric Utility Steam Generating Units and Standards of Performance for Fossil-Fuel-Fired Electric Utility, Industrial-Commercial-Institutional, and Small Industrial-Commercial-Institutional Steam Generating Units, 76 Fed. Reg. 24976, 25089 (proposed May 3, 2011) (codified at 40 C.F.R. pts. 60 & 63).

45. *Id.*

46. I will return to the issue of scale as it pertains to hotspots in Part II.A.2.

47. Evan J. Ringquist, *Assessing Evidence of Environmental Inequities: A Meta-Analysis*, 24 J. POL’Y ANALYSIS & MGMT. 223, 225–26 (2005).

48. COLE & FOSTER, *supra* note 21, at 20–21, 24–26; Gauna, *supra* note 22, at 701–03.

49. Ringquist, *supra* note 47, at 226–28.

employed, and the types of environmental risks investigated—each of which researchers argued could have a determinative effect on results.⁵⁰

Resolving the discrepancies between the studies has been complicated by the sheer volume of empirical work—there have been almost ninety studies in the United States alone.⁵¹ Attempts at literature reviews, each addressing subsets of the published studies, have hardly fared better, with divergent findings for and against the existence of significant environmental inequities.⁵² From a purely empirical perspective, the magnitude and, for some social scientists, the existence of environmental inequities in low-income and minority communities remain hotly contested.

1. Meta-Analysis of Environmental Justice Studies

A recent empirical study by Evan Ringquist, a political scientist at Indiana University, attempts to synthesize the existing literature through a meta-analysis of rigorous empirical studies.⁵³ He limited the meta-analysis to studies of environmental inequities associated with Superfund sites, industrial facilities with hazardous emissions, or elevated levels of toxic pollutants. Ringquist started with thousands of potentially relevant studies, pared this down to 297 studies that met his methodological and substantive criteria, and ultimately identified forty-nine acceptable studies.⁵⁴ Using formal meta-analytic techniques, Ringquist assessed the statistical significance and estimated the size of the inequities.

Ringquist's meta-analysis found clear, statistically significant associations between race and the location of hazardous industrial facilities and elevated pollution levels, but the results were equivocal for Superfund sites.⁵⁵ His study also showed that these results were unaffected by the geographic scale of a study, the

50. *Id.*

51. *Id.* at 230.

52. GAO, HAZARDOUS AND NONHAZARDOUS WASTE: DEMOGRAPHICS OF PEOPLE LIVING NEAR WASTE FACILITIES (1995) (concluding that no reliable evidence exists of environmental disparities); FOREMAN, *supra* note 26, at 27 (concluding that empirical research “must leave room for profound skepticism regarding the reported results” on environmental disparities); BENJAMIN A. GOLDMAN, NOT JUST PROSPERITY: ACHIEVING SUSTAINABILITY WITH ENVIRONMENTAL JUSTICE 9 tbl. 3 (1993) (review of sixty-four studies finding that the results of sixty-three showed statistically significant environmental disparities based on income or race); Paul Mohia & Bunyan Bryant, *Environmental Racism: Reviewing the Evidence*, in RACE AND THE INCIDENCE OF ENVIRONMENTAL HAZARDS: TIME FOR DISCOURSE 163 (Bunyan Bryant & Paul Mohai eds., 1992) (review of twenty-one studies finding strong support for the inequitable distribution of environmental risks based on race and income); Jean D. Brendor, Juliana A. Maantay & Jayajit Chakraborty, *Residential Proximity to Environmental Hazards and Adverse Health Outcomes*, 101 AM. J. PUB. HEALTH S37, S50 (2011) (concluding that “[a]lthough their results are mixed, many studies found significant relationships between residential proximity to environmental hazards and adverse health outcomes”).

53. Ringquist, *supra* note 47, at 223–24.

54. *Id.* at 230–31.

55. *Id.* at 232–34.

type of risk examined, or the type of control variables used.⁵⁶ Somewhat surprisingly, he did not find statistically significant associations with low-income status.⁵⁷

The statistical results of Ringquist's meta-analysis bear the claims of environmental-justice advocates that communities of color are subject to systemic inequities. However, his analysis runs counter to environmental-justice claims in one crucial respect—the magnitudes of the inequities. Ringquist's rough estimate of the association between race and environmental inequities found that “[b]y most yardsticks, the magnitude of class- and race-based environmental inequities are quite modest[,]” but he cautioned that his estimates likely represent a lower bound for the associations.⁵⁸ Thus, he finds that the magnitude of disparate risks is difficult to resolve conclusively, although it is clearly statistically significant.

2. Recent Studies Using Modeled Health-Effects Estimates

Two recent studies not considered by Ringquist, but consistent with his findings, benefit from new health-effects data released by the EPA. The data are based on the EPA's Risk-Screening Environmental Indicators (RSEI) model, which incorporates data on toxic emissions, chemical toxicity, and local fate and transport of the pollutants emitted.⁵⁹ The RSEI model estimates the cancer risks for people living in neighborhoods around major industrial sources. These studies illustrate the technical intricacies of attempting to quantify environmental inequities.

The first of the two studies assessed disparities in exposures to air toxics in 329 metropolitan areas in the United States.⁶⁰ The authors evaluated the pollution burden from industrial facilities for six racial groups using an ordinal relative-risk ranking system, which categorized each racial group according to whether it was the most pollution-burdened group, rank one, through the least, rank six.⁶¹ One of their primary findings was that African Americans and Hispanics were the first or second most pollution-burdened racial groups in more than half of the metropolitan areas studied, whereas Whites were the first or second most pollution-burdened group in only twelve percent of these areas.⁶²

56. *Id.* The only studies that failed to find statistically significant disparities used an improper method of selecting comparison communities (for example, neither low-income nor minority communities). *Id.* at 227.

57. *Id.* at 233–34.

58. *Id.* at 241. This result prompted Ringquist to argue that “promoting environmental equity, while important, ought to be viewed as one among a series of competing goals that would include enhancing the efficiency, effectiveness, innovativeness, and responsiveness of environmental regulation.” *Id.*

59. EPA, EPA'S RISK-SCREENING ENVIRONMENTAL INDICATORS (RSEI) METHODOLOGY 1 (2010), available at http://www.epa.gov/opptintr/rsei/pubs/rsei_methodology_v2.3.0.pdf.

60. Liam Downey, Summer Dubois, Brian Hawkins & Michelle Walker, *Environmental Inequality in Metropolitan America*, 21 *ORG. & ENV'T.* 270, 270 (2008).

61. The six racial groups were African Americans, Asian Americans, Hispanics, Native Americans, Pacific Islanders, and Whites. *Id.*

62. *Id.* at 278. African Americans also experienced greater disparities than other racial groups. *Id.* at 279.

Other results, however, did not reveal a consistent pattern of environmental inequities. In particular, Whites experienced pollution burdens that were higher than those of African Americans in twenty-one percent of the metro areas,⁶³ and increased segregation was correlated with African Americans being the least-burdened group.⁶⁴ Nor did the study find an association with low-income status.⁶⁵ The mixed results caused the authors to conclude that “environmental inequality outcomes vary widely across metropolitan areas” and that each racial group “experience[s] a heavy pollution burden in many metropolitan areas” from industrial sources.⁶⁶

The second study, *Justice in the Air*, employed a different set of metrics altogether.⁶⁷ The authors calculated the share of health risks from industrial facilities borne by low-income and minority communities.⁶⁸ These health-risk quotients were then compared against the low-income and minority proportion of the relevant population; if they were substantially higher, the authors inferred that inequities existed. The analysis evaluated disparities at the state and city levels as well as for specific industries and companies.

The state-level analysis found that the racial and low-income disparities were typically less than fifteen and five percent, respectively.⁶⁹ As one would expect for smaller spatial scales, the disparities were larger for cities. Racial disparities among the ten cities with the highest inequities were typically less than twenty percent, which in relative terms equates to average exposures for minorities that are two to three times those of whites.⁷⁰ Disparities based on low-income status were typically less than ten percent, or in relative terms on average about twice those of middle- and high-income communities.⁷¹ The industry-level data were of similar magnitude (about five percent), but disparities approached thirty percent for specific facilities (for example, oil refineries).⁷² Unfortunately, the authors did not analyze the disparities statistically, making it impossible to assess the degree to which the results were a product of systematic disparities as opposed to random variation.

63. *Id.*

64. *Id.* at 284–86.

65. *Id.*

66. *Id.* at 289.

67. See MICHAEL ASH, JAMES K. BOYCE, GRACE CHANG, MANUEL PASTOR, JUSTIN SCOGGINS & JENNIFER TRAN, *JUSTICE IN THE AIR: TRACKING TOXIC POLLUTION FROM AMERICAN’S INDUSTRIES AND COMPANIES TO OUR STATES, CITIES, AND NEIGHBORHOODS 2–4* (2009), available at <http://www.peri.umass.edu/justice/>. Similar to the other studies, major industrial facilities included those required to submit annual reports for the Toxic Release Inventory; health risks included cancer and non-cancer illnesses. *Id.*

68. *Id.* at 4–6.

69. *Id.* at 6–7. Tennessee had the greatest racial disparity at twenty-two percent, and Illinois had the highest income disparity at seven percent. *Id.*

70. *Id.*

71. Midsize cities in the southeast—Birmingham, Baton Rouge, and Memphis—displayed the highest racial disparities (twenty-two to thirty percent), with minority exposures in these cities on average about twice those of the White populations. *Id.* at 8. For income, Birmingham had the highest disparity (eleven percent). *Id.*

72. *Id.* at 12.

Consistent with Ringquist's meta-analysis, the RSEI-based studies found evidence that racial inequities are statistically significant, but the strength of the associations varied according to the metrics employed and the aggregation of the data. Some discrete patterns stand out in both studies. In particular, African Americans had a higher likelihood of being exposed to disparate health risks from industrial air toxics and for the risks to be greater than those of other groups.

3. Estimating Potential Inequities in GHG-Trading Systems

*Minding the Climate Gap*⁷³ assesses the potential for the proposed GHG-trading program in California to affect the geographic distribution of toxic emissions from industrial facilities. The report focused on industrial facilities with large CO₂ emissions and employed distance as a proxy for exposure to simplify their analysis further.⁷⁴ Using particulate matter (PM) that is less than ten microns in diameter as the sole pollutant, exposures were estimated according to whether a census block fell within a specified radius of a facility. The authors then calculated population-weighted emissions burdens based on demographic data.

The results revealed that disparities varied with the radius of the analysis—ranging from minor disparities at a half-mile radius to disparities of a factor of three between African Americans and Whites at a six-mile radius.⁷⁵ At every distance, pollution burdens for Whites lagged behind those for minorities. Equally important, petroleum refineries, which are concentrated in Los Angeles and the San Francisco Bay area, were found to account for ninety-three percent of the disparities between Whites and minorities.⁷⁶

The authors conclude that the potential exists for the operation of a cap-and-trade regime in California to disproportionately impact communities of color.⁷⁷ They also express particular concern about the limited number of facilities driving inequities and their geographic localization, as this could heighten the disparities experienced by neighboring communities.⁷⁸

The studies described in this subsection address one of the factual predicates—the citing of industrial sources and impacts of their toxic emissions. The next section augments this work with information on the relative contributions of industrial facilities to aggregate emissions of air toxics from all sources.

73. MANUEL PASTOR, RACHEL MORELLO-FROSCH, JAMES SADD & JUSTIN SCOGGINS, *MINDING THE CLIMATE GAP: WHAT'S AT STAKE IF CALIFORNIA'S CLIMATE LAW ISN'T DONE RIGHT AND RIGHT AWAY* 5 (2010) [hereinafter *Minding the Climate Gap*], available at <http://dornsife.usc.edu/perc/documents/mindingthegap.pdf>.

74. *Id.*

75. *Id.* at 16. The authors used a “pollution disparity index,” which “measures the relative co-pollutant [PM] burden on communities of color, as compared with non-Hispanic white communities.” *Id.* at 15. For each facility, this was the average pollution burden for people of color minus the average pollution burden for non-Hispanic Whites. *Id.*

76. *Id.* at 18. The impact of these facilities was increased further because the background levels of PM in San Francisco and particularly Los Angeles were very high (due to other sources). *Id.* at 27–28.

77. *Id.* at 21–22.

78. *Id.* at 8–10.

B. Disparities in Risks from Air Toxics—Indoor and Outdoor Sources

Several studies have been conducted on environmental inequities associated with outdoor sources of toxic air pollutants individually and collectively. They use the same basic methods as those that focus solely on emissions from industrial sources, but the process of inventorying emissions from multiple source categories is more complex and thus introduces additional sources of uncertainty. This may be particularly true for estimates of emissions from numerous smaller sources that are difficult to measure directly and thus rely on crude proxies.⁷⁹

1. Evidence of Inequities from All Outdoor Sources of Air Toxics

One of the earliest studies was focused on the Los Angeles metropolitan area and was conducted by several of the investigators who contributed to the *Minding the Climate Gap* report.⁸⁰ The authors found positive correlations between cancer risks and race that were consistent with or greater than the studies in Part I.A,⁸¹ but their most notable finding was that mobile and non-point sources accounted for about ninety percent of the cumulative cancer risks.⁸² Summing up their results, the authors concluded that “air toxics concentrations and their associated health risks originate mostly from smaller area and mobile sources.”⁸³

A second study evaluated relative cancer risks (RCRs) for different racial groups based on the degree of segregation in metropolitan areas.⁸⁴ The study found that RCRs increased with the degree of segregation in a community, but the effects varied by racial group—Hispanics were subject to the most pronounced disparities and Whites the least.⁸⁵ The RCR averaged 1.32 (1.28 to 1.36) for minorities collectively in extremely segregated metro areas and rose to 1.74 (1.61 to 1.88) for Hispanic communities.⁸⁶

79. NATA OVERVIEW, *supra* note 38, at 27–28 (describing the use of population and other proxies to apportion emissions from non-point and mobiles sources between census tracts).

80. Rachel Morello-Frosch, Manuel Pastor Jr., Carlos Porras & James Sadd, *Environmental Justice and Regional Inequality in Southern California: Implications for Future Research*, 110 ENVTL. HEALTH PERSP. 149, 150 (2002).

81. *Id.* at 151.

82. *Id.* at 152.

83. *Id.*

84. Rachel Morello-Frosch & Bill M. Jesdale, *Separate and Unequal: Residential Segregation and Estimated Cancer Risks Associated with Ambient Air Toxics in U.S. Metropolitan Areas*, 114 ENVTL. HEALTH PERSP. 386, 386 (2006) (also based on the 1995 NATA results and 1990 U.S. census data).

85. *Id.* at 387–88. The authors quantified the degree of segregation using a “dissimilarity index” (DI), “which is interpreted as the proportion of the racial group of interest that would need to relocate to another census tract to achieve an even distribution throughout a metropolitan area.” *Id.* at 388.

86. *Id.* at 390–92. Numbers in parentheses represent the ninety-five percent confidence interval.

The observed disparities were pervasive because seventy-five percent of U.S. metropolitan areas were either highly or extremely segregated.⁸⁷ They were also significant in absolute terms. In 1990, the average cancer risk from air toxics nationally was 632 per million, most of which (about eighty percent) was attributable to mobile-source emissions of diesel PM.⁸⁸ Point sources accounted for 1.3 percent of the cancer risks (or about eight per million), but the averaging over multiple metropolitan areas obscured higher industrial contributions in specific cities.⁸⁹ For Hispanics, the pollution levels translated to a cancer risk of 1084 per million in extremely segregated metro areas.⁹⁰ By comparison, EPA aims to reduce cancer risks below one per million and views cancer risks above 100 per million to be clearly unacceptable.⁹¹

Subsequent iterations of EPA's National-Scale Air Toxics Assessment (NATA) and the resulting improvements in the available data have enabled social scientists to undertake detailed assessments of source contributions to cumulative cancer risks. A 2008 study of the Houston metropolitan area is among the leading studies to evaluate racial and socioeconomic factors using census-tract data on cancer risks.⁹² The new data enabled the authors to evaluate variations in cancer risks across Houston and to correlate these patterns with demographic data.

The skewed nature of the risks was among the most important findings—a small number of census tracts were subject to about twice the average cancer risk for Houston residents, which was 676 per million in 1999.⁹³ The authors found associations between race, income, and educational level for census tracts with the highest pollution levels.⁹⁴ For example, census tracts with higher Hispanic populations were much more likely to be among tracts with the highest cancer risks.⁹⁵ Similar to other studies, the disparities were driven largely by emissions from mobile and non-point sources,⁹⁶ although cancer risks from industrial sources were not low in absolute terms (averaging eighty-five per million and ranging as high as 256 per million).⁹⁷

87. *Id.* at 391. The authors use a “multigroup dissimilarity” index (“Dm”), which “is the minimum number of people who would need to move from one neighborhood to another so that the distribution of each racial/ethnic group in every neighborhood matches that of the metropolis as a whole.” *Id.* at 388. Under the definition, a highly segregated area has a Dm of 0.40-0.60, and an extremely segregated area has a Dm of greater than 0.60. *Id.*

88. *Id.* at 389.

89. *Id.*

90. *Id.*

91. National Emission Standards for Hazardous Air Pollutants; Benzene Emissions from Maleic Anhydride Plants, Ethylbenzene/Styrene Plants, Benzene Storage Vessels, Benzene Equipment Leaks, and Coke By-Product Recovery Plants, 54 Fed. Reg. 38,044, 38,045 (Sept. 14, 1989) (codified at 40 C.F.R. pt. 61).

92. Stephen H. Linder, Dritana Marko & Ken Sexton, *Cumulative Cancer Risk from Air Pollution in Houston: Disparities in Risk Burden and Social Disadvantage*, 42 ENVTL. SCI. & TECH. 4312, 4318–20 (2008).

93. *Id.* at 4314.

94. *Id.*

95. *Id.*

96. *Id.* at 4315–17.

97. *Id.* at 4314. These figures represent the fifth to ninety-fifth percentiles.

The Houston study is particularly valuable with regard to its findings on the spatial patterns of emissions from different types of sources. While mobile and non-point sources accounted for about eighty-five percent of the cancer risks from air toxics throughout most of the city,⁹⁸ in several neighborhoods with the highest pollution levels industrial sources accounted for almost seventy percent of the cancer risks.⁹⁹ These findings highlight the geographic variation in cancer risks and source contributions that can exist in major metropolitan areas.

2. Evidence of Inequities from Outdoor and Indoor Sources of Air Toxics

As part of their efforts to construct comprehensive inventories of toxic emissions, scientists have sought to quantify risks from indoor as well as outdoor sources. The Relationships of Indoor, Outdoor, and Personal Air (RIOPA) study is a leading scientific effort to measure emissions from all sources of air toxics.¹⁰⁰ The research collected data from 1999 to 2001 on three cities (Elizabeth, NJ; Houston, TX; and Los Angeles, CA) with diverse populations and high levels of emissions from all source categories.¹⁰¹

A recent analysis of the RIOPA data compared the cumulative risks and sources of exposure for Hispanics and Whites in the study. The study found high cancer risks for both subgroups: median cancer risks of 519 and 443 per million for Hispanics and Whites, respectively.¹⁰² The racial disparities were greatest in Houston (sixty percent), but the results were truly shocking for the most exposed cohorts of Hispanic residents, who were subject to a cumulative cancer risk of 3968 per million (versus 751 per million for Whites).¹⁰³

The principal pollutants driving the cancer risks were from indoor sources,¹⁰⁴ and the highest cancer risks were strongly associated with a single chemical (p-DCB) used in air fresheners and moth repellents.¹⁰⁵ These results were consistent

98. *Id.* at 4319.

99. *Id.*

100. CLIFFORD P. WEISEL, JUNFENG (JIM) ZHANG, BARBARA J. TURPIN, MARIA T. MORANDI, STEVEN COLOME, THOMAS H. STOCK & DALIA M. SPEKTOR, HEALTH EFFECTS INSTITUTE, RELATIONSHIPS OF INDOOR, OUTDOOR, AND PERSONAL AIR (RIOPA): PART I. COLLECTION METHODS AND DESCRIPTIVE ANALYSES 1–2 (2005), available at <http://pubs.healtheffects.org/view.php?id=31>.

101. Diana E. Hun, Jeffrey A. Siegel, Maria T. Morandi, Thomas H. Stock & Richard L. Corsi, *Cancer Risk Disparities Between Hispanic and Non-Hispanic White Populations: The Role of Exposure to Indoor Air Pollution*, 117 ENVTL. HEALTH PERSP. 1925, 1925–26 (2009).

102. *Id.* at 1925.

103. *Id.* at 1927–28.

104. *Id.* at 1927 (collectively formaldehyde, para-dichlorobenzene (p-DCB), acetaldehyde, chloroform, and benzene accounted for eighty-three percent of the cancer risks for Hispanics and ninety-two percent for Whites).

105. *Id.* at 1927–28. p-DCB alone accounted for eighty-eight percent of the mean cancer risks for the top third of the Hispanic population in Houston and fifty-three percent in Los Angeles and Elizabeth. *Id.*

with cancer risk estimates and the chemicals identified as the primary risk drivers in other studies.¹⁰⁶

Collectively, the available studies on environmental inequities suggest several broad conclusions. The first is that solid, statistically significant evidence exists for racial inequities from emissions of air toxics; by contrast, the evidence for inequities based on income is equivocal. Further, racial inequities have been demonstrated for stationary sources on their own and for multiple sources collectively, including outdoor and indoor sources of air toxics.

Estimating the magnitude of racial inequities has proved more challenging and variable. The estimates range from “quite modest,” according to Ringquist’s formal statistical analysis of industrial sources,¹⁰⁷ to substantial for racial disparities from all outdoor sources of air toxics.¹⁰⁸ Among the studies of industrial sources, *Justice in the Air* provides the most direct estimates of inequities, finding racial disparities from toxic exposures to be as high as thirty-one percent in cities and forty-six percent around individual industrial facilities.¹⁰⁹ These numbers are upper bounds on risk disparities, as opposed to measures of systematic inequities. They and other data in *Justice in the Air* suggest that racial disparities from industrial emissions of air toxics typically fall below twenty percent and very rarely exceed thirty percent.

One of the clearest results of this work concerns the relative contributions from different categories of outdoor sources. The studies consistently find that emissions of air toxics from area and mobile sources are dominant in most jurisdictions, although, as demonstrated by the Houston study, important exceptions exist. The findings for outdoor sources are reinforced by the study that includes exposures from indoor sources of air toxics. Together the studies highlight the importance of addressing emissions from mobile, non-point, and indoor sources, both with respect to aggregate cancer risks and disparate impacts on minority groups.

II. U.S. TOXIC EMISSIONS INVENTORIES, CANCER RISKS, AND GEOGRAPHIC HOTSPOTS

The debate over inequities associated with toxic hotspots, as illustrated by the studies discussed in the preceding section, has focused largely on national studies of cancer risks or discrete studies of industrial emissions or cancer risks in specific jurisdictions. Empirical studies have not attempted to map toxic air emissions

106. *Id.* at 1929–30. The other studies included Benjamin J. Apelberg, Timothy J. Buckley & Ronald H. White, *Socioeconomic and Racial Disparities in Cancer Risk from Air Toxics in Maryland*, 113 ENVTL. HEALTH PERSP. 693, 697–98 (2005); Jeanetta E. Churchill, David L. Ashley & Wendy E. Kaye, *Recent Chemical Exposures and Blood Volatile Organic Compound Levels in a Large Population-Based Sample*, 56 ARCHIVES ENVTL. HEALTH 157 (2001); and Jennifer C. D’Souza, Chunrong Jia, Bhramar Mukherjee & Stuart Batterman, *Ethnicity, Housing and Personal Factors as Determinants of VOC Exposures*, 43 ATMOSPHERIC ENV’T 2884 (2009).

107. Ringquist, *supra* note 47, at 241.

108. Morello-Frosch & Jesdale, *supra* note 84, at 389–90.

109. ASH ET AL., *supra* note 67, at 11–12 (finding the minority share of toxic exposures for the Baton Rouge Refinery was seventy-eight percent, whereas the minority share of the U.S. population is nearly thirty-two percent, leading to a disparity of forty-six percent).

systematically or to examine patterns in source-category contributions across the United States. Yet, this kind of analysis has the potential to provide firmer empirical support for current debates as well as to reveal the prevalence of toxic hotspots nationally and the primary sources that are responsible for them. Moreover, EPA has been releasing increasingly detailed and accessible databases that make this type of analysis increasingly tractable.

I will provide an overview of air toxics in the United States using data on toxic emissions levels and cumulative, cancer-risk estimates (*see* Table 1). The two types of data provide complementary views of toxic hotspots across the country, as each metric has its strengths and limitations. Broad trends in emissions of air toxics, for example, reveal the relative importance of different source categories.

Table 1: EPA Emissions and Cancer Risk Databases for Air Toxics¹¹⁰

Database Name	Metric	Sources Covered	Years Compiled
Toxic Release Inventory	Emissions (Pounds)	Major point sources	Annually (1988-2010)
National Emissions Inventory	Emissions (Tons)	All source categories	Tri-Annually (2005, 2002, 1999, 1996)
National-Scale Air Toxics Assessment	Cancer Risk (excess mortality per million)	All source categories	Tri-Annually (2005, 2002, 1999, 1996)

Table 2 displays the trends in aggregate emissions of air toxics nationally, which dropped more than fifty percent between 1993 and 2005,¹¹¹ and the contributions from the three primary source categories. While the percentages of emissions from point sources have declined and those from mobile sources remained flat, the percentage from non-point sources increased substantially. As a consequence, industrial sources now account for a small fraction of air-toxics emissions, while mobile and non-point sources dominate emissions nationally.¹¹²

110. The analysis that follows utilizes the data in the three EPA databases: the *Toxics Release Inventory*, EPA, <http://www.epa.gov/tri/tridotnet/index.html>; the *2005 National Emissions Inventory Data & Documentation*, EPA, <http://www.epa.gov/ttnchie1/net/2005inventory.html>; and the *National-Scale Air Toxics Assessment*, EPA, <http://www.epa.gov/airtoxics/nata2005/tables.html>.

111. The declines in emissions of air toxics are paralleled by declines in the levels of air toxics found in human blood samples, although many of the exposures are from indoor sources. *See, e.g.*, Feng-Chiao Su, Bhramar Mukherjee & Stuart Batterman, *Trends of VOC Exposure Among a Nationally Representative Sample: Analysis of the NHANES 1988 Through 2004 Data Sets*, 45 *ATMOSPHERIC ENV'T* 4858, 4866 (2011).

112. The implications of these trends are well recognized. *See, e.g.*, C. Andrew Miller, George Hidy, Jeremy Hales, Charles E. Kolb, Arthur S. Werner, Bernd Haneke, David Parrish, H. Christopher Frey, Leonora Rojas-Bracho, Marc Deslauriers, Bill Pennell & J. David Mobley, *Air Emission Inventories in North America: A Critical Assessment*, 56 *J. AIR WASTE MGMT. ASS'N* 1115, 1115 (2006) (stating that “reductions in the largest emissions sources” have elevated the importance of smaller sources).

Table 2: Percentage of Total Air-Toxics Emissions by Source Category

Year	Total Emissions ¹¹³	Mobile Sources	Non-Point and Other Sources	Point Sources
1993	6.4	51	29	19
1999	4.6	48	34	17
2002	4.1	46	43	12
2005 ¹¹⁴	3.7	48	39	13

The data on emissions of air toxics come from two sources, the Toxic Release Inventory (TRI)¹¹⁵ and the tri-annual National Emissions Inventory (NEI).¹¹⁶ The TRI data are based on reported emissions from major industrial sources that must be submitted annually,¹¹⁷ whereas the NEI data encompass emissions from all outdoor sources of air toxics (for example, large and small stationary sources, on-road and nonroad mobile sources).¹¹⁸ A subset of the data in the TRI and NEI is derived from direct measurements of toxic air emissions, but because direct measurement of toxic emissions is difficult, most of the data are based on estimates derived from algorithms or computer models.¹¹⁹

113. Total emissions are in millions of tons. GAO, *supra* note 15, at 22. The annual emissions were altered to remove hydrochloric acid, which is not a carcinogen and had the second-largest emissions of any chemical in 2005. The method I used almost certainly overcorrects for emissions of hydrochloric acid in earlier estimates, implying that the relative drop in emissions from point sources is steeper than these figures suggest.

114. This estimate is based on the 2005 National Emissions Inventory data and excluded hydrochloric acid, ammonia, carbon dioxide, unknown (as of January 19, 2008), and all of the criteria pollutants (for example, lead, carbon monoxide, nitrogen oxides, sulfur dioxide, all particulate matter including condensable matter, and volatile organic compounds (VOCs)).

115. *See supra* note 110.

116. *Id.*

117. *Basics of TRI Reporting*, EPA (July 19, 2012), <http://www.epa.gov/tri/triprogram/bussinesscycle/index.html> (the basic reporting requirements apply to certain listed industries as well as to any company with greater than ten employees that manufactures or processes greater than 25,000 pounds of a TRI-listed chemicals annually or otherwise uses more than 10,000 pounds of a listed chemical annually).

118. *See 2005 National Emissions Inventory Data & Documentation*, *supra* note 110 (descriptions of data and detailed documentation on the 2005 NEI).

119. EPA OFFICE OF INSPECTOR GENERAL, EPA CAN IMPROVE EMISSIONS FACTORS DEVELOPMENT AND MANAGEMENT 2-4 (2006), *available at* <http://www.epa.gov/oig/reports/2006/20060322-2006-P-00017.pdf> (noting that “emissions factors are used for about 80 percent of emissions determinations”); N. AM. RESEARCH STRATEGY FOR TROPOSPHERIC OZONE, AN ASSESSMENT OF TROPOSPHERIC OZONE POLLUTION: A

EPA has a program dedicated to promulgating the “emissions factors” incorporated into the algorithms used to estimate emissions from individual sources (for example, chemical storage tanks at refineries and kilns at cement plants).¹²⁰ Notwithstanding EPA’s best efforts, the emissions factors are subject to significant uncertainties.¹²¹ EPA’s ranking of its emissions factors reflects these difficulties—sixty-two percent of the factors are rated either below average (twenty-eight percent) or poor (thirty-four percent) according to the quality of the data upon which they were derived.¹²² Independent studies have also exposed uncertainties in EPA’s emissions inventories for mobile sources and non-point sources.¹²³ However, the largest underestimates of emissions have been associated with a small number of industrial sources,¹²⁴ most notably petroleum refineries and ethanol plants; to its credit, EPA has subsequently sought to correct them.¹²⁵

NORTH AMERICAN PERSPECTIVE 3-23 (2000), available at ftp://narsto.esd.ornl.gov/pub/Ozone_Assessment/NARSTO_Ozone_Assessment_Complete.pdf [hereinafter NARSTO] (observing that “[a]fter 20 years of effort, emission estimates continue to be one of the weakest links in the air-quality management process and a major source of uncertainty”).

120. EPA OFFICE OF INSPECTOR GENERAL, *supra* note 119, at 24 (observing that “EPA officials describe the emissions inventory as the foundation for the air program, upon which everything else is built”).

121. According to one prominent public-private organization, “[d]espite the improvements that have been made, there remains a great deal of uncertainty in these estimates” for emissions of VOCs from point sources. NARSTO, *supra* note 119, at 3-28. See also Miller et al., *supra* note 112, at 1120 (stating that “[t]here can be considerable uncertainty in the data” in part because “many, if not most, source emission estimates are based on a small number of measurements that [are not representative] of process designs and operational practices”).

122. EPA OFFICE OF INSPECTOR GENERAL, *supra* note 119, at 8. Further, a majority of EPA’s emissions factors were developed using ten points of data or less, and none of them are accompanied by a formal uncertainty estimate. *Id.* at 17-18.

123. NARSTO, *supra* note 119, at 3-26 (finding studies “suggested that mobile sources are responsible for a larger portion of the total VOC emissions than that predicted by emission inventories”); *id.* at 3-28 (suggesting that “emissions from stationary area sources are even more difficult to accurately characterize” as “[d]irect measurement of area source emissions is hardly ever practical because of technical and cost considerations”). More recent work by EPA has addressed problems with low emissions estimates for mobile sources. See Richard Cook, Jawad S. Touma, Antonio Fernandez, David Brzezinski, Chad Bailey, Carl Scarbro, James Thurman, Madeleine Strum, Darrell Ensley & Richard Baldauf, *Impact of Underestimating the Effects of Cold Temperature on Motor Vehicle Start Emissions of Air Toxics in the United States*, 57 J. AIR & WASTE MGMT. ASS’N 1469, 1477-78 (2007).

124. Studies have suggested that some emissions estimates could be low by more than a factor of ten. *E.g.*, Memorandum from Brenda Shine to EPA Docket No. EPA-HQ-OAR-2003-0146 on Potential Low Bias of Reported VOC Emissions from the Petroleum Refining Industry 1 (July 27, 2007), available at <http://www.greenhoustontx.gov/reports/lowbias.pdf> (discussing studies “indicat[ing] that emissions of VOC from refineries are significantly higher (ten to twenty times) than amounts estimated using standard techniques”); Alex Cuclis, Presentation/Paper for National Petrochemicals and Refiners Association, Sweden’s Approach to Refinery Emission Inventories and Their Influence on the U.S. EPA 3-6, (Oct. 24-25, 2011) (unpublished draft) (on file with Indiana Law Journal) (finding that “over 35 studies performed between 1988 and 2008 show that measured emissions are consistently considerably higher than reported emissions,” and that model estimates can be low by a factor of ten).

125. EPA OFFICE OF INSPECTOR GENERAL, *supra* note 119, at 8, 10. Recognizing these

Quantifying the uncertainties of EPA's emissions factors and improving their accuracy is technically challenging and costly.¹²⁶ EPA's primary check has involved benchmarking its emissions inventories against direct measurements of air toxics.¹²⁷ In the most recent assessment of the 2005 NEI, EPA compared model pollutant levels derived from the NEI data against monitored ambient levels of sixty-eight air toxics.¹²⁸ The model estimates for several of the most important air toxics nationally (such as acetaldehyde, benzene, butadiene, formaldehyde, and naphthalene) were found to be within a factor of two of the ambient levels measured.¹²⁹ Other studies also suggest that the uncertainties in EPA's emissions inventories largely fall within this range.¹³⁰ Thus, although significant errors and uncertainties persist, the data are approaching a level of reliability that experts would like to attain consistently.¹³¹

underestimates, EPA enforcement actions during the mid-2000s reduced "over one million tons of pollutants for the three industries" (petroleum refineries, wood products, and ethanol production). *Id.* at 10. EPA has also taken action at other facilities. *See, e.g.*, N.Y. STATE DEP'T OF ENVTL. CONSERVATION, TONAWANDA COMMUNITY AIR QUALITY STUDY, ES-1 to ES-3, 10-1 to 10-2 (2009), available at http://www.dec.ny.gov/docs/air_pdf/tonairfinalrpt.pdf; Press Release, EPA, EPA and DEC Report Progress at Tonawanda Coke Corporation (July 20, 2011), available at <http://yosemite.epa.gov/opa/admpress.nsf/0/99741C7F3CF58E23852578D3005447FB> (reporting on new commitments imposed on Tonawanda Coke Corp. to reduce toxic emissions by a factor of ten higher than the emission-inventory estimate).

126. NARSTO, *supra* note 119, at 3-28 (observing that "it is difficult to quantify this uncertainty due, in part, to the difficulty of obtaining information on the techniques used in the development of the estimates"); Miller et al., *supra* note 112, at 1120-22 (describing the sources of uncertainty and challenges of overcoming them).

127. Nationally, air toxics are monitored at more than one thousand locations, although monitors are disproportionately located in urban areas. E. RESEARCH GRP., INC., RESULTS OF THE 2005 NATA MODEL-TO-MONITOR COMPARISON, FINAL REPORT 1-1, 2-4 (2010), available at http://www.epa.gov/ttn/atw/nata2005/05pdf/nata2005_model2monitor.pdf.

128. *Id.* at 3-1.

129. *Id.* at 2-6, 3-5. Model estimates were within a factor of two (0.5 to 2.0) of the measured pollutant levels for forty-four percent of the 5621 annual averages calculated. *Id.* at 3-5. Potentially mitigating concerns, an important factor believed to bias model estimates was background concentrations (for example, pollutants transported long distances), as most pollutants for which the models underestimated ambient levels did not take them into account. *Id.* at 3-20.

130. Touma et al., *supra* note 42, at 549 (stating that "[m]odel-to-monitor comparisons showed that the model performed within a factor of two at most sites for inert gases, such as benzene, but underpredicted for metals"); COMM. ON AIR QUALITY MGMT. IN THE U.S., NAT'L RESEARCH COUNCIL, AIR QUALITY MANAGEMENT IN THE UNITED STATES 99 (2004) (indicating that "emissions inventories are generally held to have an uncertainty of about a factor of two or more, although . . . the uncertainty factor is poorly defined"); Devon C. Payne-Sturges, Thomas A. Burke, Patrick Breysse, Marie Diener-West & Timothy J. Buckley, *Personal Exposure Meets Risk Assessment: A Comparison of Measured and Modeled Exposures and Risks in an Urban Community*, 112 ENVTL. HEALTH PERSP. 589, 597 (2004) (concluding that "results suggest that for pollutants primarily of ambient origin [EPA's model] provides a reasonable (within a factor of 2) central estimate for personal exposures").

131. Palma Interview, *supra* note 42; Interview with Dave Allen, Gertz Regents Professor of Chem. Eng'g, Univ. of Tex. at Austin (Dec. 22, 2011) [hereinafter Allen

The second type of data covers cancer risk estimates that EPA generates triannually under its National-Scale Air Toxics Assessment (NATA).¹³² Although the 2005 NATA contains data on noncancer risks, my analysis excludes them because the estimates are less reliable and because point sources generally account for a lower percentage of the noncancer risks from air toxics.¹³³ The cancer risk estimates use the NEI emissions data as an input for the EPA exposure models (such as the chemical breakdown and transport of air toxics).¹³⁴ The NATA results are thus dependent on the accuracy of the NEI data, the EPA exposure models, and toxicity estimates for each compound.¹³⁵ The complexity of the analyses that underlie the NATA cancer risk estimates introduces numerous opportunities for uncertainty and bias in the results.¹³⁶ Thus, while the NATA data provide a direct measure of risk, they must be interpreted cautiously.¹³⁷

The uncertainties and potential biases in the EPA data vary along two dimensions—geographic and source category. For intra- and interjurisdictional comparisons, it is critical to keep in mind that the quality and detail of information differ significantly depending on the geographic scale of the data (uncertainties are greater at smaller scales)¹³⁸ and by jurisdiction.¹³⁹ EPA cautions against using NATA results “as a definitive means to pinpoint specific risk values within a census tract, to characterize or compare risks at local levels such as between neighborhoods, [or] to characterize or compare risk among states”¹⁴⁰

Interview]. One should also keep in mind that a single monitor (or even a collection of them) may not be representative of ambient air toxic levels through a modeled area. Jawad S. Touma, William M. Cox & Joseph A. Tikvart, *Spatial and Temporal Variability of Ambient Air Toxics Data*, 56 J. AIR & WASTE MGMT. ASS'N 1716, 1724 (2006) (stating that “[t]he spatial variability of HAPs within cities is not consistent enough to assume that concentrations at one monitoring location are representative of other locations”).

132. The cancer risks are expressed as “typical lifetime excess cancer risk” of, for example, ten per million. NATA OVERVIEW, *supra* note 38, at 70.

133. According to the 2005 NATA, nationally point-source emissions of air toxics accounted for about 1.5% of respiratory risks from all outdoor sources and 17% of the neurological risks. EPA’s non-cancer risk data are also much less reliable at the county and census-tract levels than the cancer risk estimates.

134. NATA OVERVIEW, *supra* note 38, at 71–77 (describing the sources of uncertainty in deriving cumulative risk estimates for air toxics).

135. *Id.*

136. For example, NATA “might not accurately capture sources that have episodic emissions (e.g., facilities with short-term deviations in emissions resulting from startups, shutdowns, malfunctions, and upsets). The models assume emission rates are uniform throughout the year.” NATA OVERVIEW, *supra* note 38, at 7.

137. EPA cautions that the 1999 NATA cannot be compared with the 2005 NATA due to methodological differences between them. GAO, *supra* note 15, at 29–30. The GAO has noted that NATA “is useful for identifying the relative contribution of emissions from different sources” but questions its reliability in evaluating program effectiveness. *Id.* at 29.

138. NATA OVERVIEW, *supra* note 38, at 5 (stating that “although results are reported at the census tract level, average risk estimates are far more uncertain at this level of spatial resolution than at the county or state level”).

139. *Id.* at 77 (“EPA suggests exercising caution when using the results of these assessments, as the overall quality and uncertainty of each assessment vary from location to location and from pollutant to pollutant.”).

140. *Id.* at 5. EPA recommends that “[f]or analysis of air toxics in smaller areas, such as

Similarly, because of the spatial averaging over a census tract (or county), “individual exposures or risks might differ by as much as a factor of ten in either direction [i.e., above or below a calculated mean].”¹⁴¹ For cross-source comparisons, the quality of data for specific source categories (for example, point versus non-point sources) and for individual sources will also vary.¹⁴²

EPA asserts that the estimates of relative contributions across source categories are among the most robust,¹⁴³ but the uncertainties will be substantial for even the best, typically more-aggregated, data. The various sources of error are factored into a rough, bounding analysis described in Appendix A, which finds that, apart from a small number of jurisdictions, the potential errors would not alter the conclusions of the analysis that follows. Part IV evaluates the implications of these outlier jurisdictions for GHG-trading regimes and toxic hotspots nationally.

One final point warrants attention before analyzing the EPA data. I will emphasize relative, as opposed to absolute, emissions levels and risks. My focus on relative risks is guided by the concerns raised about the potential inequities of GHG-trading regimes, which revolve around relative disparities in risk exposures—emissions levels above those of alternative legal regimes. This should not be read to imply that high absolute risks from industrial sources can be ignored merely because they occur in highly polluted areas. Consideration of absolute risk levels is central to any regulatory policy, and, although a secondary consideration in the present analysis, it is factored explicitly into my definition of industrial hotspots. Appendix A discusses issues related to relative risks and metric selection in greater detail.

A. Industrial-Source Emissions at the Municipal and Neighborhood Levels

In this section, I will evaluate the geographic distribution of large industrial sources of toxic emissions nationally. The analysis will focus on the following variables: industry, geographic scale, class of air toxics, and source category. The first two variables are self-explanatory, but the latter two require some additional explanation. I will use two conventional classes of air toxics: (1) the class of about 200 hazardous air pollutants identified by the Occupational Science and Health Administration (OSHA) as known or suspected carcinogens (“OSHA Carcinogens”);¹⁴⁴ and (2) a small class of carcinogens that EPA has identified as national or regional risk drivers in the 2005 NATA (“NATA Toxics”).¹⁴⁵ While the

census blocks or in a suspected ‘hotspot,’ other tools such as site-specific monitoring and local-scale assessments coupled with refined and localized data should be used.” *Id.*

141. *Id.* at 69.

142. *See supra* note 117 and accompanying text.

143. *See* NATA OVERVIEW, *supra* note 38, at 5 (“EPA uses NATA results to identify pollutants and industrial source categories of greatest concern, improve understanding of health risks posed by air toxics, . . . set priorities for improving emission inventories, . . . [and] enhance targeted risk reduction activities . . .”).

144. OSHA Carcinogens are Emergency Planning and Community Right-to-Know Act (EPCRA) Section 313 listed toxic chemicals that meet OSHA carcinogen standard. *See Toxics Release Inventory (TRI) Basis of OSHA Carcinogens*, EPA (Nov. 2011), http://www.epa.gov/tri/trichemicals/OSHA/OSHA_carcinogen_table_2011.pdf.

145. The NATA Toxics include the following chemicals: 1,3-butadiene, 1,4-

class of OSHA Carcinogens is comprehensive, NATA Toxics account for a majority of the cancer risks nationally; both are important classes, but emissions of NATA toxics track cancer risks much more closely than emissions of OSHA Carcinogens.

I will use EPA's definitions of the four categories of sources (point, non-point, on-road mobile, and nonroad mobile) with one important qualification.¹⁴⁶ The terms "industrial source" and "point source" will be used interchangeably even though the point-source category includes smaller manufacturers and can encompass conventional non-point sources such as gas stations and dry cleaners.¹⁴⁷ Treating data on point sources as though they were limited to industrial sources will cause the putative estimates of emissions and risks from industrial sources to be conservative by virtue of being overinclusive. A benefit of this approach is that it operates as a rough offset for potential errors in the NEI data and particularly for any systematic biases in EPA's algorithms for emissions estimates.¹⁴⁸

1. Toxic Emissions from TRI Sources at the Municipal Level

The geographic concentration of major industrial sources of air toxics was highest in Texas and Louisiana. In each state, industrial emissions in three municipalities collectively accounted for about sixty percent of statewide emissions of OSHA Carcinogens in 2010. Among the other states with large emissions from industrial sources, Indiana, Illinois, and Alabama were in the middle with their top three municipalities accounting for about forty-five percent of the state total. In the remaining six states, the top three municipalities accounted for between twenty-five and thirty-eight percent of each state's total. Accordingly, Texas and Louisiana had both the highest aggregate TRI emissions in 2010 and the most geographically concentrated distribution of the industrial facilities responsible for them.¹⁴⁹

Industrial sources of air toxics are also unevenly distributed across metropolitan areas. Figure 1 shows that among municipalities with substantial emissions of air

bdichlorobenzene, acetaldehyde, acrylonitrile, benzene, chromium compounds, formaldehyde, naphthalene, polycyclic aromatic hydrocarbons, and tetrachloroethylene. EPA, SUMMARY OF RESULTS FOR THE 2005 NATIONAL-SCALE ASSESSMENT 3-4 (2011), http://www.epa.gov/ttn/atw/nata2005/05pdf/sum_results.pdf [hereinafter EPA, SUMMARY OF RESULTS]. Studies have shown that these are the primary drivers and contributors of cancer risks from air toxics. *See, e.g.*, Hun et al., *supra* note 101, at 1927 (describing one study that found that the main contributors to cumulative cancer risk were formaldehyde, *p*-dichlorobenzene, acetaldehyde, chloroform, and benzene).

146. See NATA OVERVIEW, *supra* note 38, at 9-10 for EPA's definitions of each category.

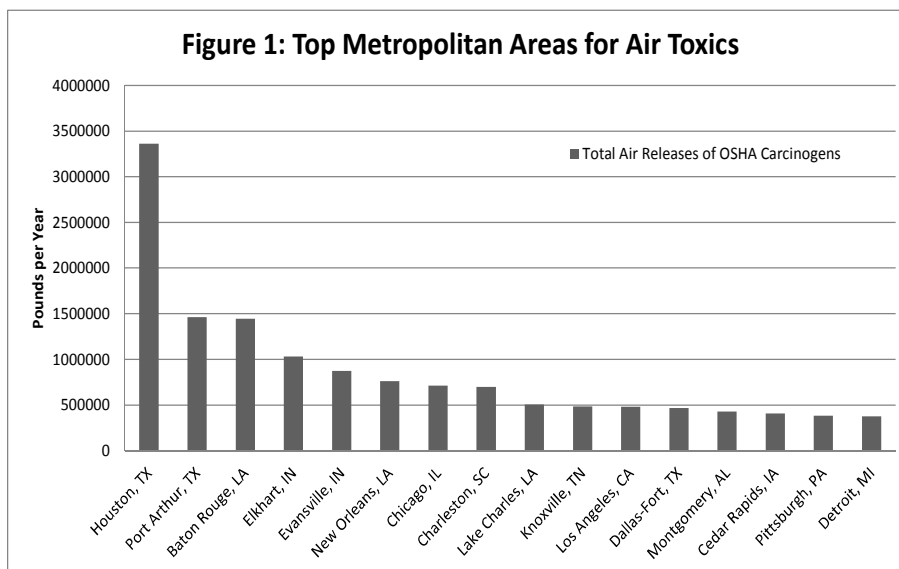
147. See NATA OVERVIEW, *supra* note 38, at 19.

148. The absence of reporting requirements for short-term releases associated with facility startup, shutdown, and unanticipated disruptions, some of which can double the annual emissions of a facility, is a major gap in the NEI data. ENVTL. INTEGRITY PROJECT, GAMING THE SYSTEM: HOW OFF-THE-BOOKS INDUSTRIAL UPSET EMISSIONS CHEAT THE PUBLIC OUT OF CLEAN AIR 1-3 (2004), http://www.environmentalintegrityproject.org/pdf/publications/eip_upsets_report_appendixa.pdf; NATA OVERVIEW, *supra* note 38, at 7.

149. Industrial emissions in California, which were much lower, were also concentrated: forty percent in the Los Angeles metropolitan area and over fifty percent in the top three metropolitan areas in the state.

toxics in 2010,¹⁵⁰ the top fifty metropolitan areas (out of about 420 nationally) accounted for more than seventy-five percent of the NATA Toxics emitted, whereas the bottom sixty percent (about 250 metropolitan areas) accounted for less than ten percent of the NATA Toxics emitted.¹⁵¹ These results demonstrate that a relatively small number of municipalities contain major industrial sources of air toxics and, by implication, that the potential for industrial hotspots of toxic air pollutants is likely to exist in only a few U.S. cities.¹⁵²

The aggregate emissions of OSHA Carcinogens from TRI sources drop off rapidly from the maximum found in Houston, which alone accounted for almost ten percent of such emissions nationally. Port Arthur, Texas and Lake Charles, Louisiana are treated separately in Figure 1 below but if aggregated would account for more than five percent of OSHA Carcinogens emissions from TRI sources in the United States. Together, the top three municipalities encompass TRI sources that account for almost twenty percent of U.S. emissions of OSHA Carcinogens.



Temporal trends indicate that industrial cities in Texas (Houston and Port Arthur) and Louisiana (Baton Rouge and New Orleans) will likely continue to have higher industrial emissions than the rest of the country. Heavily industrialized cities

150. "Substantial" is defined here as emissions of at least 10,000 pounds of NATA Toxics per year.

151. The results are similar for OSHA Carcinogens. The top fifty metropolitan areas accounted for more than sixty percent of the OSHA Carcinogens emitted, whereas the bottom sixty metropolitan areas accounted for less than ten percent.

152. It is important to remember that quantity does not track strictly with risk; emissions of highly toxic emissions (such as chromium) in relatively small amounts can pose significant risks. Madeline Strum, Rich Cook, James Thurman, Darrell Ensley, Anne Pope, Ted Palma, Richard Mason, Harvey Michaels & Stephen Shedd, *Projection of Hazardous Air Pollutant Emissions to Future Years*, 366 SCI. TOTAL ENV'T 590, 597-98 (2006).

in other states (such as Los Angeles and Chicago) have achieved emissions reductions that in absolute terms are now substantially below those in these two states. Throughout the twenty-three years of TRI data, Houston has consistently outpaced other cities by a large margin, although its aggregate toxic emissions have converged toward the levels found in the other cities with high TRI emissions levels.

The TRI data are remarkable for their consistency across a broad range of geographic scales. While emissions levels have been consistently dropping in most jurisdictions for more than two decades, major industrial sources of toxic emissions remain highly concentrated in a small number of states and localities. TRI emissions, with a few exceptions (Chicago, Illinois; Los Angeles, California; Pittsburgh, Pennsylvania), are found disproportionately in the southeastern states, with Texas and Louisiana standing out even among the states with the highest emissions.¹⁵³ At the municipal level, Houston exists in a category of its own—no other city in the country comes close to the level of air toxics emitted by its cluster of industrial sources.

2. Assessing the Available Data on Neighborhood-Scale Hotspots

The EPA emissions inventories and cancer-risk data are limited to spatial scales of counties or census tracts. Counties typically range in area from tens to more than one thousand square miles. Census tracts can also vary considerably in size, particularly in rural areas, but in cities they generally cover areas of about two square miles.¹⁵⁴ For purposes of modeling the movement and concentrations of air pollutants, obtaining a resolution below one square mile is quite challenging if the objective is to estimate air pollution levels from all sources across a metropolitan area.¹⁵⁵ However, recent studies of air pollution emanating from highways have shown that hotspots of air toxics can be localized at spatial scales of several hundred meters.¹⁵⁶

A potential problem with the EPA data is that localized hotspots can be obscured when the emissions associated with them are averaged over the much larger areas that census tracts and counties encompass. EPA acknowledges explicitly that its cancer risk data lack the resolution necessary to detect neighborhood-scale hotspots, and it notes that cancer risks within a census tract can vary by a factor of ten.¹⁵⁷ The potential therefore exists for many hotspots to exist, albeit spatially constrained and population limited, that are not captured by the EPA data. If accurate, the low resolution of the EPA data would present a misleadingly positive picture of the risks from industrial emissions of air toxics.

153. In 2010, emissions of OSHA Carcinogens from TRI-reporting sources in eleven states exceeded three percent of emissions from major sources nationally, and all but Ohio were either located in the southeast or contain a major center for petrochemical facilities. Industrial emissions of NATA Toxics in Texas were twice those of second-place Louisiana and more than four times those in most other states.

154. NATA OVERVIEW, *supra* note 38, at 27–28.

155. *See supra* notes 42–45 and accompanying text.

156. *See supra* notes 42–45 and accompanying text.

157. *See supra* notes 42–45 and accompanying text.

The data available to test this hypothesis are quite limited. The technical limitations of existing air pollution models and the high costs of operating high-resolution monitoring networks are the primary impediments. Some of the best scientific work on microscale hotspots has been conducted in Corpus Christi, Texas, which is home to several major petroleum refineries.¹⁵⁸ The site is particularly suitable for detecting microscale hotspots for three reasons: (1) the petroleum refineries are among the largest in the country, (2) petroleum refining is among the industrial processes with the highest emissions of air toxics, and (3) the toxic emissions from mobile and non-point sources in the Corpus Christi area are relatively low by urban standards.¹⁵⁹ The Corpus Christi data are also virtually unique with respect to the number of monitors around the refineries and the length of time over which high-quality measurements have been collected.¹⁶⁰

The results of this work are encouraging. Overall, the researchers found that the average annual ambient levels of pollutants such as benzene and 1,3-butadiene, both major air toxics nationally, were below the target level set by the Texas Commission on Environmental Quality (TCEQ) in all but one case.¹⁶¹ Further, most of the monitors were located either on the fence lines of the facilities or within a few hundred meters of them, and thus within a radius designed to detect microscale hotspots.¹⁶²

158. DAVID ALLEN, CORPUS CHRISTI AIR MONITORING AND SURVEILLANCE CAMERA INSTALLATION AND OPERATION PROJECT: ANNUAL PROGRESS REPORT FOR THE PERIOD OCTOBER 1, 2009 THROUGH SEPTEMBER 30, 2010, at 21, 26 (2011), available at <http://www.utexas.edu/research/ceer/ccaqp/PDF/COCPreports/2010-2011/COCPr%20Annual%20Rpt%203-24-11.pdf>; Interoffice Memorandum from Neeraja Erraguntla, Toxicology Division, Chief Engineer's Office, Tex. Council on Env'tl. Quality, Health Effects Review of 2010 Ambient Air Network Monitoring Data in Region 14, Corpus Christi (Aug. 25, 2011), available at http://www.tceq.texas.gov/assets/public/implementation/tox/monitoring/evaluation/2010/reg_14_corpus.pdf.

159. See ELENA McDONALD-BULLER, GARY MCGAUGHEY, YOSUKE KIMURA, DAVID ALLEN, EDWARD TAI, CHRIS COLVILLE, UARPORN NOPMONGCOL & GREG YARWOOD, DISPERSION MODELING OF BENZENE AND 1,3-BUTADIENE IN CORPUS CHRISTI, TEXAS iii (2010) (reporting that point sources accounted for about fifty percent of benzene emissions and about thirty percent of 1,3-butadiene emissions in Corpus Christi), available at [http://www.utexas.edu/research/ceer/ccaqp/PDF/NATPreports/2010/dispersion_modeling_report_v25%20\(2\)_PDF.pdf](http://www.utexas.edu/research/ceer/ccaqp/PDF/NATPreports/2010/dispersion_modeling_report_v25%20(2)_PDF.pdf); Allen Interview, *supra* note 131.

160. See generally McDONALD-BULLER ET AL., *supra* note 159 (describing the monitoring sites set up by the University of Texas).

161. *Id.* at iv–ix, x–xiii (finding that average annual ambient levels at all monitors surrounding the Corpus Christi facilities are now below 1.5 parts per billion by volume (ppbV) and all but one is below 1 ppbV); AIR POLLUTANT WATCH LIST BOUNDARY SUPPLEMENT DOCUMENTATION: BENZENE IN GALENA PARK 1, TEX. COMM'N ON ENVTL. QUALITY (July 2011), http://www.tceq.texas.gov/assets/public/implementation/tox/apwl/gp_boundary.docx (stating that the “no adverse long-term health effects are expected if the annual average benzene concentration for an area remains below 1.4 ppbV”); DEVELOPMENT SUPPORT DOCUMENT: 1,3-BUTADIENE 6, TEX. COMM'N ON ENVTL. QUALITY (Aug. 7, 2008), http://tceq.com/assets/public/implementation/tox/dsd/final/butadiene_1-3-_106-99-0_final.pdf (stating that the long-term Effects Screening Level (ESL) for 1,3-butadiene is 4.5 ppb).

162. McDONALD-BULLER ET AL., *supra* note 159, at ii. The specific sources within the refineries are also diverse, ranging from large stack emissions to diffuse fugitive emissions

The direct measurements of ambient levels of air toxics were complemented by high-resolution modeling studies. This work generated detailed maps of ambient pollution levels in and around the refineries. They revealed that the spatial extent of plumes with elevated concentrations of air toxics was small, such that cancer risks beyond the refinery fence lines had a low probability of exceeding a cancer risk of ten per million.¹⁶³ An important limitation on these results is that the time averaging of the measurements makes them insensitive to short-term spikes in emissions, which occur when a facility starts up, shuts down, or experiences a technical malfunction. These events, which are common, can cause large increases in annual emissions, but the impacts on cumulative risks have not been accurately assessed.¹⁶⁴

These results are obviously not representative of other industrial facilities—particularly steel mills and foundries—but they do provide a useful benchmark for whether microscale hotspots are likely to be a significant source of cancer risks around major sources of air toxics. Refineries are among the largest industrial facilities in the country and a leading industrial source of air toxics. The modeling results in Corpus Christi are also consistent with estimates of cancer risks from air toxics emitted by industrial facilities (including refineries) in California.¹⁶⁵

The prevalence of microscale hotspots can also be explored using data on the emission rates for individual facilities. This approach assumes that microscale hotspots are unlikely to occur below certain threshold levels of toxic emissions, which will vary according to the characteristics and location of a facility.¹⁶⁶ I will attempt to account for this variability by selecting conservative thresholds for toxic emissions at individual facilities. The Corpus Christi refineries provide an initial benchmark for large sources that are more likely to cause microscale hotspots. Their emissions individually were 100 to 180 pounds of OSHA Carcinogens per day (38,000 to 68,000 pounds per year) and 30 to 100 pounds of NATA Toxics per day.

The number of facilities potentially at risk differs between the two classes of air toxics. If thresholds of twenty and fifty pounds per day are used for OSHA Carcinogens (less than half those of the Corpus Christi refineries), the number of facilities at risk nationally of generating microscale hotspots is 1390 and 750, respectively. If thresholds of ten and twenty pounds per day are used for NATA Toxics, the number of facilities at risk is 715 and 550, respectively.¹⁶⁷ In California, the numbers are low in absolute terms: forty-one facilities emit more than twenty pounds of OSHA Carcinogens per day and thirteen emit more than ten pounds of NATA Toxics per day.

that emanate from elaborate networks of pipelines and storage tanks. *See id.* at iii.

163. *See id.*

164. *See generally* ENVTL. INTEGRITY PROJECT, *supra* note 148, at 1–3; NATA OVERVIEW, *supra* note 38, at 7.

165. *See infra* note 221 and accompanying text.

166. *See supra* notes 42–45 and accompanying text.

167. These estimates ignore heightened risk from highly toxic compounds, such as chromium. If facilities with highly toxic emissions, largely steel plants and electric utilities, are included, the estimates for NATA Toxics rise to 600 and 765 facilities for the ten- and twenty-pound thresholds, respectively.

These conservative estimates suggest that a substantial number of facilities nationally could cause microscale hotspots. However, most of these sources do not emit significant quantities of GHGs. If sources with significant GHG emissions are singled out, the number of facilities potentially at risk of causing microscale hotspots falls dramatically—at the ten- and twenty-pound thresholds for NATA Toxics, about 350 and 250 facilities nationally.¹⁶⁸ Moreover, many of these facilities are associated with hotspots at the census-tract level and thus would be covered by the definition of “industrial hotspot” used in Part IV to analyze the implications of the EPA data for GHG trading programs in California and at the national level.

The results of existing monitoring data and my largely heuristic assessment of at-risk facilities are far from definitive. They are nevertheless suggestive that the limited spatial resolution of the EPA data is unlikely to be a major source of error for the limited purpose of assessing whether a GHG trading regime is likely to exacerbate toxic hotspots. More work is clearly needed to help resolve these uncertainties¹⁶⁹ and would be of great value to communities located along the fence lines of major industrial facilities.

B. NEI Data—Industrial Emissions As a Share of Toxic Air Emissions from All Outdoor Sources

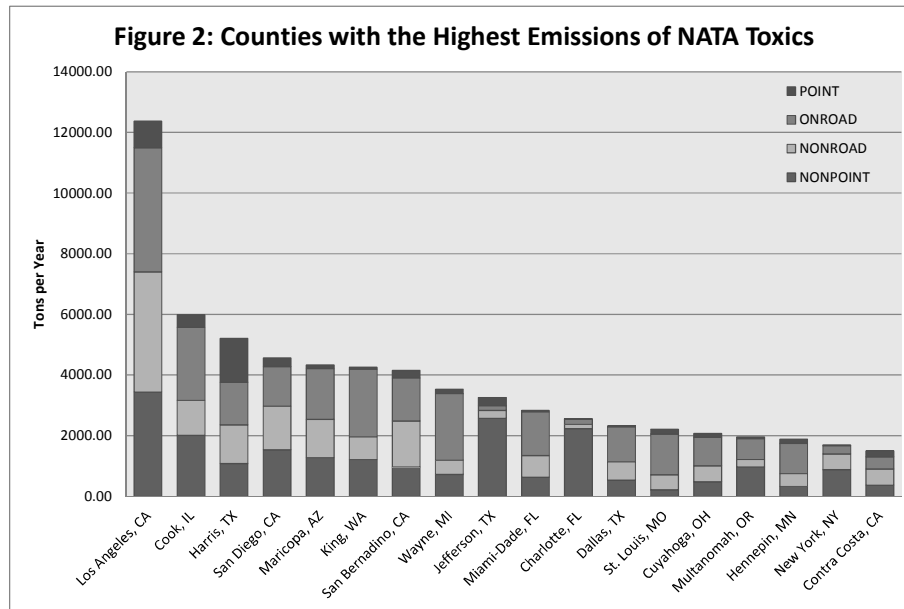
Up to this point the analysis has focused solely on toxic emissions from major industrial sources. Yet, emissions from small, non-point sources (gas stations and dry cleaners) and mobile sources (cars, trucks, and trains) collectively account for much larger proportions of toxic air emissions nationally (Table 2). Evaluating industrial emissions of air toxics in isolation ignores these sources and thus cannot be used to infer the relative impacts of industrial emissions or the significance of the geographic disparities revealed by the TRI data.

When non-point and mobile sources are taken into account, the geographic variation in toxic emissions, which in this section includes particulate matter (PM), shifts dramatically. Perhaps most significantly, the Houston area is no longer an outlier. The emissions of air toxics in several major metropolitan areas, many in California, are comparable to or exceed those in Houston, including Los Angeles, Chicago, New York, and the San Francisco Bay Area (see Figures 2 & 3). One important limitation of the NEI is that it does not include census-tract-level data, which is significant because the geographic areas encompassed can vary considerably—tens to thousands of square miles—from county to county.

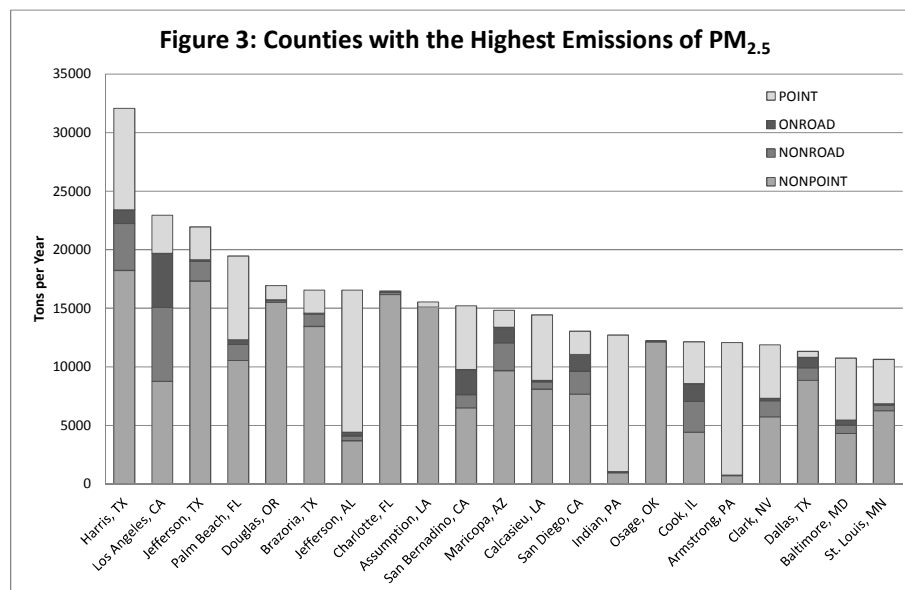
168. NATA Toxics are the appropriate metric because a relatively small number of air toxics (for example benzene, formaldehyde, and 1,3-butadiene) are emitted in significant quantities by the combustion processes that drive GHG emissions at industrial facilities.

169. Although not focused on industrial sources, EPA has launched an initiative to obtain representative data on hotspots around highways. See Karner et al., *supra* note 43, at 5534 (“[EPA]’s 2001 Mobile Source Air Toxics (MSAT) Rule identified locations near heavily trafficked roads as important microenvironments for MSAT exposure.”); Allen Interview, *supra* note 131.

Figures 2 and 3 present a representative collection of counties with some of the highest aggregate emissions of air toxics.¹⁷⁰ They reveal that the highest levels of toxic emissions from all outdoor sources are concentrated in a small number of metropolitan areas and that the patterns for cumulative emissions of air toxics differ substantially from those for industrial sources alone. Data on emissions from specific industries, which are provided in Appendix B, further bear out these findings.



170. This is not a strictly ordered listing of the counties with the highest emissions. If limited to the counties with the highest emissions, the figures would have more counties in southern California and Texas.



As mentioned above, the variable geographic sizes and population densities of these counties complicate analysis of the data. To put this in perspective, New York County encompasses a mere twenty-three square miles but has a population of 1.6 million, which equates to 71,000 people per square mile. Toward the other end of the spectrum, Harris County (the county in which Houston is located) encompasses 1729 square miles and has a population of 4.1 million, which equates to 2367 people per square mile. Despite this enormous variability, metropolitan areas with the highest emissions of NATA Toxics are generally the jurisdictions with the highest cancer risks, but total population and population densities are critically important factors as well.¹⁷¹

Appendix B also provides data on emissions of criteria pollutants, including NO_x and SO₂, by industrial sector. The 2005 NEI data show that emissions from major industrial sources varied markedly across the jurisdictions with the highest emissions from point sources. Emissions from electric utilities, which are of particular significance given their status as the largest industrial source of GHGs,¹⁷² were a minor point source of NATA Toxics and contributed modestly to NO_x emissions,¹⁷³ but they dominated point-source emissions of PM and SO₂ in a

171. This correlation might be expected given EPA's use of population as a proxy for apportioning emissions between census tracts from mobile and non-point sources. See NATA OVERVIEW, *supra* note 38, at 73 (describing population as one of the surrogates used in allocating emissions).

172. EPA, INVENTORY OF U.S. GREENHOUSE GAS EMISSIONS AND SINKS: 1990 – 2009, at ES-15 (2011) (showing that electric utilities, largely coal- and natural gas-fired facilities, generated thirty-three percent of U.S. GHG emissions in 2009).

173. Utility emissions averaged twenty-nine percent for the ten states with the highest point-source emissions of NO_x; the variation ranged from a low of eleven percent in Texas to a high of fifty-five percent in West Virginia (driven in part by the low, in relative terms, total emissions of NO_x in the state). By contrast, electric utilities accounted for one percent of

number of states and counties. As a share of total emissions in states with the highest emissions, utilities dominated SO₂ emissions but represented about one tenth of the total emissions of PM (in California the fraction was less than one percent).¹⁷⁴ Thus, while electric utilities were an important source of PM, the NEI data suggest that their localized (as opposed to regional) impacts dominate a small number of counties and states nationally. Utility emissions of NO_x, and particularly SO₂, were substantial, but their ambient levels (as well as those of PM) are regulated separately through the criteria pollutant program under Title I of the Clean Air Act (CAA).¹⁷⁵

The NEI data are perhaps most valuable for what they reveal about the relative contributions of the four different source categories. Among the counties with the highest NATA Toxics emissions, industrial sources were almost always minor contributors to toxic emissions. Harris County, Texas stands out as the only county with high emissions of NATA Toxics in which the contributions from all four source categories were comparable (Figure 2). Industrial sources accounted for more than fifty percent of NATA Toxics emissions in just three counties with substantial emissions (i.e., about 420 counties with annual emissions above 500 tons per year),¹⁷⁶ and, on average, point sources accounted for just five percent of the NATA Toxics emitted in this cohort of counties.

Consistent with the studies described in Part I, non-point and mobile sources accounted for a majority of air toxics emissions in most jurisdictions. In assessing these results, one must consider the inaccuracies in the NEI emissions data and variation in data quality between jurisdictions and source categories. The low contributions of stationary sources in most jurisdictions suggest that, absent truly extraordinary errors, the inaccuracies in EPA's emissions estimates would not overcome the predominance of mobile and non-point sources of toxic emissions found throughout most of the country.

NO_x emissions in California.

174. On average, electric utilities accounted for seventy-nine percent of SO₂ emissions among the ten states with the highest emissions nationally in 2005. Among the ten states with the highest emissions from point sources of PM, electric utilities accounted on average for eleven percent of PM emissions (in Pennsylvania and Ohio, electric utilities accounted for sixteen and nineteen percent, respectively). While the variation at the county level was greater, emissions of PM from electric utilities in just three counties (Indiana, Pennsylvania; Jefferson, Ohio; and Titus, Texas) accounted for a majority of local PM emissions and exceeded 5000 tons per year.

175. *See generally* HOLLY DOREMUS, ALBERT C. LIN, RONALD H. ROSENBERG & THOMAS J. SCHOENBAUM, ENVIRONMENTAL POLICY LAW: PROBLEMS, CASES, AND READINGS 611–17 (5th ed., 2008) (describing the rigorous CAA framework for regulating criteria pollutants).

176. The three counties were Palm Beach County, Florida (fifty-two percent point; 3849 tons per year); Weld County, Colorado (seventy-six percent point; 1898 tons per year); and Sweetwater County, Wyoming (seventy percent point; 543 tons per year). By contrast, total outdoor emissions of NATA Toxics in Harris County (Houston) and Los Angeles County were, respectively, 5488 and 12,374 tons per year.

III. RISK ESTIMATES FOR TOXIC AIR EMISSIONS: GEOGRAPHIC SCALE, DENSITY,
AND POPULATION

Studies of cumulative cancer risks from air toxics remain relatively rare in the United States. The first major studies based on reasonably comprehensive data did not emerge until the early 2000s, and most rely on EPA data and estimates of chemical toxicity. Using emissions data from 1990, a group of EPA scientists published a study showing that about three-quarters of the cancer risk from air toxics was attributable to just five types of compounds.¹⁷⁷ Consistent with other studies, they also calculated that the median cumulative cancer risk from air toxics in the United States was about 200 per million in 1990.¹⁷⁸

A second major study focused on cancer risks from a select group of high-risk chemicals and evaluated the relative impacts of indoor and outdoor sources of air toxics.¹⁷⁹ Using more recent 1999 emissions data, the study found that thirty-five percent of the cumulative cancer risk from air toxics derived from indoor sources, fifty percent from outdoor sources (both stationary and mobile), and fifteen percent from exposures through food.¹⁸⁰ It also found that five air toxics accounted for seventy to eighty percent of the cumulative risks from air toxics,¹⁸¹ which nationally averaged 600 per million (see Figure 4).¹⁸² Together, these studies revealed the importance of a small number of air toxics to cumulative cancer risk, the high levels of risk at stake, and the importance of indoor as well as outdoor sources.

177. Tracey J. Woodruff, Jane Caldwell, Vincent J. Coglianò & Daniel A. Axelrad, *Estimating Cancer Risk from Outdoor Concentrations of Hazardous Air Pollutants in 1990*, 82 ENVTL. RES. 194, 194 (2000). The specific compounds were polycyclic organic matter (forty percent), 1,3-butadiene (seventeen percent), formaldehyde (eight percent), benzene (seven percent), and chromium compounds (seven percent). *Id.* at 201.

178. *Id.*

179. Miranda M. Loh, Jonathan I. Levy, John D. Spengler, E. Andres Houseman & Deborah H. Bennett, *Ranking Cancer Risks of Organic Hazardous Air Pollutants in the United States*, 115 ENVTL. HEALTH PERSP. 1160, 1164–65 (2007) (the study focused on a subset of air toxics from the Woodruff study that accounted for more than eighty-seven percent of the cumulative cancer risk).

180. *Id.* at 1164 (showing most of the contaminants in food were attributable to outdoor sources). Consistent with the Woodruff work, Loh et al. found that the dominant outdoor air toxics included acetaldehyde, butadiene, benzene, and PAHs. *Id.* These chemicals were also identified as risk drivers in the two environmental inequity studies described in Part I. *See supra* Part I.B.

181. Loh et al., *supra* note 179, at 1164–65. Note that the California risk estimates are based on the California Office of Environmental Health and Hazard Assessment (OEHHA), but the EPA numbers were taken from the Integrated Risk Information System (IRIS). *Id.* at 1160.

182. The cumulative cancer risk estimate based on the California potencies, 600 per million, was within an order of magnitude of the value calculated by Woodruff. *See id.* The cumulative cancer risk using EPA's potencies, 1000 per million, was dramatically higher, but this difference is largely attributable to EPA's higher estimated risk for dioxin in food, which was not considered in the other studies. *See id.* at 1164.

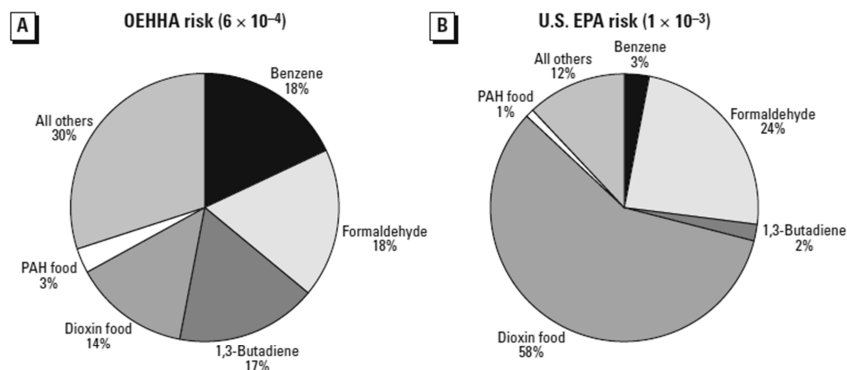
Figure 4: Mean Cancer Risk Estimates for Chemicals of Greatest Concern¹⁸³

Figure 3. Risk from ingestion and inhalation. (A) Mean total risk calculated with OEHHA's cancer potency values (6×10^{-4}). (B) Median total risk using the federal U.S. EPA's values (1×10^{-3}). The inhalation fraction is broken down further into several of the higher-risk compounds.

This work represented a major advance over studies based on emissions alone, but estimates of cancer risks entail numerous simplifying assumptions and require the use of complex exposure models.¹⁸⁴ Each of these elements of the analysis introduces potential sources of error and uncertainty that are exceedingly difficult to quantify rigorously. In EPA's description of the NATA, the Agency cautions against overinterpreting data.¹⁸⁵ The analysis that follows adopts this cautious approach when reviewing existing studies and analyzing the results of the 2005 NATA. I will attempt to be especially careful when relying on census-tract data, which are subject to the largest uncertainties.

A. Results of the 2005 National-Scale Air Toxics Assessment

Most of the existing studies use data from the 1990s, which, given the significant declines in toxic emissions during the intervening years, dates their risk estimates. In March of 2011, EPA issued updated risk estimates for air toxics as part of its 2005 NATA.¹⁸⁶ The new data—which do not include cancer risks from PM—showed that the average cumulative cancer risk from air toxics in 2005 had

183. Loh et al., *supra* note 179, at 1164 fig. 3 (reprinted with permission).

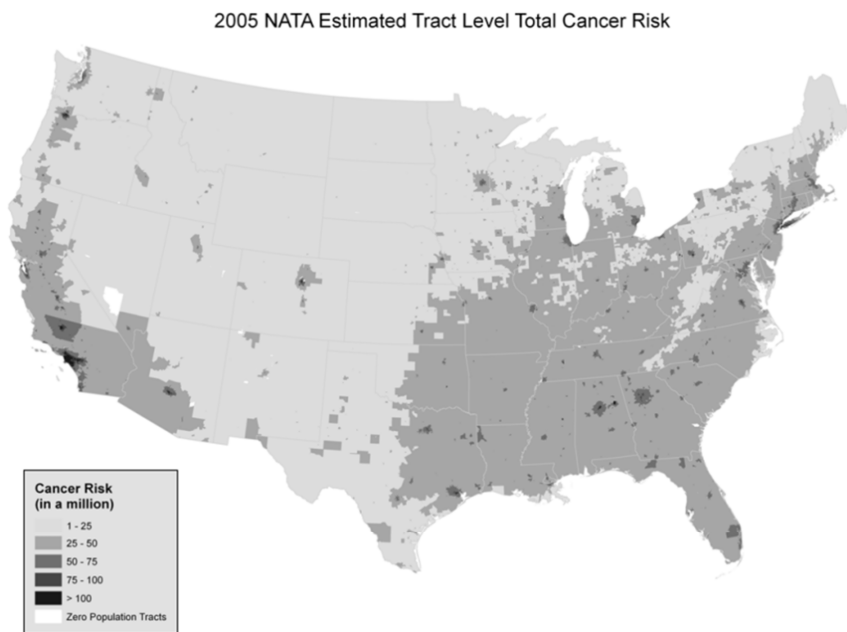
184. See NATA OVERVIEW, *supra* note 38, at 71–77; see also *supra* notes 128–130 and accompanying text.

185. EPA specifically warns users of the data against making interjurisdictional comparisons of cumulative risks because the certainty and quality of data available can vary considerably by jurisdiction. NATA OVERVIEW, *supra* note 38, at 77.

186. See generally EPA, NATIONAL-SCALE AIR TOXICS ASSESSMENT FOR 2005 FACT SHEET, http://www.epa.gov/ttn/atw/nata2005/05pdf/nata2005_factsheet.pdf (discussing the release of the 2005 NATA and differences from past NATAs). The 2005 NATA examined 177 air toxics plus diesel PM and evaluated the cancer and noncancer risks from inhalation of 139 of those air toxics. EPA, SUMMARY OF RESULTS, *supra* note 145, at 1.

dropped to fifty per million.¹⁸⁷ This result still left 285 million people exposed to toxic emissions with cancer risks above ten per million and 13.8 million exposed to cancer risks above 100 per million.¹⁸⁸ Thus, despite significant declines in emissions between 1990 and 2005, cancer risks for most Americans from air toxics exceeded the Clean Air Act's target risk level of one excess death per million.¹⁸⁹

Figure 5: Map of 2005 NATA Cumulative Cancer Risks by County¹⁹⁰



The 2005 NATA identified specific toxic chemicals of “regional”¹⁹¹ and “national”¹⁹² importance. EPA’s analysis designated formaldehyde as a national cancer risk driver and several regional cancer risk drivers, including benzene, polycyclic aromatic hydrocarbons, and naphthalene.¹⁹³ It also identified several

187. EPA, SUMMARY OF RESULTS, *supra* note 145, at 4–5.

188. *Id.*

189. *See* Clean Air Act, 42 U.S.C. § 7412(f)(2) (2006).

190. EPA, SUMMARY OF RESULTS, *supra* note 145, at 6.

191. A “regional cancer driver” is defined as either a risk of greater than ten per million affecting a population of at least one million people or a risk of greater than 100 per million affecting a population of at least ten thousand people. *See id.* at 3. A “regional noncancer driver” is defined as a risk of greater than one per million affecting a population of at least ten thousand people. *See id.*

192. A “national cancer driver” is defined as a risk of greater than ten per million (noncancer one per million) affecting a population of at least twenty-five million people. *Id.*

193. *Id.* The 2005 NATA also identified “noncancer” hazard drivers, which included acrolein nationally and several regional noncancer hazard drivers (such as diesel PM and chlorine). *Id.*

weaker national cancer risk “contributors,”¹⁹⁴ which included 1,3-butadiene, chromium, acetaldehyde, acrylonitrile, and 1,4-dichlorobenzene.¹⁹⁵ Overall, despite several shifts in their relative significance, the same collection of chemicals found to dominate cancer risks in earlier studies were again singled out in the 2005 NATA.

The census-tract data generated by the 2005 NATA is a significant advance over earlier assessments notwithstanding the higher uncertainties that come with higher spatial resolution.¹⁹⁶ The added detail confirms the variation in cancer risks from air toxics across the country and exposes the strong association of urban areas with the highest cancer risks (see Figure 5).¹⁹⁷ They also exhibit the pattern described earlier of non-point and mobile sources accounting for a majority of the cancer risks from air toxics in most jurisdictions.

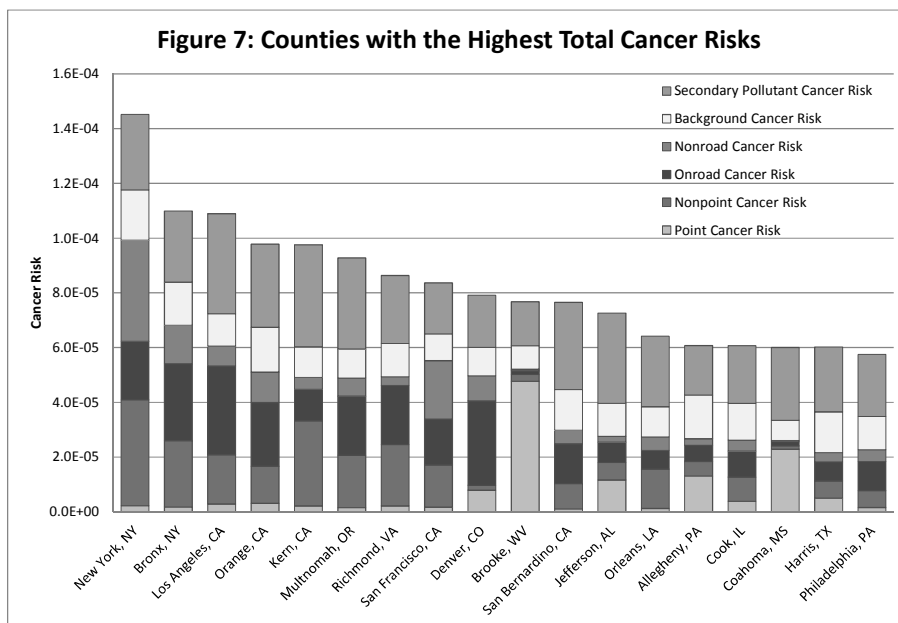
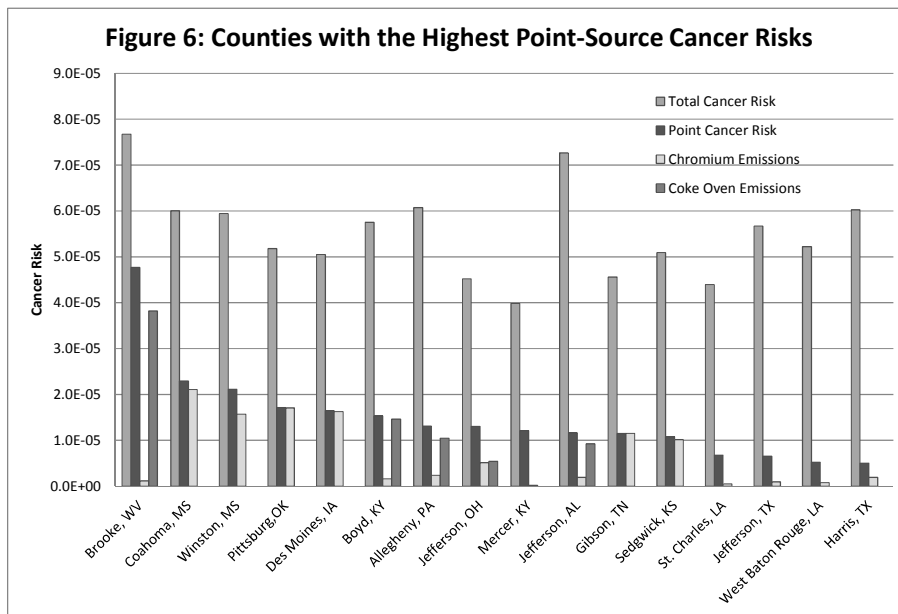
The 2005 NATA results are striking at multiple levels. Focusing first on industrial sources, one can evaluate the counties and census tracts with the highest absolute cancer risks attributable to industrial facilities (see Figure 6). At the county level, the data show that with very few exceptions—just twelve counties—cancer risks from industrial sources rarely exceeded ten per million. Further, in counties subject to the highest cumulative cancer risks (greater than fifty per million), industrial sources accounted for a small percentage of the cancer risks. The four outliers in Figure 7 are each attributable to emissions from steel foundries and mills. This pattern holds in large industrial centers such as Houston (Harris County), in which industrial sources accounted for about ten percent of the cancer risks. By contrast, the counties with the highest risks from industrial sources alone were rural or encompassed small cities.

194. EPA defined a “national risk contributor” as a chemical with an excess cancer risk greater than one per million with an exposed population of at least twenty-five million. *Id.*

195. *Id.*

196. See NATA OVERVIEW, *supra* note 38, at 5.

197. These results are reinforced by recent studies based on direct air-quality monitoring data of specific air toxics. See, e.g., Michael C. McCarthy, Theresa E. O’Brien, Jessica G. Charrier & Hilary R. Hafner, *Characterization of the Chronic Risk and Hazard of Hazardous Air Pollutants in the United States Using Ambient Monitoring Data*, 117 ENVTL. HEALTH PERSP. 790, 790 (2009).



One could reasonably object that averaging emissions at the county level obscures toxic hotspots that occur at smaller spatial scales. Moreover, the importance of this concern is heightened by the variability in the sizes of counties across the country. The census-tract data could thus be more informative and consistent with conventional views about the relative importance of industrial sources to cumulative cancer risks from air toxics.

The census-tract estimates in the 2005 NATA do not bear out this intuition. Among the 100 census tracts with the highest cancer risks from industrial sources,

most were located in small cities or rural counties in states such as Alabama, Ohio, and Pennsylvania, which, consistent with the county-level data, were dominated by toxic emissions from steel mills and foundries.

Table 3: Census Tracts in Which Cancer Risks from Industrial Emissions of Air Toxics Exceed Ten per Million

	<i>Percent Point Sources</i>	<i>Percent Mobile Sources</i>	<i>Percent Secondary Sources</i>
Min	5.19	0.49	6.24
First Quartile	17.18	9.43	24.13
Second Quartile	22.36	15.40	29.41
Third Quartile	30.06	25.36	33.96
Max	87.48	66.09	54.39

The impact of industrial-source emissions on air toxics can also be viewed in terms of their absolute risks. Toxic emissions from industrial facilities in approximately 1180 census tracts, about two percent of tracts nationally, produced cancer risks above ten per million. Of these census tracts, point sources caused cancer risks above twenty per million in 314 tracts and above fifty per million (the national average for all sources) in eighty tracts; the overall average was twenty per million. In relative terms, industrial sources accounted for more than fifty percent of the cumulative risks from all outdoor sources of air toxics in just sixty-five tracts and for more than thirty percent in 297 tracts (see Table 3). To put these numbers in perspective, cancer risks from industrial sources exceeded five per million in 3792 census tracts and one per million (EPA's target risk level) in 23,705 tracts—or about a third of the 65,000 census tracts nationally.

Alternatively, one can evaluate the relative contribution to cancer risks from industrial sources in the census tracts with the highest cumulative risks from all outdoor sources of air toxics. Cumulative cancer risks exceeded 100 per million in 3100 census tracts nationally (see Table 4). A disproportionate share of these census tracts were located in the Los Angeles or New York metropolitan areas,¹⁹⁸ which highlights the strong correlation between heightened risks from air toxics and intense urbanization.

Among the census tracts with the highest cumulative cancer risks, industrial sources were typically a minor factor in relative terms. For about three-quarters of the tracts, industrial emissions accounted for less than three percent of the cumulative cancer risk (see Table 4). These numbers are dramatically lower than most people might predict, and they challenge conventional beliefs about industrial

198. Of the thousand census tracts with the highest cancer risks, 557 are located in the Los Angeles and 342 are in the New York metropolitan areas; together the two cities encompass ninety percent of the top thousand tracts.

emissions of air toxics and their association with cancer risks in the vast majority of U.S. jurisdictions. They also highlight the degree to which the highest cancer risks from air toxics are largely attributable to mobile and non-point sources.

Table 4: Census Tracts in Which Cumulative Cancer Risks from All Outdoor Sources of Air Toxics Exceed 100 per Million

	<i>Percent Point Sources</i>	<i>Percent Mobile Sources</i>	<i>Percent Secondary Sources</i>
Min	0.09	6.24	0.81
First Quartile	1.42	22.44	31.43
Second Quartile	2.01	27.98	36.48
Third Quartile	3.14	33.69	42.14
Max	87.48	56.96	74.70

These results must be interpreted cautiously for three primary reasons. First, the spread in cancer risks over the top one thousand census tracts was small in relative terms—a little more than a factor of two.¹⁹⁹ Yet, an ordinal ranking of census tracts obscures this trend. Second, the significance of these differences is diminished further by the uncertainties in the cancer risk estimates—uncertainties in the data are also about a factor of two.²⁰⁰ Ultimately, this is of secondary importance because the differences in cancer risks between jurisdictions are far less important than the relative contributions of each source category within them. Third, the cancer risk estimates do not include diesel PM,²⁰¹ which is generated largely by mobile sources and is a major, and often the largest, source of cancer risks in urban areas.²⁰²

Notwithstanding these qualifications, the 2005 NATA estimates of cancer risk reinforce the evidence showing that mobile and non-point sources are the primary contributors in most counties and census tracts. This pattern should not obscure the fact that the absolute levels of cancer risks associated with industrial sources can be very high on their own—for the most severely impacted areas, cancer risks substantially higher than twenty per million. Industrial sources of air toxics therefore cannot be overlooked if EPA and the states are to succeed in reducing

199. This fact is reflected in the risk levels for the top-ranked tracts: No. 1—cancer risk of 2.88×10^{-4} ; No. 27—cancer risk of 2.00×10^{-4} ; No. 500—cancer risk of 1.48×10^{-4} ; and No. 1000—cancer risk of 1.27×10^{-4} .

200. NATA OVERVIEW, *supra* note 38, at 71–77.

201. EPA, SUMMARY OF RESULTS, *supra* note 145, at 1.

202. Morello-Frosch & Jesdale, *supra* note 84, at 389; *see also* MATES III, *supra* note 19, at ES-2 to ES-3.

cancer risks below the Agency's regulatory target for cancer risks of one per million.²⁰³

B. The Urban-Rural Divide in Risks from Air Toxics

The highest risks from air toxics are largely an urban phenomenon. The largest cities in the country—Los Angeles, New York, Chicago, and Houston—each encompass numerous census tracts and counties with the highest cancer risks. It was surprising to me that levels of air toxics in Houston are comparable to those of other large cities even though Houston is the fourth largest city by population and has the largest concentration of industrial facilities nationally.²⁰⁴ While the relative importance of emissions from other sources partially explains why Houston is not more of an outlier with respect to cancer risks, it is not the only factor.

It goes without saying that major metropolitan areas in the United States vary greatly in geographic area and total population. The importance of these factors is most pronounced in the New York City area, which does not initially appear to have especially high emissions of air toxics. Table 4 below provides basic information on the geographic size of several counties and their population densities.²⁰⁵ These statistics suggest that density is a critical factor and that it is the primary factor (along with total population) driving poor air quality in the New York City area.²⁰⁶

The 2005 NATA estimates and population data suggest that, once an urban area reaches a sufficient size or density, the number of mobile and non-point sources of air toxics become the primary threats to air quality.²⁰⁷ The geographically compact New York City area, with a population of about 8.18 million in 2010, is an extreme case along both of these dimensions. The same factors drive toxic air pollution in Chicago (Cook County), though to a lesser extent. The significance of these associations is heightened by the fact that about seventy-five percent of air toxics emissions from all sources nationally occur within urban areas.²⁰⁸

203. See Clean Air Act, 42 U.S.C. § 7412(f)(2) (2006).

204. Emissions from industrial sources, in relative or absolute terms, do not place Harris County in the top ten nationally nor was it among the top ten counties based on cumulative cancer risks. Harris County is number forty-three and Jefferson County (Port Arthur) was number twenty-five according to the 2005 NATA estimates of cumulative cancer risks.

205. This variability is much less for U.S. census tracts (especially in urban areas). See *Decennial Management Division Glossary*, CENSUS.GOV, <http://www.census.gov/dmd/www/glossary.html#C> (explaining that a census tract can vary from 1000 to 8000 people but is optimally 4000 people).

206. EPA, *supra* note 39, at 1-2 (“The urban environment is very unique since the combination of high population densities and large concentrations of commercial activity provide the conditions conducive to high exposures and health risks as a result of the emissions of air toxics.”).

207. This is consistent with EPA's association of elevated toxic emissions with “[t]he concentration of activities [that] leads to the presence of multiple emission sources and proportionately higher emissions of multiple [air toxics].” National Air Toxics Program: The Integrated Urban Strategy, Notice, 64 Fed. Reg. 38706, 38711–12 (July 19, 1999).

208. *Id.*

Table 5: County Geographic Scales and Population Densities

<i>County</i>	<i>Geographic Size (miles²)</i>	<i>Population Density (people/mile²)</i>
New York, NY	23	70,951
Los Angeles, CA	4,000	2,066
Cook, IL	946	5,491
Harris, TX	1,729	2,367

The drivers of elevated risks from air toxics nevertheless vary between major urban centers. Los Angeles, for example, does not have an exceptionally high population density. It appears instead that the large geographic scale of the Los Angeles metropolitan area is offset by the meteorological problems created by the mountains to the east of the city (severe temperature inversions can trap air pollution over the city for extended periods of time) and the aggregate population—about thirteen million people in 2010.²⁰⁹

The conditions in Houston lie somewhere between those found in New York and Los Angeles. Houston lacks the population density of New York and the total population of Los Angeles, as well as the latter's more severe meteorological events.²¹⁰ Geography and population together mitigate the impacts of air toxics emissions in Houston and apparently trump its status as the city with the largest concentration of industrial sources nationally. Put differently, once a metropolitan area exceeds a mid-level size and density, emissions of air toxics from industrial sources are overwhelmed by those from mobile and nonpoint sources—when averaged over the spatial scale of a census tract or county.²¹¹

The converse also appears to be true. Namely, that where major industrial sources of air toxics are located in rural or smaller urban areas, they are more likely to drive cumulative cancer risks. This inference is consistent with Houston having a single census tract (ranked thirteenth) among those with the highest *absolute* level of risk from air toxics emitted by industrial sources, and only one among the top

209. See generally James M. Lents & William J. Kelly, *Clearing the Air in Los Angeles*, SCI. AM., Oct. 1993, at 32, 33 (describing the phenomena of temperature inversions and the impacts on air quality in Los Angeles).

210. While the population density of Houston is close to that of Los Angeles, its aggregate population in 2010 was only about 5.95 million. Similarly, although not as severe as the inversions in Los Angeles, stationary air conditions in Houston can trap pollutants over the city for extended periods of time. See R.M. Banta, C.J. Senff, J. Nielsen-Gammon, L.S. Darby, T.B. Ryerson, R.J. Alvarez, S.P. Sandberg, E.J. Williams & M. Trainer, *A Bad Air Day in Houston*, BULL. AM. METEOROLOGICAL SOC'Y 657, 658–60 (2005); Lents & Kelly, *supra* note 209, at 33.

211. The large population and size of Houston are associated with large numbers of vehicles and high per capita driving patterns—Houstonians collectively are estimated to average more than 140 million miles per day. MAYOR'S TASK FORCE ON THE HEALTH EFFECTS OF AIR POLLUTION, A CLOSER LOOK AT AIR POLLUTION IN HOUSTON: IDENTIFYING PRIORITY HEALTH RISKS 8–9 (2006), available at <http://www.greenhoustontx.gov/reports/UTreport.pdf>.

one hundred census tracts (ranked forty) with the highest *percentage* of cancer risks attributable to industrial sources.²¹² The pattern is bolstered further by the number of census tracts and counties, largely in rural areas or encompassing small cities, in which the risks from point sources were greater than those in Houston.

Together the preceding analyses reveal several general patterns in industrial emissions of air toxics. While industrial emissions are minor contributors to emissions of and risks from air toxics in most jurisdictions, they are much more likely to account for a significant fraction of toxic emissions in rural or small, urban environments. The jurisdictions with the highest relative impacts from industrial emissions are also surprisingly uniform—most of the cancer risks are from highly toxic compounds emitted by steel plants. The strength of these patterns ought to make it straightforward to identify industrial hotspots and their small numbers ought to simplify oversight and monitoring.

IV. REASSESSING THE DEBATE OVER POLLUTION TRADING PROGRAMS AND TOXICS HOTSPOTS

The EPA data provide a global picture of the major sources of air toxics and their geographic distribution. The small relative contributions of industrial facilities and dominance of nonpoint and mobile sources are evident in all of the empirical studies. These patterns recur whether one evaluates source contributions in terms of emissions levels or cancer risks, and they persist at geographic scales ranging from census tracts to the nation as a whole.

The geographic clustering of major industrial sources in the southeast and a few midwestern and northeastern states is also clearly evident in the emissions and risk data. The degree to which industrial sources of air toxics are concentrated in discrete areas is a striking result, but the number of facilities along the Gulf coast in Texas and Louisiana is truly exceptional. Yet, even in these areas mobile and nonpoint sources largely dominate. The Houston data are exemplary in this respect—nonpoint and mobile sources are dominant in the largest industrial hotspot nationally. This extreme result highlights the degree to which toxic air pollution is a collective problem for which we are all responsible.

An important implication of these findings is that toxic air pollution generated by many small sources may be less likely to concentrate in certain neighborhoods. Localized concentrations of small sources (such as congested highways), however, can be and are associated with urban hotspots.²¹³ Further, while disparities do and will continue to exist, the data suggest that the contributing sources will vary substantially by jurisdiction. Above all, the EPA data show that reductions in emissions from small businesses, the transportation sector, and the public will be critical to lowering levels of toxic air pollutants, which by any measure remain high in urban areas where more than eighty percent of the U.S. population lives.²¹⁴

212. The highest cancer risk census tracts were localized around the intensely industrialized area around the ship channel, particularly the East Houston, Channel View, and San Jacinto neighborhoods.

213. See *supra* notes 19 & 42.

214. PAUL MACKUN & STEVEN WILSON, POPULATION DISTRIBUTION AND CHANGE: 2000 TO 2010, at 4 (2011), available at <http://www.census.gov/prod/cen2010/briefs/c2010br->

The sections that follow address the regulatory and policy implications of the preceding analysis for GHG trading regimes. Subparts A and B examine the potential for GHG trading regimes to create or exacerbate toxic hotspots around industrial sources under the California climate change legislation and an anticipated national program in the future. Subpart C discusses the legal mechanisms available to eliminate the potential for GHG trading-induced hotspots to occur, as well as any associated environmental inequities.

A. Reevaluating Environmental Justice Objections to the GHG Trading System Proposed in California

A GHG trading regime will not cause significant inequities in exposures to air toxics unless certain factors align. Beyond facility owners choosing to purchase permits over reducing GHG emissions, three primary conditions must be met: (1) industrial facilities must be concentrated in poor communities of color, (2) industrial sources of GHG emissions must account for a significant fraction of aggregate toxic air emissions, and (3) emissions of toxic co-pollutants must be correlated with GHG emissions. If these conditions are met, a GHG trading regime could allow industrial emissions of toxic co-pollutants to remain roughly the same in minority communities while emissions decline elsewhere. Under these circumstances, the net effect of the regulatory flexibility afforded by a GHG trading program could heighten environmental inequities between poor communities of color and other neighborhoods.

The critical factors for California's GHG trading system are the geographic distribution of industrial sources and their relative contributions to cancer risks from air toxics. The studies discussed in Part I found that industrial sources of air toxics were more likely to be located in minority communities and that they disproportionately impacted them. A 2006 estimate found that, on average, racial disparities from exposures to industrial emissions of air toxics were about fifteen percent.²¹⁵

Measures of racial disparities from industrial facilities alone omit the impacts from mobile and non-point sources of toxic air pollutants. On average, industrial sources accounted for about thirteen percent of the NATA Toxics emitted statewide in 2005 and for about two percent of the cumulative cancer risks. Furthermore, industrial sources typically had a modest impact on levels of air toxics even in the most industrialized counties and census tracts.²¹⁶ For the state as a whole, the extreme scenario of a twenty percent disparity from industrial sources (i.e., no

01.pdf (stating that almost eighty-four percent of the U.S. population lives in urban metropolitan areas).

215. ASH ET AL., *supra* note 67, at 6.

216. In all but a handful of census tracts, industrial emissions accounted for less than fifteen percent of the cumulative cancer risks from air toxics, and in no case more than twenty percent. Industry generated less than ten percent of NATA Toxics emitted in all but one county in California. For example, Contra Costa County has a significant number of the largest industrial sources of toxic emissions, including petroleum refineries, and industrial sources accounted for fifteen percent of toxic emissions countywide.

direct GHG reductions occurring in minority communities) would translate to roughly a 0.4% disparity in cancer risks for all sources of air toxics.²¹⁷

These figures represent upper bounds on the relative impacts of industrial sources in California. EPA's NATA does not include cancer risks from diesel PM, which in many urban areas is the single largest source of cancer risk. For example, in Los Angeles, which contains a large number of both mobile and stationary sources, mobile sources accounted for ninety-four percent of the 1200 per million average cancer risk in 2005.²¹⁸ This estimate is more than ten times higher than the 109 per million cancer risk reported in the 2005 NATA for Los Angeles County. The divergence between the two estimates is explained almost entirely by the absence of diesel PM in the NATA estimates,²¹⁹ which on average accounted for eighty-four percent of the cancer risk from air toxics in the Los Angeles air basin.²²⁰

Diesel PM is generated overwhelmingly by mobile sources, a fact reflected in the statistic that stationary sources (non-point and industrial sources) on average accounted for six percent of the cancer risks from air toxics in Los Angeles.²²¹ The more complete data suggest that industrial sources in metropolitan areas—where the largest industrial sources are located in California—are likely to account for less than two percent of the cancer risks from air toxics at either the county or census tract level. This evidence is bolstered by estimates of the cancer risks from individual industrial facilities in California, which in all but a handful of cases were below ten per million.²²²

The small contributions of industrial facilities to aggregate emissions of air toxics limits their capacity to cause significant racial disparities at the census tract and county levels. The reasoning is simple: if industrial sources account for roughly ten percent of emissions, racial disparities from industrial sources would on average be diluted at least twentyfold. Moreover, this analysis assumes that emissions of air toxics and GHGs would decline at the same rate, which is an absolute upper bound on the association between GHG and air toxics emissions.²²³

Direct estimates of correlations between GHGs and toxic air emissions do not exist to my knowledge. The TRI data on toxic air emissions do, however, provide separate values for stack and fugitive emissions (e.g., from storage tanks, pipes, valves), which can serve as a rough proxy for emissions that are unlikely to be correlated with GHG emissions. Fugitive emissions from petroleum refineries, which in California have the greatest potential to impact minority communities, on

217. This is consistent with California's twenty percent target for reductions in GHG emissions by 2020. *See supra* note 20.

218. MATES III, *supra* note 19, at ES-2 to ES-3.

219. EPA, SUMMARY OF RESULTS, *supra* note 145.

220. MATES III, *supra* note 19, at ES-2 to ES-3.

221. *Id.*

222. The data are available through a database maintained by the California Air Resources Board (CARB). *See* CARB, *Facility Search Engine*, CA.GOV, <http://www.arb.ca.gov/app/emsinv/facinfo/facinfo.php>. The cancer risk estimates for petroleum refineries in California ranged from roughly 2.7 to 9.6 per million, which was more than two orders of magnitude lower than the average cancer risk in the Los Angeles air basin. *Id.*

223. *See infra* Part IV.B.

average accounted for about forty-nine percent of the OSHA Carcinogens they emitted. While this is an admittedly crude proxy, it suggests that percentage reductions in GHG emissions are likely to be associated with substantially smaller percentage reductions—close to a factor of two—in emissions of air toxics.

The demographic data provided in *Minding the Climate Gap* enable a more refined analysis of the potential impacts from industrial facilities on minority communities.²²⁴ Recall that the *Minding the Climate Gap* report evaluated the potential impacts of reducing GHG emissions from about 150 of the largest GHG sources in California. One of the report's most significant results was that the projected racial disparities were driven almost entirely by a subset of ten major facilities, most of which were oil refineries located in Los Angeles and the San Francisco Bay Area.²²⁵ The emissions from these facilities consequently encompass most of the potential for environmental disparities to arise under a GHG trading system.

Using the EPA emissions data, it is possible to calculate the share of outdoor toxic air emissions for each county in which the top ten MCG facilities are located, as well as their share of the cumulative cancer risks at the county and census-tract levels. The top ten *Minding the Climate Gap* facilities accounted for less than five percent of county-level emissions of NATA Toxics and less than twenty percent of PM emissions (excluding diesel PM).²²⁶ The estimated cancer risks from these facilities, which ranged from 2.7 to 9.6 per million, were no more than one percent of the average cumulative cancer risk of 951 per million in Los Angeles.²²⁷ Under plausible scenarios for the near-term, given the twenty percent reduction target for GHG emissions, industrial emissions appear to be too low for a GHG trading regime to cause material disparities in aggregate toxic air emissions at the county or census tract levels.

The EPA air toxics data suggest strongly that a GHG trading regime would not increase disparities in exposures to toxic emissions in California and that the potential for racial or income-based inequities is equally slight. The small relative contributions of industrial sources to aggregate levels of air toxics throughout California are central to this conclusion. Confidence in these findings is bolstered by their focus on relative impacts across different source categories, as these comparisons avoid the large uncertainties associated with direct estimates of risk and are believed to be among the robust uses of the EPA data.²²⁸

224. See *infra* Part IV.B.

225. PASTOR ET AL., *supra* note 73, at 18–19 (eight refineries accounted for ninety-three percent of the inequities in the pollution burden from PM emissions).

226. In Los Angeles County, petroleum refineries accounted for about 0.33% of the NATA Toxics emitted in 2005 and about five percent of the nondiesel PM; in the Contra Costa County (the county in the Bay Area where most of the petroleum refineries are located), they accounted for about four percent of the NATA Toxics emitted and eighteen percent of the nondiesel PM.

227. MATES III, *supra* note 19, at ES-2 (these estimates include the cancer risk associated with diesel PM and are thus substantially higher than the 2005 NATA estimates for Los Angeles).

228. See, e.g., David E. Adelman, *The False Promise of the Genomics Revolution for Environmental Law*, 29 HARV. ENVTL. L. REV. 117, 122–24 (2005).

B. Likelihood of a National GHG Trading Program Exacerbating Industrial Hotspots

Unlike California, jurisdictions elsewhere in the country do exist in which industrial sources account for a large fraction of total toxic air emissions, cancer risks, or both. The potential for disparate impacts in these jurisdictions cannot be foreclosed. While unnecessary due to the low industrial emissions in California, this broader analysis must assess the likelihood that such facilities will rely disproportionately on purchasing permits (over reducing GHG emissions) and the degree to which GHG emissions are correlated with toxic emissions.

For purposes of this discussion, I will define “industrial hotspot” using separate metrics for counties and census tracts to account for the much larger areas over which emissions are averaged in the county-level data. For counties, the only requirement is that the cancer risks from industrial facilities exceed ten per million; for census tracts, industrial emissions must produce cancer risks of at least twenty per million and account for a minimum of thirty percent of the cumulative cancer risk across a given tract.²²⁹ The latter cancer risk cutoff is intended to be conservative—it is a factor of twenty above EPA’s target risk level and a factor of five below the cancer risk EPA deems to be clearly unacceptable;²³⁰ it is also less than half of the fifty-per-million national average for cancer risks from air toxics.²³¹

1. The Infrequency of Industrial Hotspots Nationally

Twelve counties and 240 census tracts nationally satisfy my definition of an industrial hotspot. Somewhat surprisingly, none of the counties is in either Texas or Louisiana, which are the two states most commonly associated with toxic emissions from large industrial facilities. They are instead spread over ten states and encompass just four cities with populations greater than two hundred thousand (only one of which, Pittsburgh, is over one million). Within this group, industrial sources accounted for more than fifty percent of the cumulative cancer risks in only one county and for more than thirty percent in just half of them; the average for all twelve counties was thirty-one percent.

The 240 census tracts with industrial hotspots were spread across seventy-three counties located in twenty-six states, but were most prevalent in Pennsylvania (seventy-one), Ohio (twenty-five), Indiana (twenty-three), Kansas (fifteen), Texas

229. Among the census tracts in which point sources produced cancer risks greater than twenty per million but accounted for less than thirty percent of the cumulative risk, the average point-source cancer risk was twenty-four per million and the maximum was thirty-eight per million (five tracts exceeded thirty per million).

230. See *supra* note 91 and accompanying text. In California, regulatory action is also triggered when a facility’s cancer risk exceeds ten per million. S. COAST AIR QUALITY MGMT. DIST., ANNUAL REPORT ON AB 2588 AIR TOXICS “HOT SPOTS” PROGRAM 2–3 (2011) available at http://www.aqmd.gov/prdas/ab2588/pdf/Annual_Report_2010.pdf.

231. EPA, SUMMARY OF RESULTS, *supra* note 145, at 5. This definition provides a rough margin of error beyond the putative factor-of-two uncertainty in EPA’s risk estimates, as the cancer risks from industrial sources not located in hotspots would still fall substantially below the national average if EPA’s estimates were off by a factor of two.

(fifteen), and Alabama (thirteen). Industrial hotspots were closely associated with steel mills and foundries, about 200 tracts or eighty percent of the total; the primary pollutants were chromium (ninety-nine tracts) or coke oven emissions (ninety-six tracts).²³² Among the sixty-five census tracts in which point sources accounted for more than half of the cumulative cancer risk, only one in Houston, Texas, and another in Lincoln, Nebraska, were associated with other types of emissions.²³³ Further, all but the census tracts in Pittsburgh, Birmingham, and Houston, which each were outliers with respect to the volume or toxicity of their industrial emissions,²³⁴ were located in rural communities or midsized cities with low population densities.

Industrial hotspots were closely associated with highly toxic industrial emissions and low—relative to major urban areas—emissions from mobile and nonpoint sources. Industrial sources rarely dominated emissions from the other source categories apart from these exceptional circumstances. Setting aside the distinctive conditions in Birmingham, Houston, and Pittsburgh, this phenomenon effectively forecloses industrial hotspots in large urban areas.

The demographics of the census tracts with industrial hotspots are notable because they were bimodal. This pattern follows from the split between the geographic centers for steel production in the southern states and those in and around Pittsburg. In 2005, the demographics of communities with steel mills were on average twenty-four percent minority, whereas the percentage for iron and steel foundries was forty-one percent; both were located in communities in which seventeen percent of the population was low-income.²³⁵ By comparison, minorities made up thirty-two percent of the U.S. population and low-income individuals accounted for thirteen percent.²³⁶ Minorities were thus overrepresented in communities with iron and steel foundries but underrepresented in communities with steel mills, whereas the low-income percentages were close to the national averages in both cases.²³⁷

The disparities for minority populations living around steel mills and foundries cut both ways.²³⁸ If an average is calculated for steel mills and foundries

232. Both chromium and coke oven emissions have high relative toxicities. Adel M. Zayed & Norman Terry, *Chromium in the Environment: Factors Affecting Biological Remediation*, 249 PLANT & SOIL 139, 141 (2003) (describing chromium VI as an “extremely toxic” compound).

233. In Houston, it was benzene and butadiene from refineries and chemical plants, but in Lincoln it was naphthalene from commercial boilers. The 2005 NATA data do not identify the chemical compounds responsible for the high risks from industrial sources in the other seven census tracts.

234. Houston was an outlier as we have seen with regard to the volume of its industrial emissions; Pittsburgh and Birmingham were outliers with respect to the toxicity of emissions from local steel mills and foundries.

235. ASH ET AL., *supra* note 67, at 6–8.

236. *Id.*

237. Although the study does not provide this number explicitly, the minority average for all industries was approximately fifteen percent. *See id.*

238. Basic statistics tells us that the variance in racial composition will increase as the geographic scale is decreased. Accordingly, the observed siting patterns could be consistent with a random distribution of facilities.

collectively, the value for the minority share of toxic exposures is thirty-two percent,²³⁹ which is identical to the minority share of the population nationally. If the results are disaggregated, however, one could conclude that iron and steel foundries disproportionately impact minority communities. In the analysis below I evaluate them together, but the data can be presented in a variety of ways so long as one is clear about the rationale for their approach.

However one aggregates the data, they suggest that the potential is low for a national GHG trading program to cause racial or income-based inequities in exposures from air toxics and that if hotspots were to materialize they would be limited to a small number of counties and census tracts. The data also show that large populations and high population densities all but foreclose the emergence of industrial hotspots in metropolitan areas where the cancer risks from air toxics are typically the highest.

2. Evaluating the Heightened Risk of Inequities in Industrial Hotspots

Conditions across the country are more variable than those in California with respect to local demographics and the impacts of toxic emissions from industrial facilities. Nevertheless, the racial disparities nationally attributable to industrial emissions of air toxics, about three percent for all industries, were lower than for California according to *Justice in the Air*.²⁴⁰ This is to be expected given the much larger geographic area over which inequities are being averaged, but the country-wide average reinforces other indications that a national GHG trading program is unlikely to cause systematic racial inequities.

This national average obscures local variations in toxic air emissions. Among the sixty-five census tracts with the highest relative risks from air toxics, industry accounted on average for sixty percent of the cumulative cancer risks. In these tracts, disparities in industrial emissions would be discounted forty percent on average, rather than the factor of ten typical in California. As a consequence, racial disparities would be much less likely to be obscured by emissions from other sources. Further, the high proportion of emissions from industrial sources elevates the significance of errors in EPA risk estimates as they too are less likely to be obscured.²⁴¹ These factors lead to a straightforward inference: GHG trading-induced hotspots will occur, if at all, in jurisdictions defined here as industrial hotspots.

239. I calculated a weighted average of the two numbers using the toxic scores for “Steel Works, Blast Furnaces, Rolling and Finishing Mill” (SIC code 331)—toxic score of 1054, and “Iron and Steel Foundries” (SIC code 332)—toxic score of 939. ASH ET AL., *supra* note 67, at 6–8.

240. *Id.* at 10–11 (disclosing that the share of health impacts from emissions for air toxics by all industrial sectors was 34.8% compared to their 31.8% share of the population). Table 3 discloses that the racial share for the ten industrial sectors with the highest levels of inequities was 37.3, which leads to a disparity on average of 5.5%. *Id.* In this discussion, I will focus on racial inequities given the weak evidence for income disparities discussed in Part I.

241. *See supra* notes 121–25.

The analysis is simplified by the modest number of industrial hotspots nationally, 240 census tracts in all, and the small number of industries implicated. As noted above, steel mills and foundries were associated with eighty percent of the industrial hotspots at the census-tract level, the most notable exception being the Houston petrochemical complex.²⁴² Refineries and steel plants also have among the highest GHG emissions of any industrial sector,²⁴³ and because of this would be an important (and early) target for any national GHG trading system.²⁴⁴

The presence of an industrial hotspot does not automatically lead to racial inequities. The demographics of the communities surrounding a facility may or may not be disproportionately minority groups. In the present context, the racial demographics around petroleum refineries were extreme (fifty percent minority),²⁴⁵ while the demographics around steel production facilities as noted above were mixed.²⁴⁶ One's view of potential disparities in risk exposures will rest on how the data are aggregated. If the impacts of steel mills and foundries on surrounding populations are evaluated together, the minority share of the risks from exposures to their emissions would match the minority share of the U.S. population and thus preclude racial inequities.²⁴⁷

The central question being addressed here is whether a national GHG trading program would cause or exacerbate racial disparities in exposures to toxic air pollutants. The natural geographic scale on which to evaluate such inequities is therefore the national level. The data from *Justice in the Air* also can be used to derive the minority share of the risks (cancer and noncancer) from industrial hotspots nationally. Since petroleum refineries and steel production facilities account for most of the industrial hotspots in the United States, I will use a

242. The number of industrial hotspots may be declining as a consequence of more stringent standards for steel mills and foundries that EPA issued in 2005 and 2007. *See, e.g.*, National Emission Standards for Hazardous Air Pollutants for Iron and Steel Foundries, 73 Fed. Reg. 7210 (Feb. 7, 2008) (codified at 40 C.F.R. pt. 63); National Emission Standards for Hazardous Air Pollutants for Coke Ovens: Pushing, Quenching, and Battery Stacks, 70 Fed. Reg. 44,285 (Aug. 2, 2005) (codified at 40 C.F.R. pt. 63). Emissions of chromium compounds from steel plants in Alabama, Indiana, Ohio, and Pennsylvania (the states with the bulk of the industrial hotspots), collectively declined fifty percent between 2005 and 2010.

243. *See* EPA, *supra* note 172, at ES-5 (disclosing that steel production was ranked second only to electric utilities and that petroleum production was ranked twelfth among all sources and second among industrial sources).

244. They are already being targeted under EPA's new GHG regulations under the Clean Air Act. *See* Mandatory Reporting of Greenhouse Gases, 74 Fed. Reg. 56,260 (Oct. 30, 2009) (codified as amended in scattered sections of 40 C.F.R.); EPA, SETTLEMENT AGREEMENTS TO ADDRESS GREENHOUSE GAS EMISSIONS FROM ELECTRIC GENERATING UNITS AND REFINERIES: FACT SHEET 1-2 (2010), available at <http://epa.gov/carbonpollutionstandard/pdfs/settlementfactsheet.pdf>.

245. ASH ET AL., *supra* note 67, at 14. The racial demographics around the Exxon Baytown Refinery in Houston were slightly higher, fifty-four percent minority. *Id.* at 12.

246. *Id.* at 10.

247. The minority share for steel mills and iron and steel foundries was twenty-four and forty-one percent, respectively. *Id.* If the two numbers are averaged by weighting them according to their respective toxic score, collectively the minority share is thirty-two percent, which is essentially identical to the minority share of the U.S. population. *See id.* at 10-11.

weighted average of the human health risks for these industries. The minority share of the health risks in 2006 was 33.3 percent for these industries, which represents a racial disparity of 1.5 percent.²⁴⁸ The roughly proportional racial demographics of industrial hotspots effectively preclude a national GHG trading program from causing racial disparities.

As a practical matter, disagreements about the geographic area over which inequities are evaluated are probably secondary. Other factors are likely to preclude a national GHG trading program from exacerbating environmental inequities. First, similar to California, the targets for reducing GHG emissions are likely to be modest in the near term (about twenty percent over the first decade or so).²⁴⁹ Reduction targets for GHG emissions create an absolute upper bound on the potential disparities of industrial emissions, which are then discounted forty percent to account for the average relative contributions of industrial sources in hotspots. This results in an upper bound for racial disparities among the highest risk jurisdiction of twelve percent or, in absolute terms, an increase in cancer risk on average of approximately eight per million for the top sixty-five census tracts or three per million for all industrial hotspots.²⁵⁰

Second, a GHG trading program would operate in parallel with existing regulations under the CAA, which impose technology-based and residual-risk standards on industrial sources of air toxics.²⁵¹ These regulatory programs establish basic limits on emissions from petroleum refineries and steel production facilities, and thus would bound the impacts of GHG trading on emissions of toxic co-pollutants and potential disparities between jurisdictions. This would not eliminate disparities, but recent declines in emissions—including those from steel plants—suggest that they could be substantially dampened.²⁵²

Third, disparities will be limited because the techniques used to reduce GHG emissions often will not affect emissions of air toxics. In some cases, toxic emissions will not be associated with the processes (mostly combustion) that generate GHG emissions.²⁵³ Indeed, a significant fraction of the GHG emissions

248. *See id.* (the minority share of the U.S. population was 31.8 percent). This estimate is averaged over all petroleum refineries, even though Houston is the only jurisdiction in which petroleum refineries were associated with an industrial hotspot. This weighted the result towards the higher racial disparities found around refineries.

249. The target in California is a twenty percent reduction in GHG emissions below 1990 levels by 2020. *See supra* note 20. The failed congressional climate change legislation, the American Clean Energy and Security Act, contained a weaker target of seventeen percent below 2005 GHG emissions levels by 2020; its long-term target was an eighty-three percent reduction below 2005 by 2050. John M. Broder, *Ambitious Energy and Climate Bill Clears a House Hurdle, But Others Remain*, N.Y. TIMES, May 22, 2009, at A13.

250. These estimates are based on the average cancer risks associated with industrial sources in industrial hotspots; I have simply multiplied the average cancer risk by percentage disparity.

251. DOREMUS ET AL., *supra* note 175, at 611–19.

252. *See* ASH ET AL., *supra* note 67 at 10–12.

253. This is true for both steel plants and petroleum refineries. *See* EPA, AVAILABLE AND EMERGING TECHNOLOGIES FOR REDUCING GREENHOUSE GAS EMISSIONS FROM THE IRON AND STEEL INDUSTRY 3 (2010) [hereinafter EMERGING TECHNOLOGIES STEEL]; EPA, AVAILABLE AND EMERGING TECHNOLOGIES FOR REDUCING GREENHOUSE GAS EMISSIONS

attributable to steel production derive from offsite electric power production,²⁵⁴ which is by definition spatially separated and decoupled from onsite toxics emissions. Similarly, use of energy-efficient measures to reduce GHG emissions from combustion processes will often impact only a portion of the emissions at a facility. Many fugitive emissions, as described above, are not caused by combustion processes and on average they accounted for more than forty percent of the toxic emissions at steel plants and petroleum refineries in 2010.²⁵⁵

Fourth, exploiting opportunities for increasing the energy efficiency of industrial operations is a, if not the, primary means by which GHG emissions are likely to be reduced over the near term.²⁵⁶ The EPA believes that this may be especially true in energy-intensive sectors, such as the steel and petroleum industries, where most GHG emissions are generated by onsite combustion of fossil fuels or offsite generation of electricity.²⁵⁷

The favorable economics suggest that steel and petroleum industries will often choose to reduce GHG emissions over purchasing credits.²⁵⁸ The EPA has identified numerous options for reducing GHG emissions with payback periods of less than five years.²⁵⁹ This situation differs radically from other areas where the only options for reducing emissions are fuel switching or costly emissions-control technologies.²⁶⁰ The option of enhancing energy efficiency creates a choice between making annual payments for GHG credits and investing in cost-benefit

FROM THE PETROLEUM REFINING INDUSTRY 3 (2010) [hereinafter EMERGING TECHNOLOGIES REFINERIES].

254. EMERGING TECHNOLOGIES STEEL, *supra* note 253, at 1.

255. On average, fugitive emissions in 2010 accounted for about forty percent of OSHA Carcinogens from steel plants in states with industrial hotspots (the range was roughly from about twenty to sixty percent). For the petroleum refineries in Houston, fugitive emissions in 2010 on average also accounted for about forty percent of OSHA Carcinogens emitted (the range was roughly from fifteen to sixty-five percent).

256. EMERGING TECHNOLOGIES STEEL, *supra* note 253, at 9–12 (focusing almost exclusively on energy efficiency measures to reduce GHG emissions including tables detailing the options); EMERGING TECHNOLOGIES REFINERIES, *supra* note 253, at 4 (stating that “a primary option to reduce CO₂ emissions is to improve energy efficiency” at petroleum refineries).

257. In the steel industry, direct combustion and electricity generation account for about eighty-five to ninety percent of the industry’s GHG emissions. *See* EMERGING TECHNOLOGIES STEEL, *supra* note 253, at 3. Combustion sources alone account for sixty-three percent of GHG emissions at oil refineries; this rises to eighty-seven percent if catalytic regeneration processes, which also involve combustion, are included. EMERGING TECHNOLOGIES REFINERIES, *supra* note 253, at 3.

258. In other pollution trading programs, the larger and dirtier facilities have often reduced their emissions the most dramatically. DALLAS BURTRAW & SARAH JO SZABELAN, U.S. EMISSIONS TRADING MARKETS FOR SO₂ AND NO_x 8 (2009) (describing studies showing that under the U.S. SO₂ trading market, “[t]he greatest reductions in emissions by far (in tonnage and percentage) were in the Midwest, the area with the greatest power plant emissions historically”), available at <http://www.rff.org/RFF/Documents/RFF-DP-09-40.pdf>.

259. *See* EMERGING TECHNOLOGIES REFINERIES, *supra* note 253, at 11–16; EMERGING TECHNOLOGIES STEEL, *supra* note 253, at 9–12.

260. BURTRAW & SZABELAN, *supra* note 258, at 11–14 (discussing the fuel-switching and emissions controls available under the SO₂ trading program).

justified technologies with substantial short- and long-term benefits. If capital costs are reasonable, as the EPA projects in many cases, facility owners ought to have a significant incentive to reduce GHG emissions,²⁶¹ and inequities would not arise because purchasing GHG credits would not be preferred over reducing GHG emissions.

The availability of cost-effective options for reducing GHG emissions could, of course, change as the cap on GHG emissions is reduced over time. The impact of more stringent caps on pollution trading and technological options is difficult to predict. Some emerging technologies for reducing GHG emissions could reduce toxic emissions dramatically,²⁶² but the economics of new technologies and their effects on co-pollutants will clearly vary across industries and by technology. New technologies could also be used to mitigate the emergence of hotspots by enabling more effective oversight of a GHG trading program, such as through new technologies that enhance the accuracy and reduce the costs of emissions monitoring. These uncertainties are substantial and are grounds for caution going forward, but they may not be a major factor in the near term given the emphasis on energy efficiency.

Taken together the EPA data and existing scientific studies reveal that a national GHG trading regime would not exacerbate environmental inequities in most counties and census tracts. Similar to California, the small contributions of industrial facilities to aggregate emissions of air toxics would in most cases effectively foreclose such inequities. Of equal importance, among the census tracts in which industrial emissions dominate, the local demographics mirror those of the country as a whole and thus could not cause significant racial inequities in aggregate.

Regardless of how the racial issues are evaluated, the potential for a national GHG trading program to exacerbate disparities in the emissions of air toxics between industrial hotspots and other jurisdictions cannot be ignored. In the near term, the modest targets anticipated for a national GHG trading regime would limit the magnitude of any potential disparities, and this backstop would be reinforced by direct regulation of toxic emissions under the CAA. These limits are in turn likely to be augmented by the weak correlation between GHG and toxic emissions and favorable economics for reducing GHG emissions.

261. Other reasons may exist that override the favorable economics—perhaps the ones keeping companies from investing in energy efficiency now—but a GHG trading regime would make energy-efficiency options more attractive and potentially cheaper if new technologies are developed or new knowledge enhances the effectiveness of existing ones or reduces their costs. Adam B. Jaffe, Richard G. Newell & Robert N. Stavins, *A Tale of Two Market Failures: Technology and Environmental Policy*, 54 *ECOLOGICAL ECON.* 164, 165–66 (2005) (describing how environmental regulations, including pollution trading, can stimulate technological innovation and adoption, but also detailing other market failures that can create barriers to innovation). EPA technical reports suggest that a wide range of options are potentially available. See *EMERGING TECHNOLOGIES REFINERIES*, *supra* note 253, at 11–16; *EMERGING TECHNOLOGIES STEEL*, *supra* note 253, at 9–12.

262. See, e.g., *EMERGING TECHNOLOGIES STEEL*, *supra* note 253, at 40 (describing an emerging technology that would reduce emissions from iron and steel foundries—volatile organic compounds by eighty-seven percent and mercury by fifty-eight percent).

It is my hope that the EPA data and preceding analysis will assuage concerns that toxic hotspots will be an unavoidable and substantial byproduct of implementing a national GHG trading regime. More broadly, I hope that this work will lower health-equity concerns about market-based regulations generally—including taxes. In the long term, conditions will change and emissions caps will be lowered, either of which could introduce new factors that qualify my analysis. For these and other reasons, the public is likely to demand that additional measures be taken to ensure that communities located in industrial hotspots are adequately protected.

3. Preventing Inequities Without Sacrificing the Efficiency of a National GHG Trading Regime

The preceding analysis has shown that inequities cannot be foreclosed in jurisdictions for which industrial sources contribute significantly to toxic emissions. Further, risk estimates in these jurisdictions will be more sensitive to the acknowledged uncertainties in the EPA data, which could fuel skepticism about the metrics—particularly cancer risks—upon which the preceding analysis is based. The remaining uncertainties, as well as the prospect of more aggressive measures in the future, are likely to prompt calls for additional legal protections to safeguard potentially vulnerable communities.

Adapting pollution trading regimes to prevent the emergence of toxic hotspots has been contentious.²⁶³ Proponents of market-based regulations worry that mechanisms for addressing inequities will sacrifice the efficiency of pollution markets by either increasing the costs of transactions or placing restrictions on the trades that can occur (for example, geographic limits).²⁶⁴ The debate then becomes one of balancing the efficiency of pollution markets against distributional concerns about environmental inequities,²⁶⁵ although few studies have attempted to assess the potential for pollution trading regimes to generate significant environmental inequities.²⁶⁶

263. Nash & Revesz, *supra* note 1, at 572 (stating the proposals to mitigate hotspots have had “significant drawbacks, either providing only an incomplete solution to the problem or introducing complexity that could stand in the way of the efficient functioning of the market”); Alan J. Krupnick, Wallace E. Oates & Eric Van De Verg, *On Marketable Air-Pollution Permits: The Case for a System of Pollution Offsets*, 10 J. ENVTL. ECON. & MGMT. 233, 242–43 (1983) (discussing the tradeoffs between efficiency and spatial disaggregation of a market to accommodate geographic and other variables in the area covered by a trading program).

264. See, e.g., EPA, TOOLS OF THE TRADE: A GUIDE TO DESIGNING AND OPERATING A CAP AND TRADE PROGRAM FOR POLLUTION CONTROL 3–22 (2003) (discussing the tradeoffs between efficiency and distribution concerns).

265. See *supra* Part I.C.

266. See PASTOR ET AL., *supra* note 73, at 21–22 (finding that a GHG trading program in California could cause hotspots around certain facilities that result in environmental inequities); SCHATZKI & STAVINS, *supra* note 13, at 15–18 (using air pollution data for Los Angeles to argue that hotspots are unlikely to arise from a GHG trading program in California); Ringquist, *supra* note 5, at 301–02, 321–22 (describing the handful of existing studies that exist; finding a negative correlation between the minority status of local

The geographically discrete nature of industrial hotspots described above suggests that a targeted strategy for mitigating potential inequities ought to be feasible. This approach would have two obvious benefits. First, it would avoid the added costs of imposing additional measures on the entire system, which for a national market would be considerable. Second, the small number of jurisdictions involved would enable refinement of legal mechanisms to the circumstances in each jurisdiction without greatly impacting the administrative costs of the program.

One can gain a sense of the relative size of the potential impacts on a national GHG trading system through the inventories of GHG emissions for steel plants and petroleum refineries. In 2009, the two industries collectively accounted for about five percent of the GHGs emitted by industrial sources in the United States and less than one percent of total GHG emissions.²⁶⁷ The percentages for the subset of facilities located in industrial hotspots would be a fraction of these percentages. At this scale, market interventions targeted to specific industrial hotspots would not materially affect the efficiency of a GHG trading program.

I will not attempt to describe the details of how a national GHG trading program could be modified in this Article. My central point is simply that the tradeoff between equity and efficiency is largely neutralized by the infrequency of industrial hotspots nationally. In addition, a large literature exists on modifying pollution trading regimes to prevent the emergence of toxic hotspots,²⁶⁸ and there is every reason to believe that these well-developed policies would work well under circumstances where hotspots are rare. In the analysis that follows, I assume that targeted policies would apply only to sources with large emissions of both air toxics and GHGs that are located in jurisdictions with industrial hotspots.²⁶⁹

To illustrate the feasibility of mitigating toxic hotspots, I will briefly describe three prominent mechanisms for modifying pollution trading regimes in the literature. The examples are (1) heightened monitoring and informational requirements for trades, (2) geographic restrictions on trading (often referred to as

communities and the likelihood that a facility would purchase emissions credits in the SO₂ trading program).

267. Collectively, steel production facilities accounted for less than one percent of total U.S. GHG emissions in 2009 and about five percent of industrial emissions of GHGs. EPA, *supra* note 172, at ES-5. Petroleum refineries collectively accounted for less than a tenth of a percent of GHG emissions nationally and less than half a percent of industrial emissions of GHGs. *Id.*

268. See, e.g., Chinn, *supra* note 2, at 115–22; Drury et al., *supra* note 3, at 284–88; Meredith Fowlie & Nicholas Muller, *Designing Markets for Pollution When Damages Vary Across Sources: Evidence from the NO_x Budget Program 2–4* (U.C. Berkeley & Nat'l Bureau of Econ. Research, 2010), available at <http://ei.haas.berkeley.edu/pdf/seminar/Seminar20111202.pdf>; Evan Goldenberg, *The Design of an Emissions Permit Market for RECLAIM: A Holistic Approach*, 11 UCLA J. ENVTL. L. & POL'Y 297, 313–17 (1993); Johnson, *supra* note 2, at 147–64; Kaswan, *supra* note 2, at 10304–07; Krupnick et al., *supra* note 263, at 238–42; Nicholas Z. Muller & Robert Mendelsohn, *Efficient Pollution Regulation: Getting the Prices Right*, 99 AM. ECON. REV. 1714, 1735–37 (2009); Nash & Revesz, *supra* note 1, at 572–73; Tom Tietenberg, *Tradeable Permits for Pollution Control When Emission Location Matters: What Have We Learned?*, 5 ENVTL. & RESOURCE ECON. 95, 103–10 (1995).

269. The definition of industrial hotspots need not be identical to mine, but it must be defined in absolute terms (minimum cancer risk) and relative terms (percentage of cumulative cancer risks attributable to industrial sources).

“zonal trading”), and (3) pollution offset markets in which sales of credits to sources located in industrial hotspots would be subjected to a premium (i.e., greater than a one-to-one ratio of credits to GHG emissions).²⁷⁰ My primary purpose in discussing them is to highlight the increased efficacy and administrative ease of implementing them when the number of potential hotspots is small and readily identifiable. I will briefly describe each of the strategies and then highlight how they benefit from a targeted approach.

Heightened monitoring and informational requirements could be imposed in a variety of ways. At minimum, they could involve reporting the increased emissions of toxic co-pollutants that would be associated with purchasing pollution credits. As a purely informational approach, this would minimize transaction costs and could limit them further by using EPA emissions factors to calculate emissions differentials. More elaborate requirements could include added monitoring requirements in and around facilities, as well as high-resolution modeling of the impacts on local pollutant levels and risks. In other words, facilities wishing to purchase credits could be required to provide high-quality information on the local impacts of their proposed trades and to make this information available to the public. These added requirements would increase the effective cost of GHG credits and potentially lead to public pressure, both of which could reduce the attractiveness of purchasing GHG emissions credits.

A virtue of an approach that singles out higher-risk facilities is that the information would be required only where it would be most valuable. Nor would these requirements lead to an unmanageable amount of new information that could be difficult for either the EPA or the general public to absorb and utilize effectively. Imposing elevated standards selectively could also have spillover benefits. For example, higher-quality information on emissions of air toxics could be used to improve emissions inventories and risk estimates for facilities elsewhere, and this in turn could be used to set priorities under the CAA toxics program.

Geographic restrictions on pollution trading have long been discussed as a strategy for protecting against hotspots under cap-and-trade regulations.²⁷¹ They often take the form of strict limits on trades between geographically delimited zones within a trading area.²⁷² The pollution trading program in southern California (RECLAIM), for example, has two zones (one coastal and one inland); trades that could increase pollutant levels in the more-industrialized coastal zone are banned.²⁷³ Numerous variations exist on this basic strategy, including highly calibrated systems that restrict trades if they could “lead[] to a violation of an ambient standard at any receptor point.”²⁷⁴ Regardless of the specifics, restrictions on pollution trading cover specified sources or trades and are designed to prevent increases in aggregate emissions in a particular geographic zone.

270. See, e.g., Nash & Revesz, *supra* note 1, at 614–15.

271. See *supra* note 268.

272. See *supra* note 268.

273. Specifically, new sources and existing sources seeking to exceed their initial allocation of emissions can only purchase credits within the coastal zone. Nash & Revesz, *supra* note 1, at 611–12.

274. *Id.* at 624–25.

The EPA emissions and cancer risk data provide reliable metrics for identifying the geographic zones (on the order of 240 census tracts in seventy-three counties) in which trading might be restricted under a national GHG trading system.²⁷⁵ Further, the distinctive characteristics of industrial hotspots ought to simplify implementation of such restrictions as most of the industrial hotspots in the United States are caused by a single, or small number of, facilities. They are also located predominantly in rural areas and small cities, which means that few, if any, other sources would be affected by targeted restrictions on trading. The most significant exceptions would be Houston, and to a lesser extent Pittsburg and Birmingham, but even for large cities precedent exists for pollution trading as demonstrated by Los Angeles. Concerns about the complexities of implementing trading restriction in urban settings may be mitigated further by the large scale of a national GHG trading market.

The third mechanism, pollution offsets, can be structured around specific classes of facilities or geographic zones. For example, either steel mills alone or all facilities within a certain radius of a steel mill could be subjected to a premium for purchasing GHG credits (e.g., required to purchase 1.2 credits for every ton of GHGs). Furthermore, the tractable numbers could allow offset ratios to vary according to the risks posed by toxic co-pollutants in the associated jurisdictions. These measures could be combined with the heightened monitoring requirements discussed above, each of which would increase the costs of emitting GHGs and thus enhance incentives to reduce emissions of GHGs and toxic co-pollutants.

The relative ease of implementing a pollution offset program when the number of facilities or jurisdictions is small mirrors that for the heightened monitoring and information requirements. When delimited geographically, pollution offsets share many similarities with zonal programs, with the primary difference being that premiums are imposed rather than rigid restrictions on trades. For a pollution offset program, arguably the greatest benefit of the scarcity of potential hotspots is that the offsets could be set for specific facilities or jurisdictions and be optimized over time. The limits of administrative capacities for such refinements would be drastically reduced if the numbers were large.

This brief discussion of legal mechanisms is merely intended to highlight several of the practical benefits of being able to readily identify and characterize the small number of industrial hotspots potentially at risk under a national GHG trading program. The primary virtues are straightforward: addressing potential inequities would be very unlikely to impact the efficiency of a national GHG trading market, and the tractable numbers would enable accurate monitoring as well as optimization of legal mechanisms over time.

CONCLUSION

This Article presents the first synthesis of EPA emissions inventory and cancer risk data for air toxics in the United States. The data show clearly that vehicles and small stationary sources emit a majority of the air toxics nationally and account for

275. Recent reductions in toxic emissions from steel plants could cause the number of industrial hotspots to fall. *See supra* note 252.

most of the cancer risks. This basic pattern is replicated at spatial scales ranging from census tracts to the nation as a whole. It is most pronounced, however, in large metropolitan areas, which have the lowest air quality and are home to eighty-percent of the U.S. population. Industrial facilities account for a higher proportion of air toxics in rural or small-urban jurisdictions, but this occurs in fewer than 250 census tracts nationally and is closely associated with a handful of industries.

The secondary status of industrial facilities as sources of air toxics largely neutralizes the potential for GHG trading programs, whether at the state or national level, to cause toxic hotspots. In the vast majority of jurisdictions, industrial emissions are simply too low, and in the few jurisdictions in which disparities cannot be ruled out, targeted policies exist to prevent them without compromising market efficiency. Further, the racial demographics of the industrial hotspots identified in the present study mirror those of the country as a whole and thus would not give rise to significant racial disparities.

Thus, while localized hotspots cannot be ruled out, the likelihood in the near term of a GHG trading program, whether in California or at the national level, causing disparities in exposures to toxic co-pollutants is exceedingly low. These findings are likely to be generalizable to other pollution-trading programs and to market-based regulations generally.

APPENDIX A: SELECTING METRICS AND MANAGING UNCERTAINTIES IN EPA'S AIR TOXICS DATA

This appendix addresses two issues related to the EPA data. In Part A, I examine questions about choosing a metric to assess environmental inequities, including the range of sources covered. In Part B, I discuss the uncertainties and potential errors in EPA's emissions inventories and their potential impacts on cancer risk estimates. I will briefly review the technical issues and their implications for my conclusions.

A. Choosing the Right Metric for Studies of Environmental Inequities

The focus of the environmental justice literature on industrial facilities in isolation was likely driven by the unavailability of other data but has persisted despite the emergence of major databases for all outdoor sources of air toxics. Utilization of the different types of data will nevertheless depend on the specific questions at issue. The various metrics (geographic distribution of sources, emissions levels, excess cancer risks) used in the studies described in Parts I–III illustrate this point—each one sheds a unique light on the nature and magnitude of the harms.

The data also vary with regard to their accuracy and the robustness of the methods used to generate them, and inevitable tradeoffs will exist in choosing to rely on one form of data over another. One researcher might, for example, choose a direct measure of harm in the form of a cumulative risk assessment, whereas another may favor studies of geographic disparities in the siting of industrial facilities. The former decision could be motivated by a concern that location alone does not provide information about risks to human health—at best, it operates as a loose proxy for inferring harm. The latter decision could be motivated by concern that cancer risk estimates are based on uncertain estimates of chemical toxicity or dubious models for projecting the dispersion of chemical toxins in the atmosphere.²⁷⁶

These tradeoffs involve a challenging mix of quantitative uncertainties and qualitative judgments. Fortunately, the tradeoffs between different metrics are less significant for the present study. One of the more striking aspects of the data on air toxics is the consistency of the results across different metrics and spatial scales. The congruence of the empirical studies mitigates the potential for fundamental points of disagreement.

People may also view the data presented here differently depending on whether they focus on relative or absolute risk levels, and perspectives may shift according to the context. Cumulative risks from air toxics in the United States span more than three orders of magnitude—from below one in a million to above 1000 in a million.²⁷⁷ This is part of the context relevant to evaluating, for example, a thirty percent disparity in which white communities experience a cancer risk of twenty

276. See, e.g., Adelman, *supra* note 228, at 122–24; Howard Latin, *Good Science, Bad Regulation, and Toxic Risk Assessment*, 5 YALE J. ON REG. 89, 90 (1988).

277. EPA, SUMMARY OF RESULTS, *supra* note 145, at 3–4; MATES III, *supra* note 19, at ES-2 to ES-3.

per million while minority communities experience one of twenty-six per million. In most counties and census tracts the potential disparities will fall far below this, and where they may not (for example, industrial hotspots), I highlight them. In doing so, I am seeking to be attentive to concerns about relative and absolute cancer risks.

B. Accounting for Errors and Uncertainties in EPA's Emissions Inventories and Cancer Risk Estimates

For purposes of my argument, the impact of errors in EPA's emissions inventories for air toxics will be greatest in jurisdictions in which industrial sources dominate. The estimate as shown before involves basic discounting: estimates that are low by a factor of two when industrial sources account for five percent of aggregate emissions would cause a ten percent change in total emissions, whereas the same relative error would alter aggregate emissions by fifty percent if industrial sources accounted for half of total emissions. For this reason, errors in the EPA's emissions data for industrial sources will rarely have a material impact on emissions level or cancer risk estimates at either the census-tract or county level.

The secondary status of industrial emissions in most jurisdictions simplifies my analysis but does not allow me to ignore the uncertainties in the EPA's data altogether. To the contrary, it is precisely those jurisdictions in which industrial emissions dominate, and thus are at greatest risk of creating hotspots, that the uncertainties in the EPA's data must be considered. As described in further detail in Part III.A and IV.B.1, industrial hotspots are closely associated with steel production facilities, which are found in multiple jurisdictions, and the highly industrialized area around the ship channel in Houston, Texas. In both cases, emissions from industrial sources are significant in relative and absolute terms.

The uncertainties in the EPA's data cannot be quantified rigorously. Scientists have directly monitored emissions from major industrial sources, such as oil refineries, and compared the measured levels to those in the EPA emissions inventories. Some of these studies suggest that the EPA's emissions factors could be low by more than a factor of two and possibly more than a factor of ten in certain cases.²⁷⁸ These results must be evaluated carefully, though, as they involve sources for which the errors were anticipated to be the largest.²⁷⁹ The EPA maintains that for most sources its emissions factors are unlikely to be off by more than a factor of two.²⁸⁰ On balance, given the small number of industrial sources for which large errors are suspected, I will use EPA's factor of two as the upper bound on potential errors in its data.

If we assume a one-to-one linear relationship between cumulative cancer risk and the emissions level of an air toxic,²⁸¹ rough estimates of the impact of revising

278. See *supra* notes 120–25 and accompanying text.

279. See *supra* notes 120–25 and accompanying text.

280. See *supra* notes 126–31 and accompanying text.

281. This assumption mirrors the use of emissions factors by the EPA. See *Emissions Factors & AP 42, Compilation of Air Pollutant Emission Factors*, EPA, <http://www.epa.gov/ttn/chief/ap42/index.html> (describing the simple linear formula in which emissions factors are used).

the EPA's emissions factors upwards can be calculated. Under this simplified approach, a doubling of an EPA emissions factor would double the cancer risks associated with an industrial source. These estimates represent an upper bound on the derived cancer risks where EPA emissions factors off by the factor of two are believed to bound EPA's estimates.²⁸²

This analysis can then be applied to the two classes of jurisdictions subject to heightened risks from industrial sources. Toxic air emissions from steel mills or foundries were largely, if not exclusively, responsible for fifty-eight of the sixty-five census tracts in which industrial emissions were the dominant source of cancer risks in 2005.²⁸³ In these census tracts, steel plants on average accounted for about sixty percent of the cancer risks from air toxics.²⁸⁴ If EPA emissions factors were off by a factor of two, the average cumulative cancer risk in these census tracts would rise from about 112 to 179 per million, which would place most of these tracts among the top one hundred for cancer risk nationally.²⁸⁵

The contribution to emissions of air toxics from industrial sources in Houston was more modest. Among the most industrialized census tracts, emissions from industrial sources on average accounted for thirty-seven percent of cumulative cancer risks.²⁸⁶ My rough error estimation method predicts that errors in EPA emissions factors of a factor of two would cause estimates of the cumulative cancer risk to rise from roughly 150 per million to about 300 per million. This error correction would cause cumulative cancer risks from air toxics in these census tracts to be among the very highest in the country.

These calculations demonstrate that errors in EPA's emissions factors could have a substantial impact on the estimates of cumulative cancer risks in these jurisdictions. The errors could both dramatically underestimate the relative impacts of industrial sources and obscure the potential for a GHG trading regime to cause hotspots of toxic co-pollutants to arise. The jurisdictions that I have designated as industrial hotspots consequently have two strikes against them. The high relative contribution of toxic air emissions from industrial sources makes them both the most likely sites for GHG-trading-induced hotspots and the most sensitive to uncertainties in EPA's emissions and cancer risk data.

282. The approach is conservative to the extent that a one-to-one relationship does not exist between emissions levels and risk. See Bryan Hubbell, *Understanding Urban Exposure Environments: New Research Directions for Informing Implementation of U.S. Air Quality Standards*, 5 AIR QUALITY ATMOSPHERE & HEALTH 259, 261–62 (2012) (describing the complexities of air toxic exposures in an urban setting); Ariel F. Stein, Vlad Isakov, James Godowitch & Roland R. Draxler, *A Hybrid Modeling Approach to Resolve Pollutant Concentrations in an Urban Area*, 41 ATMOSPHERIC ENV'T 9410, 9413–14 (2007).

283. See *supra* Part IV.B.1.

284. See *supra* Part IV.B.1.

285. The two census tracts, one in Allegheny, Pennsylvania, and the other in Brooke, West Virginia, were ranked first and fifth with respect to the cumulative risks from air toxics.

286. Point sources accounted for thirty to sixty-four percent of the cumulative risks from air toxics in these tracts.

