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The Distributional Impact of Air Quality Regulations: Smog Controls in Los Angeles and Toxic Air Releases in Houston and Los Angeles

by

Chang-Hee Christine Bae
University of Washington, USA

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INTRODUCTION

This paper addresses the important issue of the distributional impacts of air quality regulations in the United States. It discusses the problems related to environmental justice and equity, an increasingly debated issue. It also compares the relative impacts of direct regulation and market-based incentives. It evaluates these results with two case studies, air quality regulations (primarily relating to smog) in Southern California and toxic air releases in Houston and Los Angeles.

ENVIRONMENTAL JUSTICE, RACISM AND EQUITY

The terms “environmental racism,” “environmental justice,” and “environmental equity” overlap. The former two are more applicable in the United States, although even there “environmental equity” may be the preferred language because adverse environmental impacts affect all low-income groups, not only minorities. To apply the concept of “environmental racism” as a universal precept prejudges motivations that are not always valid. For example, if negative externalities are capitalized in house prices and rents, many low-income (or minority) households will seek out cheaper neighborhoods despite their greater risks of environmental exposure

As for “environmental justice,” it is more appropriate in activist discussions (Hofrichter, 1993). Striving for justice is a desirable goal, but attaining it is rare. “Environmental equity” is appealing because society’s concerns should be for everyone exposed to dangerous pollution levels, regardless of income, ethnicity or other socio-economic characteristics. Equal equity is infeasible (just as equal incomes are impossible), but it is reasonable to argue that society is obliged to aim for minimizing environmental inequities among groups, especially in contexts that may be hazardous.

President Clinton’s Executive Order 12898 on Environmental Justice, February 11, 1994, was a key event. It required Federal agencies to address disproportionately negative health and/or environmental impacts on minority and/or low-income populations. The task force referred to three seminal events: i. in 1982, demonstrations against a proposed site for a PCB landfill in Warren County, North Carolina, received national attention; ii. in 1983, a GAO (General Accounting Office) study found that 75 percent of the hazardous waste sites investigated in the Southern U.S. were in African-American communities; and iii. in 1987, the UCCCRJ (the United Church of Christ Commission on Racial Justice) study found that communities that contained hazardous waste facilities had disproportionately high minority populations. The task force drew the distinction among racism, justice and equity, but obviously had a preference for “environmental justice.”

AQMD (the Southern Californian air pollution agency with an African-American as the Chair of its Board), on the other hand, defines its equity-oriented task as promoting “equitable environmental policymaking and enforcement to protect residents, *regardless of age, culture, ethnicity, gender, race, socioeconomic status, or geographic location*, from the health effects of air pollution (my italics).” This is an acceptable approach: Nationally, poor whites have been unnecessarily exposed as well as poor minorities, so while race is an important variable in the United States it is not obvious that is more important than income or class. Tiefenbacher and Hagelman (1999) suggest a somewhat different distinction depending upon the issue emphasized: the health hazard problem (environmental equity), a political assessment of the distribution of the environmental impacts (environmental racism), and the response to the distributional issues (environmental justice).

Kraft and Scheberle (1995) emphasize: the distribution of environmental risks across population groups over space and time (environmental equity); efforts to overcome the reluctance of public officials to clean up minority neighborhoods (environmental justice); and discriminatory interventions (or non-interventions) that result in higher minority exposure to pollution (environmental racism).

Assessing this literature strengthens the argument for the concept of "environmental equity," although a case can be made for the alternative terms. The point is that the poor of all races (including whites) are at risk from environmental hazards, so "environmental racism" is problematic as a comprehensive descriptor Also, although "environmental justice" has some appeal, there is no guarantee, perhaps even few prospects, that the poor and/or minorities will get real-world justice.

CONCEPTUAL ISSUES IN ENVIRONMENTAL EQUITY

i. The Chicken vs. Egg Controversy

Which came first, the pollution source or the low-income population?[1] Most researches use cross-sectional data to draw causal inferences that can be tested only with time series data. Unfortunately, we do not have many time series data. The identification of a major local environmental problem may result in declining property values that attract lower-income households who did not live there prior to the discovery of the disamenity (this is the "market dynamics" hypothesis, Been, 1994a). There is a serious endogeneity problem between exposure to environmental risks and the location of low-income populations. This is the most important aspect of the chicken-egg controversy. With respect to TRI releases, Mitchell *et al.* (1999) concluded that the facilities came first, providing empirical evidence for the "market dynamics" hypothesis. There is an additional complication. The environmental risks of a particular site may be known only recently. Low-income households may have willingly located within visual site of an industrial facility, where it is the blight factor that explains the low property values. Yet it may have been much later when society became aware of the real dangers associated with the site (soil contamination, air pollution, etc.) that were unknown when the facility was established or when the residents located there.

ii. Scale

The "chicken-egg" debate is much easier to answer at the most macro of scales (e.g. the choice of landfill sites within a State). Much of the research (e.g. Cutter and Solecki, 1996), while valuable, uses County level-data that is not disaggregated enough to measure the more critical neighborhood impacts. While Tiefenbacher and Hagelman (1999) found that Texas counties with a high percentage of minorities were more likely to have higher toxic releases (both chronic and acute), Census Tract level analysis discovered such a relationship in only one (Bexar County, San Antonio) of three sample counties. Similarly, in an earlier study of EPA (Environmental Protection Agency) criteria pollutants, Napton and Day (1992), using Census Tract data for Texas urban areas, found no evidence that the poor were more exposed to environmental risks. Furthermore, they argued that many were willing to trade off air pollution exposure for improved accessibility. Scale also affects statistical results and their interpretations. Cutter *et al.* (1996), for instance, showed that correlations and other statistical measures are highly sensitive to levels of scale.

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[1] This paper will use the term "low-income," but it should be remembered that in the United states the equation of "low-income" with "minority" is very common and often appropriate.

iii. Distance Decay

Spatial externalities are very important at the intrametropolitan level. The hedonic pricing literature on housing markets has shown repeatedly how proximity to these externalities (both positive and negative) affect housing prices. Distance from a major pollution source may be positively associated with house prices, but the slope can vary every 100 meters. Kohlbase (1991) found that damage to residential property values dissipated 6.2 miles way from a toxic waste dump. Other disamenities dissipate at different distances, e.g. hog operations at two miles (Palmquist *et al*., 1997), waste incinerators at 3.5 miles (Kiel and McClain, 1995), and landfills at 2.5 miles (Genereux et al., 1992). More generally, house prices show an inverted-U function with distance from a pollution source.

iv. Income

Many observers (e.g. Been, 1995; Boer et al., 1997; Szasz and Meuser, 1997; Sadd et al., 1999) have discovered an inverted U-shaped function when your relate pollution exposure to income. The lowest-income areas are often residential neighborhoods with few economic activities with minimal exposure to air pollution from stationary sources, while high-income neighborhoods have successfully avoided exposure to unwanted land uses. Hence, lower-middle class neighborhoods tend to be the most vulnerable. This is especially the case in Los Angeles County where the most impacted neighborhoods and cities are working class (usually Latino) communities close to industrial zones (Boer *et al*., 1997); finding affordable housing close to work may help to bring about this result (Anderton *et al.*, 1994a and b).

v. Stationary vs. Mobile Sources

There is a major difference between stationary source and mobile source pollution. Stationary sources that are spatially very constrained and isolated are often difficult to deal with (except via closure or very expensive cleanup, and too few people are affected). There is more of a consensus, among the rich and poor alike, for cleaning up mobile sources (especially ozone precursors) because the benefits are often region-wide rather than neighborhood specific. However, these benefits are not uniform among groups; surprisingly, the poor and minorities gain more from cleaning up the air. They are more likely to live in polluted neighborhoods where air pollution is capitalized in low housing prices (and a surprisingly large percentage may be homeowners rather than renters). Also, gains in higher property values are more important than the unemployment risks associated with pollution control costs. There are several reasons: emission technology costs are widely spread among all drivers, pollution control creates almost as many jobs as it loses, and the job losses are not skewed towards low-wage occupations (Bae, 1997a and 1997b and the first case study in this paper). CO (carbon monoxide) and PM-10 are more of a local problem (e.g. pollution levels next to the freeway are 30 times higher than at neutral locations, although they have already declined by 70 percent at a distance of 300 meters). Thus, the low-cost housing adjacent to freeways is susceptible not only to noise but also to PM-10 and to invisible CO emissions. There is little that can be done (at least in the short run), except by disallowing housing there, and thereby prohibiting the poor their free choice of access to low-cost housing. Another potential problem is the “hot spots” issue, whereby serious neighborhood polluters are given credit for non-local pollution clean-up [e.g. subsidizing “clunker/guzzler” return programs, and obtaining the right to pollute more at stationary sources (e.g. petroleum refineries)], often in low-income neighborhoods. This frequently involves a transfer of a mobile source into a stationary pollution issue

vi. The Scope for GIS Analysis

Some of the recent GIS research has used TRI and other pollution maps as overlays on income and racial composition maps. This has generated quite useful results, but the maps are only as good as the input data and how we should interpret these data. For example, Ringquist (1997) found that “background conditions” (such as manufacturing employment, the level of urbanization, the risk of groundwater contamination and potential political power) are much better predictors of TRI exposure risk than either race or class. Another major problem with GIS analysis is that we remain severely deficient in our knowledge of how the health impacts of pollutants are attenuated by distance. The typical micro-spatial analysis using GIS of an overlay of income spatial distributions over a geographical distribution of air pollution levels, toxic releases or noxious sites (hazardous wastes, landfills, etc.) is very helpful, but yet insufficient. It primarily emphasizes the locational aspects of the issue (e.g. the proximity of exposed populations to dangerous sites and releases), and both geographers and planners are very interested in this information. However, it provides only a relevant and important information base, not the full picture.

A major problem with spatial techniques such as GIS is that we do not know enough about how health and other damage impacts decay with distance; knowledge will be correlated with smell, visual aesthetics and other attributes that may or may not be closely correlated with the more serious health damage variables. Health impacts also vary widely among pollutants. As social scientists, we are dependent on natural scientists for information, and even their knowledge base is lacking. Other disciplines are needed, such as climatology and public health. There are few areas of research where interdisciplinary collaboration is needed more to advance the knowledge base.

vii. Overcoming the Problem of Limited Monitoring Stations

We could better measure the distribution of air quality impacts if we had more air pollution monitoring stations that would permit better use of dispersion models. These models raise the possibility of estimating air pollution levels at individual locations based on extrapolations from even a modest number of air monitoring stations. In earlier research (Bae,1997a and b), I derived a pollution exposure surface for the Los Angeles metropolitan region as a whole from point estimates at 33-37 monitoring stations (depending on the pollutant). I derived estimates of dose exposures per capita for 149 individual cities via distance-decay functions from the *n* closest monitoring stations (where n = or < 4). I then converted these dose exposures into potential health benefits (if this pollution could be reduced or eliminated) by assuming linearity between dose exposures and health benefits and adjusting countywide estimates of health benefits to the individual city level. This approach, while of value, has several limitations, one of the most important being the too few monitoring stations in most metropolitan areas. For example, in Houston there are only 23 monitoring stations (none in the fast-growing south-western part of the city or in four of the eight counties of the metropolitan region). Also, monitoring stations usually focus only on the criteria pollutants, obviously a response to the fact Clean Air Act non-attainment can lead to Federal sanctions, such as a cut-off of transportation funds.

DIRECT REGULATION AND ITS ALTERNATIVES

Direct regulations are typically of four types: controls on the amount of pollutants generated with penalties levied for exceeding those amounts; the setting of technology standards (e.g. for automobile emissions); prohibitions on the use of certain products or on the location of particular firms at specific sites (a permit system may alternatively be adopted, setting limits rather than prohibitions); and controls on behavior (for example, employee trip reduction programs or limits on the use of private automobiles by location or time of day).

One problem with many direct regulations is that they impose uniformity. For example, smog inspection and maintenance programs may be imposed on all vehicles regardless of vehicle age or location. This can be a source of both inefficiencies and inequities.

The major alternative to direct regulation is market-based incentives. An obvious example is emission charges (Table 1 presents a list of direct regulations and market-based incentives with respect to transportation measures to control air pollution). However, these would have to be crude (e.g. based on the average emissions of a specific type of vehicle per mile rather than on real-time emissions). Road congestion pricing is another possibility because freer traffic flows imply less pollution. Another possibility is subsidies for pollution-control technology; these are more likely to be available for stationary sources. Public transit subsidies may qualify as a market-based incentive if the transit operators are private, but the largest subsidies in this area are often for public investment. Marketable (i.e. tradable) pollution permits is another favored idea, although the implementation of a program for tradeable air pollution credits in Southern California (RECLAIM, i.e. the Regional Clean Air Incentives Market) did not work very well because the market was so "thin" – almost no firm wanted to trade. The assignment of property rights is another policy instrument, but is much easier to apply to, say, the preservation of environmentally sensitive areas (where ownership may be very concentrated) rather than in the air quality area (where the polluters are many and very heterogeneous).

Economists prefer using emission fees and other incentive-based approaches to direct regulations. Yet policymakers have shown a consistent trend to prefer the latter. They require less imagination, and are consistent with bureaucratic theories that suggest that bureaucrats favor promoting large agencies. The administrative and control costs of direct regulations are much higher than those of market-based incentives.

One argument that is sometimes used is that direct regulation is more equitable than pricing approaches. For example, Lee (2002) argues that two very common forms of direct regulation (emissions standards and employee trip reduction programs) are much more equitable than market-based incentive approaches, such as per-mile road pricing and old car scrappage schemes (her rationale for the last measure is questionable). As another example that is marginally related to air quality, consider freeway ramp metering (primarily adopted to relieve on-highway traffic congestion with unclear impacts on air pollution because of engine idling on the ramp). Freeway ramp metering is the preferred alternative in most American cities to road congestion pricing, perhaps on political acceptability grounds (Harrington, Krupnick and Alberini, 2001). However, one claimed advantage is its distributional equity. Time is distributed evenly among the rich and poor alike (everyone only has 24 hours each day), whereas money is very unevenly allocated. Hence, time charges are much more equitable than monetary charges because the rich and poor alike wait on the ramp, and the time cost may be progressive because the value of travel time increases with income.

DISTRIBUTIONAL IMPACTS

Society usually prefers policies that have a progressive distributional impact. However, there may be trade-offs between equity and efficiency that are difficult to resolve. One possible stance is to avoid interventions that do not widen prevailing income equities. Policy measures can be assessed across a broad set of criteria. Table 2 shows a few representative examples. Policies that stand relatively well on distributional criteria may be unsatisfactory on others.

Most of the sources of pollution in social equity discussions are stationary. Major exceptions are the possibility of cutting new highways or rail lines through low-income neighborhoods and transportation that is closely connected to a stationary source (such as trucking associated with a port or a major distribution yard).

On the other hand, some measures to reduce mobile source pollution may have a pro-equity impact. For example, consider the case of automobile-related air pollution. Although pollution decays with distance from freeways (especially carbon monoxide and PM-10), serious attempts to improve mobile source pollution have to tackle the problem region-wide, mainly because ozone is widely diffused and may not be at all correlated with local traffic levels. Such measures to improve *region-wide* air quality tend to benefit the poor and minorities because the rich have previously purchased the good air quality sites (Bae, 1997a, 1997b).

For example, in the Los Angeles metropolitan region, the well-off tend to live on the Westside and near the ocean, where the ocean breezes blow the pollutants inland, and where there are few major stationary sources. The poor and the lower middle class, on the other hand, live inland in areas that are both "smog traps" and exposed to higher levels of stationary source pollution. The geography of pollution is very complex, but anti-smog strategies have to apply to the region as a whole and cannot be targeted to specific sites. Hence, air quality improves throughout the region, but by much more in the poor, previously polluted neighborhoods.

However, it is not all good news. The costs of emission equipment are more or less fixed from vehicle to vehicle so the costs of additional emission controls tend to have a regressive impact (i.e. their cost accounts for a higher proportion of a poor household's income). If emission fees were to be introduced (an interesting idea that has not been given much attention), their distributional impact is unclear. In the used car market, the poor would probably pay more in emission fees because of their propensity to own cheap old cars; when they enter the new car market they may pay less because of their purchase of less expensive, much less powerful cars.

Both emission charges and road pricing are inequitable among drivers because costs are assigned to vehicles regardless of income. The regressive impact is stronger with emission charges because poor motorists are more likely to drive older and less well maintained cars.

Whether gasoline taxes are regressive depends on how fuel consumption varies with income. U.S. data suggests that these taxes are very regressive (e.g. Crawford and Smith, 1995). In addition, spending on automobile use across income groups varies much less than income. A related point is the regressivity of per mile charges, e.g. emission fees. Cameron (1991, 1994) provided some evidence on this issue. A five cents per mile charge would reduce regional (Southern California) auto travel by 11 percent, but the lowest income quintile group would cut trips by 29 percent while the highest income quintile group would reduce travel by only 3 percent. The problem with the 29 percent cutback by the poor is possibly adverse welfare impacts associated with abandoning relatively "essential" trips. Nevertheless, he concluded that: "Market-incentive policies … are neither inherently equitable nor inherently inequitable. How they are designed and implemented will determine their impacts on income distribution" (Cameron, 1994, viii). The impact on air quality, however, might be modest: Harvey (1991) estimated that even a 15 percent per mile tax would reduce automobile emissions by only 4 percent. It is not difficult to explain these results. Almost two-thirds of emissions are associated with stopping and starting (the "cold start" and the "hot soak" problems) rather than on-road driving. For technological and cost reasons (e.g. sensors and other equipment), emission fees would most likely be imposed only on freeways; traffic might be diverted to arterial roads with worsening congestion and more air pollution. If emission fees speeded up auto travel from moderate to high speeds, pollution would increase because the optimal driving speed (from the air pollution perspective) is 30 to 40 mph. If Harvey's analysis is even approximately correct, a uniform per mile emission fee is less efficient than a one-time emission fee paid upon purchase proportional to lifetime emissions because the latter is more likely to affect purchase choices. Also, it is probably more equitable because the high-polluting vehicles (such as SUVs) tend to be more expensive.

Most environmental improvement measures may be expected to have pro-rich redistributive impacts in the absence of specific redistributive measures to assist the poor. Although air quality improvements benefit all, the imputed monetary value is higher for the well-off because the income elasticity of demand for environmental quality is high (for an alternative view, see Kriström and Reira, 1996).

The problem may be especially acute with market-incentive approaches targeted at the motorist, for example, the imposition of emission fees and changes in gasoline taxes. This problem can be mitigated by compensation measures. Redistribution of revenues from a program may improve its political feasibility, for example Klein's (1995) proposal for repair subsidies in a remote-sensing smog testing. Small (1983) examined several alternative revenue redistribution schemes.

Nevertheless, the use of direct regulations, such as mandated equipment to cover the costs of environmental regulation, is also almost invariably regressive in the United States. In addition, most environmental regulation is paid for either out of general revenues or dedicated sales tax increments. Sales taxes are always regressive because the propensity to consume declines with rising incomes so that the less well-off pay a higher proportion of their income in sales taxes. Also, despite appearances, the income tax system in the United States is, at best, neutral and probably regressive; the reason is that the rich can use the deductions allowed under the Income Tax Code to avoid the on-paper progressivity in income tax rates. In addition, higher taxes (regardless of their direct distributive burden) have indirect adverse impacts on the less well off because higher taxes mean less consumer demand, and the brunt of the induced consumption impacts fall on the working poor in the retail and services sector.

An example of mandated equipment costs is smog emissions equipment and its maintenance, especially in California. In general, the cost of emissions equipment varies little with the cost of the vehicle and the regular smog tests are more or less fixed. Moreover, it is the older vehicles owned by the poor that are much more vulnerable to expensive repairs for failing the tests. When the costs of mandated environmental technology fall directly on industrial corporations, the effects are more mixed; it depends on the specific industry, the type of product (e.g. whether a consumer good, and the market for that good), and the occupations directly and indirectly impacted. It is not possible to generalize about the distributional consequences without detailed and very specific analysis.

Another important point, discussed by Stoft (1993), is whether imposition of regulatory standards adversely affects the poor. The analysis can become very complex, branching into risk portfolio analysis and other aspects of business economics. But the key result is very simple: The payback from improved health as a consequence of automobile emission standards is quite rapid and insensitive to discount rate assumptions, and the discounted cost of emissions equipment relative to the diversified expenditures of even the poor is low. So, there is a significant payoff to all, including the poor, from mandated investments in catalytic converters.

The benefits side of distributional impacts is much more complicated because much depends on the resulting changes in driving behavior and the extent to which behavior varies across income groups. The key issue is whether equity concerns are enough justification to jeopardize the efficiency of market-related types of intervention.

DIESEL PARTICULATES

The Europeans have been much more aggressive addressing the global warming issue[2], but the resulting priority given to diesel fuel (because of its lower CO_2 emissions) may have been problematic. In the interests of reducing the rate of global warming and promoting commerce, many European countries tax diesel fuel at a significantly lower rate than gasoline. Fuel consumption is more efficient with diesel, and weaker performance characteristics are mitigated by the proliferation of turbo-diesel engines; these factors induce many Europeans to buy diesel cars (35 percent of the European automobile stock, and accounting for almost one-half of the fleet in some countries). Although European diesel vehicles are much cleaner than those in the U.S. because of reliance on low-sulfur fuel, total emissions of other pollutants remain substantial, aggravated by compactness and narrow streets where 4-5 story buildings trap pollutants in corridors.

In the United States, the diesel share in fuel consumption rose from 8.8 percent in 1975 to 19.9 percent in 1999. Recent medical research has revealed that diesel particulates are the most dangerous of air pollutants, especially in terms of carcinogenic effects, far more dangerous than ozone, where the damage is largely related to lung function, asthma, etc. Yet the trucking lobby in the United States has, up to very recently, been quite successful in resisting adverse legislation. The new diesel regulations will not be implemented until 2007 (although some States are going ahead with regulations sooner), and there is anecdotal evidence that trucking companies are buying up the cheaper and better performing high-polluting trucks in anticipation of the new rules (note that the expected lifetime of 1990 heavy trucks is 29 years).

One important development in Southern California is that the Air Quality Management District has imposed a regulation requiring government agencies to substitute alternative fuel for diesel vehicles, and a challenge by the Engine Manufacturers Association and the Western States Petroleum Association was rejected by the US Ninth Circuit Court of Appeals in October 2002. Unless there is a successful appeal to the U.S. Supreme Court, this augurs well for similar regulations throughout the United States.

There is a potentially important trade-off (CO_2 emissions vs. diesel use) here, but the topic has been little researched, perhaps because the research into the health impacts of diesel fuel is so recent. The preferential taxation for diesel fuel may be an example of wrong-headed policies based on incomplete or out-of-date information, but this must await the outcome of a detailed cost-benefit analysis and perhaps better data (the

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[2] The most troublesome issue in the distributional effects of air quality is that among generations (intergenerational equity), especially the global warming question. The problem arises from three sources: doubts about whether the next couple of generations (particularly on a global scale) will be richer or poorer than today's; uncertainties about the degree of regulation needed now to ensure that the future is safer (e.g. the "safe minimum standard;" Toman, 1994); and speculation about what the benefits might be in, say, 50 years.

The global warming issue remains somewhat controversial (National Research Council, 2002), at least in the United States. There is probably now a consensus that global temperatures are rising and that this has all kinds of climatic consequences. There is still some debate about the rates of change (depending on time period chosen) and on the relative contributions of natural and manmade sources. Also, although while many of the impacts are clearly adverse (e.g. coastal flooding because of rising ocean levels, more skin cancers) not all are (e.g. there are some benefits at some locations in terms of agricultural productivity). The key point is that the United States does not control auto-related CO_2 emissions (California proposes to do so in the near future). CO_2 emissions from motor gasoline are 60.3 percent of the total, while 98 percent are from petroleum fuels. There was an opportunity to include CO_2 emissions in the 1990 Clean Air Act, but it did not happen for undisclosed political considerations.

US Environmental Protection Agency announced in September 2002 that diesel exhaust may account for one-quarter of all soot particles, and declared it a proven carcinogen).

With respect to the intergenerational distribution issue (protecting future generations) that has dominated these discussions, the conflict may be more apparent than real. The current poor would benefit greatly from reduced diesel particulates (which are higher in poor industrial and port neighborhoods) while the future poor would benefit from reduced global warming (associated with current CO_2 emissions). The obvious win-win solution is technological: cleaner diesel fuel and vastly improved emissions equipment that would reduce diesel particulates emissions. Some have argued that for these reasons the United States may move more in favor of diesel engines in the future. This seems unlikely, because hybrid (gasoline-electric) and fuel-cell powered engines appear a preferred technological solution. However, this latter approach may have adverse distributional impacts, because such vehicles will be much more costly, at least until the production numbers are large enough to maximize economies of scale.

CASE STUDY: AIR QUALITY REGULATIONS IN SOUTHERN CALIFORNIA

Southern California has been the most aggressive region in the United States for direct regulation of air pollution, especially after the late 1980s when a successful Federal lawsuit galvanized the Air Quality Management District into a much tougher stance. The result was stronger regulations and much higher control costs than elsewhere in the United States, e.g. NO_x reductions in Southern California were five times more costly (at $25,000 per ton) than the national average (Lee, 2002). The State of California has had stronger automobile emission controls since the early 1970s, and still leads the way with CO_2 controls to be imposed soon. In general, the burden on the poor may be expected to increase as the costs of compliance rise.

There have been almost no studies that have addressed the question of the aggregate costs and benefits of air quality regulations. AQMD (1988) attempted to answer it in the specific case of Southern California. They estimated the benefits at $11.9 billion (mainly health benefits) and the costs at $1.7 billion. Hence, air quality regulation was overwhelmingly justified in the public interest (of course, the appropriate comparison is equating marginal [not aggregate] benefits and costs but this would not change the conclusion). The problem with AQMD's analysis is that its costs refer only to the direct regulatory control costs incurred by industry. As pointed out by Gordon and Richardson (1988), when you take account of indirect and induced impacts of the control costs, the administrative costs of the California Air Resources Board and the AQMD huge bureaucracies (AQMD's headquarters in Diamond Bar were described as the "Pollution Palace," but now its budget and staffing have been cut), the burdens on consumers in terms of restricted choices (e.g. aerosols, barbecue lighters, furniture purchases, etc.), and most important the congestion costs imposed by the transportation components of the air quality regulations (e.g. more spending on transit at the expense of congestion relief on the highways), the costs can balloon as high as $25 billion. This would appear to be a case of over-regulation. Unfortunately, there has been no more recent aggregate analysis, especially of the benefits side. Because of significant air quality improvement in the last decade and a half and the shift in AQMD's stance to a more business-friendly rather than a command-and-control approach, both the costs and benefits of air quality regulation are probably much lower now. An important qualification to this conclusion is that the benefits and costs of planned diesel particulate controls have not yet been estimated, but are certain to be substantial (e.g. higher truck freight costs and major health benefits).

A major problem in evaluating the costs of air pollution control is the differences in estimates, according to source of authority and methodology. For example, Sierra Research (in an automobile industry sponsored research project) estimated the dollar cost of removing a ton of ozone precursors (NOx and VOCs) between 7-11 times higher than the official source, the California Air Resources Board, depending upon vehicle type. Table 3 (derived from Lee, 2002) compares cost-effectiveness estimates of direct regulations vs. market incentive approaches targeted at vehicle characteristics and travel behavior, partly based on the individual policy measure costs shown in Table 4. Although these data do not directly address the distributional impact issue, it is clear that the very high costs of attempting to regulate travel behavior via regulation must impose an intolerable burden on the poor. This is consistent with the results of much earlier research by Bae (1993) suggesting that technological mandates reduced air pollution much more than measures to reduce vehicle miles traveled by changing travel behavior, such as promoting transit, ridesharing and telecommuting. An important point is that reducing the costs per ton of eliminating major pollutants may be more important in aiding the less well-off than being concerned about the distributional impacts on different income groups.

The best news is that region-wide efforts to reduce smog benefit the less well-off because they are already trapped in the current and past highly polluted communities. The explanation is that smog is a regional not a neighborhood phenomenon and can only be tackled on a macro scale. So, when regional air quality improves, the poor benefit most. I quantified this impact a few years ago (Bae, 1997a, 1997b). The dollar amounts would be different today because of recent improvements in air quality, but the generality of the results remain robust. My methodology involved estimating a Net Welfare Impact (NWI) function for individual households that added up the benefits and costs of air quality improvement by income group, location and race.

The NWI function is calculated as follows:

ΔB_{ri}	$=$	$s_r h_r$	$\pm$	R_{ri}	$+$	v_{ri}	$-$	$u_{mr} w_{mr}$	$-$	Δp_i	$-$	Δt_i
change in welfare		health benefits		housing impact		visibility benefits		unemployment risk		price impact		tax changes

where:

ΔB	=	changes in welfare
s	=	household size
h	=	health benefits per capita
R	=	change in annualized housing value or rent
v	=	visibility benefit
u	=	unemployment rate resulting from AQMP
w	=	wage
Δp	=	change in prices
Δt	=	change in taxes
r	=	location
i	=	income
m	=	industry.

The point is that the health benefits and for homeowners, the windfall property value gains, swamp the costs of higher prices and taxes and the unemployment risks (for example, pollution controls do not eliminate many low-paying jobs). The NWI estimates run into thousands of dollars per year. The key result is that poor households (especially homeowners) in low-income communities gain most. Even within other communities, the low-income and minority households do quite well.

Whereas the cost items are clearly separable, there is a concern that the benefit terms may overlap resulting in some double-counting. While this is possible and leads to the conclusion that the NWI estimates are overbounded, there are sound reasons to separate out the benefit items. First, the health benefits are, by far, the most important, but households lack detailed information about their magnitude so they are not fully capitalized in property values. Second, renters do not benefit from property value capitalization; in fact, they pay a cost in higher rents (that is why the housing impact variable may be negative or positive in the net welfare impact equation). Third, visibility problems are episodic and may not be capitalized, once again because of information deficiencies. In any event, their imputed value is quite small, and visibility does not vary much spatially.

There has been no recent calculation of the price impacts of air quality regulations. An early study (NERA, 1988) assumed that local demand was inelastic, resulting in estimates that were much too high. Conversely, the Los Angeles region is so large in terms of gross regional product that it does not face a perfectly elastic demand curve for its goods and services. Compromising between these extreme positions, and assuming that control costs are evenly split between producers and consumers (an arbitrary assumption certainly, but one with precedent; Pechman, 1985) yields price costs ranging from about $500 for the lowest income groups to over $5,000 for those with incomes in excess of $162,500. These very large numbers would be lower today because regulatory costs have dropped significantly as air quality has improved. These price impacts are regressive, varying from 5 percent of income for the $25,000 household to about 3.1 percent for the richest households. The price impacts ripple through the local economy, but the transportation expenditures are the most drastically affected (also, a few other sectors that are severely regulated such as dry cleaning costs and the price of paint). The tax impacts are also potentially significant, although less burdensome because many of the air quality regulations are self-financing. Assuming that tax requirements are evenly split between sales taxes and a higher gasoline tax, the tax impact would vary from 3.5 percent for the $25,000 household to 1.75 percent for the highest household income group. Most of these revenues would be for transport-related air quality controls, such as public transit investments. The unemployment consequences of regulations are modest, about one per cent of total jobs at the peak of air quality controls, largely because environmental regulations and transportation investments create jobs.

In many, but not all, cities the high costs of air quality regulations are outweighed by the benefits in the form of favorable health effects and capitalized improvements in house values. However, the variation is very wide, ranging between + 8 percent and – 7.6 percent of mean household income. In general, the poor cities gain most because they experience the largest improvements in air quality, and homeowners gain more than renters. The conclusion is that the costs of air quality regulations tend to be regressive, but the benefits are progressively distributed. It should be noted that Los Angeles is an extreme case, and all these estimates would be much lower in other U.S. metropolitan areas. Also, even in Los Angeles, the estimates would be much lower today because of significant improvements in air quality in recent years.

Regulation XV

One of the most notorious examples of direct regulation was Regulation XV imposed by the Air Quality Management District (AQMD) on Southern California between 1989 and the mid-1990s (technically, the regulation remains in place, but it has been downgraded to a voluntary plan). The regulation required employers to develop a ridesharing plan to achieve target levels of employee-vehicle ratios (ranging from 1.3 to 1.7 depending upon the density of the location). In order to implement this, a full-time (or part-time, depending upon the size of the firm) Ridesharing Coordinator had to be appointed. Failure to submit a feasible plan resulted in substantial fines and penalties, but there were no penalties for non-attainment of the plans (the program never reached that stage).

The air pollution impacts of Regulation XV were modest but non-negligible. In the first year, the average vehicle ratio increased by 2.7 percent (Giuliano, Hwang and Wachs, 1992). After that, it slowed down because those able to adjust their travel behavior do so first. The regulation was very unpopular with employers because of the high cost, estimated at $164 million in the first three years. A more meaningful measure is that it cost $3,000 for each private car eliminated by the regulation. To put that in perspective, this is 3-4 times more costly than the "clunker" redemption programs where vehicle owners could receive a cash payment (typically $750) for turning in an old, high polluting vehicle. Although the regulation is no longer being enforced in Southern California, similar programs (generally called Employee Trip Reduction Programs) have been adopted in many cities across the United States under ISTEA (Intermodal Surface Transportation Efficiency Act) and the subsequent TEA-21 (Transportation Equity Act – 21st Century) legislation.

Despite its unpopularity, the distributional impacts of Regulation XV are unclear. Although firms' operating costs increased as a result of its implantation, the cost per employee was quite modest. In fact, the regulation might have had a modest progressive impact as lower-income workers were more likely to be tempted by some of the incentives offered (such as transit passes, company vanpools and "cashing out" employee parking). The "clunker" redemption programs might appear to benefit the poor more, enabling them to obtain a substantial cash deposit for a more reliable vehicle, but this refers to their direct effects. The private companies participating in these programs (e.g. the oil company, UNOCAL) were allowed to take emission credits, allowing them to pollute at their stationary sources (e.g. petroleum refineries) usually located in poor neighborhoods.

CASE STUDY: TOXIC AIR RELEASES IN HOUSTON AND LOS ANGELES

Introduction

This analysis is based on a comparison of Houston and Los Angeles, the two leaders among polluting cities in the United States (Bae, 2002). It focuses on how minorities and the poor are impacted by local toxic air releases (the data here are much better at the spatially disaggregated level than for smog).

Minority Population Shares

The county-wide non-white population shares [including Hispanic whites] are now much higher (58 percent in Harris County [mainly the City of Houston], 69 percent in Los Angeles County [more than two-and-half-times the size of the City of Los Angeles], according to the 2000 Census) than in 1990 (46 percent in Harris, 59 percent in Los Angeles). Although the Latino population in Houston is twice as large as the African-American population, it is much more scattered than the latter. As a result, it is more difficult to apply environmental justice/racism analysis which is usually locationally specific. In Los Angeles County on the other hand, the Latino population is much larger (two-fifths of the total population, more than four times larger than the African-American population) and is growing much faster. For example, there has been a high degree of infiltration by the Latino population into formerly African-American neighborhoods (e.g. in South Central Los Angeles; see Allen and Turner, 2002). Also, the Latino population has gravitated towards many suburban industrial cities in addition to being heavily represented in the City of Los Angeles itself. The coexistence of a large white population and a large Latino population and the sharp average income differentials between the two racial groups make it hard to disentangle the racial, income and spatial effects of environmental exposure. However, one study (Boer *et al.*, 1997) found that 4.2 percent of the Los Angeles County population lived in a Census tract with a toxic site disposal facility, but for whites the share was 2.9 percent while for minorities it was 5.2 percent. Moreover, 20 percent of minorities lived within a mile of a large-capacity facility (i.e. one that processes more than 50 tons per year); the ratio for whites was one-half that.

Analysis

This analysis is based on the more accessible TRI (toxic release inventory) data rather than the less ubiquitous ozone data (too few monitoring stations, especially in Houston) or diesel particulates data (hitherto very sparse). The locations of toxic release sites are compared with the geographical distributions (at the Zipcode level) of minority population shares and median household income levels. Visual comparisons suggest a superficial relationship between the location of toxic sites and race and/or income. The TRI data (based on number of sites rather than emission volumes) are used rather than ozone data because the databases have a much greater spatial coverage at the metropolitan level.

The results are reported in Table 7.[3] In Houston the minority population share is statistically insignificant. However, TRI sites are found disproportionately in low-income Zipcodes (Rho-squared = 0.38) although on average each Zipcode would be expected to have more than two toxic sites regardless of income level. In Los Angeles, on the other hand, the results are different. Household income is again a significant predictor as in Houston but with a much higher degree of predictive power (Rho-squared = 0.60). However, the minority population share is almost as significant (Rho-squared = 0.55). Minority population shares and income levels are more closely correlated in Los Angeles than in Houston. When both minority population shares and median household income levels are introduced in the same equation, the result remains the same in Houston (only income is significant), but in Los Angeles County only the minority population share is significant. This finding, however, may be somewhat problematic because of a degree of spatial multicollinearity between incomes and minority population shares. The Los Angeles results are particularly illuminating. The locations of the toxic sites in many instances antedate the arrival of the minority (typically Latino) populations. In a high housing cost region, many of these recent immigrants made a conscious rational decision to live in relatively inexpensive neighborhoods close to their work, fully aware that the environmental quality was problematic, provided that toxic sites were not so close that their families were at risk.

These results join the debate in the literature about whether race or income is the better predictor of proximity to environmental hazards, but the analysis confirms that toxic air releases have regressive equity impacts. Mohai and Bryant (1992) emphasized race, but several more recent studies (e.g. Bowen et al., 1995, on Cuyahoga County, Ohio; Downey, 1998, on Detroit and other Michigan urban areas; and Stretesky and Lynch, 1999, on Hillsborough County, Florida) have suggested that income is more important. The tendency for income to trump race in most recent studies strengthens the case for “environmental equity” as the preferred concept. This case study straddles the fence with divergent results for Houston and Los Angeles.

CONCLUSIONS

An important point is the distinction between whether low-income households suffer disproportionately from adverse environmental impacts (in this paper, air pollution impacts) and whether they benefit more or pay more from direct regulations (including significant indirect costs). There is no simple generalization possible, although in most cases the poor both pay more but also benefit more. Because many of the benefits are imputed rather than monetary, a key issue is whether they pay disproportionately more for the

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[3] It was intended to use a Poisson regression analysis on the data. Poisson regression analysis was chosen rather than standard regression methods because a discrete number of sites was regressed against minority population shares (and/or median household income) in a discrete number of Zipcodes (TRI emissions in much smaller geographical units might have yielded close to a continuous analysis permitting the more standard approach). However, the Poisson distribution requirement of the mean being equal to the variance was not met, so the normal substitute of a negative binomial distribution was made.

costs of improvement in air quality, and this varies widely. We can start from an initial state, and unequivocally assume that minorities and the poor suffer most from air pollution. But when regulations are taken into account, the poor can benefit, especially if the discussion is about region-wide air quality improvements such as smog reductions. Another implication is that if society desires pro-equity policy measures, these can usually be designed both with command-and-control and market incentive measures. Direct regulation is the preferred approach in practice, despite its inefficiencies and its high operating and enforcement costs. It does have the superficial attraction of being more equitable because the same rules apply to the rich and poor alike. However, in many circumstances, the indirect costs may weigh heavily on the poor, but often not enough to offset the significant health benefits of air quality improvement. Market-based incentive approaches require more imagination and are more efficient, but because they introduce prices (with the implication of the ability to pay) they are often considered regressive. However, because they are usually revenue-generating, they can almost always be made pro-equity via revenue redistribution. The bottom line, regardless of the approach adopted, is that the poor will gain from measures to improve air quality, whether the sources are localized or region-wide.

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Table 1: Regulatory and Incentive-based Measures to Reduce Transportation-Related Emissions

Regulatory Measures	Incentive-based Measures
Emission Standards for Vehicles	Subsidies for Public Transportation
Emission Control Retrofit Requirements	Old Vehicle Scrapping
Alternative Fuel Requirements	Emission Reduction Credits
Inspection/Maintenance Requirements	Motor Vehicle Emission Charges
Employer Trip Reduction Programs	Congestion Pricing
Auto-Use Restrictions	Parking Charges
	Fuel Tax Increases

Source: Lee, 2002.

Table 2: Illustrative Evaluation Criteria for Emissions-Reduction Policy Measures

	Emission Reduction Potential	Cost-effectiveness	Administrative Convenience	Convenience of Information Collection	Equity
Exhaust Emission Standards	moderate	very low	moderate	low	high
Employer Trip Reduction Programs	very low	moderate	high	high	high
Old Car Scrappage	low	high	moderate	moderate	moderate
Congestion Pricing	moderate	high	high	very low	low

Source: Lee (2002).

Table 3. Cost of Ton of Pollutant (VOC and NOx) Reduced

	Vehicle Characteristics -Targeted	Travel Characteristics -Targeted
Direct Regulation	**A** **$19,200**	**B** **$104,657**
Market Incentives	**C** **$ 6,450**	**D** **$3,184**

Source: Lee (2002).

Table 4: Cost-Effectiveness of On-Road Mobile Source Control Measures

Strategies/Measures		Emissions Reductions VOC (tons/day)	NOx (tons/day)	Costs (dollars)	Cost-Effectiveness (dollars per ton)
Advanced Technology Vehicles	TLEV	18.16kg/vehicle life		$61/vehicle	$3,083
	LEV	59.93kg/vehicle life		$114/vehicle	$1,746
	ULEV	64.47kg/vehicle life		$207/vehicle	$2,947
California Phase 2 Gasoline		42.0*	25.0*	$0.07-0.19/gallon	$17,200-46,500
Enhanced Inspection/Maintenance		33.6	58.7	$50,400,000/year	$2,168
Accelerated Vehicle Retirement		13.8	10.7	$63.6million/year*	$14,500
Work Trip Reduction	Alternative Work Schedules Telecommunications Non-Motorized Transportation	21.3	19.9	$277,757,000-$415,942,000/year	$26,966-$40,382
Mode Shift via TDM Measures	Employer Rideshare and Transit Incentives Parking Management Merchant Transportation Incentives Auto Use Restrictions	4.3	5.1	$197,627,000-$316,746,000/year	$85,080-$136,361
Mode Shift via HOV Facilities		1.8	2.2	$34,151,000/year	$34,305.37
Mode Shift via Transit Improvement		3.0	3.6	$798,968,000/year	$81,547.76
Traffic Flow Improvement	Traffic Controls (TSM Measures)	5.4	3.9	$5,154,000/year	$2,202.66
	Goods Movement	2.8	50. 6	$5,171,000/year	$387.50
	Non-recurrent Congestion	0.0044	0.0032	$260,000/year	$137,326.36
Freeway Lane-mile Increase		17.6	14.2	$37,721,000/year	$4,747.77
Emission/VMT Charges	Flat Rate VMT charge	43.8	29.8	$118,000,000	$4,397
	Emission-weighted, 1c/mi	87.5	29.8	$118,000,000	$2,757
	Emission-weighted, 2c/mi	273.5	81.8	$118,000,000	$910
	Emission-weighted, 3c/mi	404.8	126.5	$118,000,000	$609
Fuel Tax Increase	Increase by $0.50	34.8	28.3	$78,000,000	$3,388
	Increase by $2	108.7	92.3	$159,000,000	$2,168
Employee Parking	$1/day	8.5	6.7	$78,000,000	$14,072
	$3/day	22.1	18.6	$78,000,000	$5,254
Congestion Pricing	15c all roads	68.8	26.8	$369,000,000	$10,580
	5/10c freeway only	21.9	7.4	$309,000,000	$28,874
	10/20c all roads	32.8	22.3	$369,000,000	$18,334
	15/30c all roads	43.8	29.8	$369,000,000	$13,751

Notes:

i. Emissions Reduction figures represent average daily reductions in VOC and NOx emissions by 2010, except California Phase 2 Gasoline representing average daily reductions by 2000.

ii. The cost figures are adjusted to 1993 dollars.

Source: Air Quality Management District data assembled and analyzed by Lee (2002).

Table 5: The Influence of Income and Race on TRI sites by Zipcode in Harris and Los Angeles Counties

A. Harris County

TRI Sites	Constant	Minority %	Med HH INC	Rho Squared
TRI Sites	Constant		Med HH INC	Rho Squared
127	2.26397		- 0.36845 E-04	0.37368
	4.232		-2.661	
TRI Sites	Constant	Minority %		Rho Squared
127	0.50713	0.12461 E-01		0.37986
	2.177	2.065		
TRI Sites	Constant	Minority %	Med HH INC	Rho Squared
127	2.5179	- 0.23922 E-02	- 0.41272 E-04	0.37341
		-0.26	-1.801	

B. Los Angeles County

TRI Sites	Constant	Minority %	Med HH INC	Rho Squared
TRI Sites	Constant		Med HH INC	Rho Squared
264	2.95309		-0.38859	0.60341
	8.465		-4.673	
TRI Sites	Constant	Minority %		Rho Squared
264	-0.12858	0.27586 E-01		0.55838
	-0.777	8.359		
TRI Sites	Constant	Minority %	Med HH INC	Rho Squared
264	-1.14436	0.33659 E-01	0.18004 E-04	0.55515
	-1.682	6.456	1.59	