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October 2000 • Discussion Paper 00–46

RESOURCES FOR THE FUTURE

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Abstract

In developing countries, urban clusters of manufacturers which are "informal"—small-scale, unlicensed and virtually unregulated—can have severe environmental impacts. Yet pollution control efforts have traditionally focused on large industrial sources, in part because the problem is not well-understood. This paper presents a benefit-cost analysis of four practical strategies for reducing emissions from traditional brick kilns in Ciudad Juárez, Mexico. To our knowledge, it is the first such analysis of informal sources. We find very significant net benefits for three of the four control strategies. These results suggest that informal polluters should be a high priority for environmental regulators.

Key Words: benefit-cost analysis, informal sector, air pollution, US-Mexico Border, brick kiln

JEL Classification Numbers: 013, 017, 054, Q25, Q28

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1. Introduction

In developing countries, urban clusters of manufacturers which are "informal"—small-scale, unlicensed and virtually unregulated—can have severe environmental impacts (Bartone and Benavides 1997). There are a number of reasons. Most important, such firms are exceptionally numerous. Informal firms account for over half of non-agricultural employment in Latin American and Africa (Ranis and Stewart, 1994) and a significant share of these firms are in polluting sectors such as leather tanning, metalworking, ceramics, textiles, and food processing. For example, in Mexico, 38% of informal firms are classified as industrial (U.S. Department of Labor 1992). In addition, informal firms are usually more pollution-intensive than large firms in the same industry because they lack pollution control equipment and access to basic sanitation services (Lanjouw 1997). Finally, informal firms are a significant source of employment and are often situated in poor residential areas. As a result, their emissions directly affect a considerable population.

Even though informal firms create acute environmental problems, pollution control efforts in developing countries have traditionally focused on large industrial sources. One reason is that applying conventional regulatory instruments in the informal sector is problematic. Informal firms are difficult to monitor since they are small, numerous and (by definition) have few preexisting ties to the state. Also, such firms have few resources to invest in pollution control. Yet a growing body of evidence suggests that unconventional regulatory approaches

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We are grateful to the Tinker Foundation for financial support, and the following individuals for data, helpful comments and logistical support: Fransisco Alfaro, Charles Bruce, Bob Curry, Alan Krupnick, Antonio Lara, Robert Marquez, Anders Johnson, Wen-Whai Li, Gerardo Tarin, Victor Valenzuela, and Jesus Reynoso. Remaining errors are our own.

such as peer monitoring, educational campaigns, and subsidies to clean technological change can motivate informal firms to significantly cut emissions (Blackman 2000; Blackman and Bannister 1998; Dasgupta 2000; Frijns and Van Vliet 1999).

A second reason that informal sector polluters have received relatively little attention may simply be that the problem is not well-understood. Policy-makers lack a clear understanding of the magnitude and incidence of the damages caused by informal polluters and of the costs of mitigating these damages. As a first step toward filling this gap, this paper presents a benefit-cost analysis of four practical strategies for reducing emissions from informal brick kilns in Ciudad Juárez, Mexico which are significant sources of air pollution owing to their reliance on cheap dirty fuels. To our knowledge, this is the first rigorous analysis of the benefits and costs of informal sector pollution control. We find that the expected benefits of three of the four control strategies are considerably higher than the costs. We also found that, while brick kilns have significant area-wide health impacts, the main beneficiaries of pollution control efforts are likely to be the residents of the poor neighborhoods surrounding the brickyards.

Our results do not make a definitive case for redirecting scarce pollution control resources away from large industrial sources and to the informal sector. This would require comparing the net benefits (benefits minus costs) of targeting informal sources with the net benefits of targeting large industrial sources. While such a comparison is beyond the scope of this paper, the magnitude of our estimates of the net benefits of reducing emissions from informal sources suggests that they are likely to be at least as great as those associated with large sources.

The paper is organized as follows. The second section provides background on air pollution in Ciudad Juárez and on traditional brickmaking. The third section details our data and methodology and the fourth section presents our results. The last section sums up and concludes.

2. Background

Air quality in Paso del Norte—the metropolitan area comprised of Ciudad Juárez and its sister city, El Paso, Texas—is among the worst in North America (Nuñez, Vickers and Emerson 1994). In 1999, the city of El Paso was classified by the US Environmental Protection Agency as a non-attainment area for carbon monoxide, particulate matter and ozone. Pollution in Ciudad Juárez—separated from El Paso only by the Rio Grande—is at least as bad. Poor air quality in the sister cities stems from rapid industrialization and population growth over the last several decades and from the fact that they are located in a high desert valley that fosters temperature inversions. Leading sources of air pollution include vehicle emissions, dust from unpaved roads, industrial pollution and open-air fires.

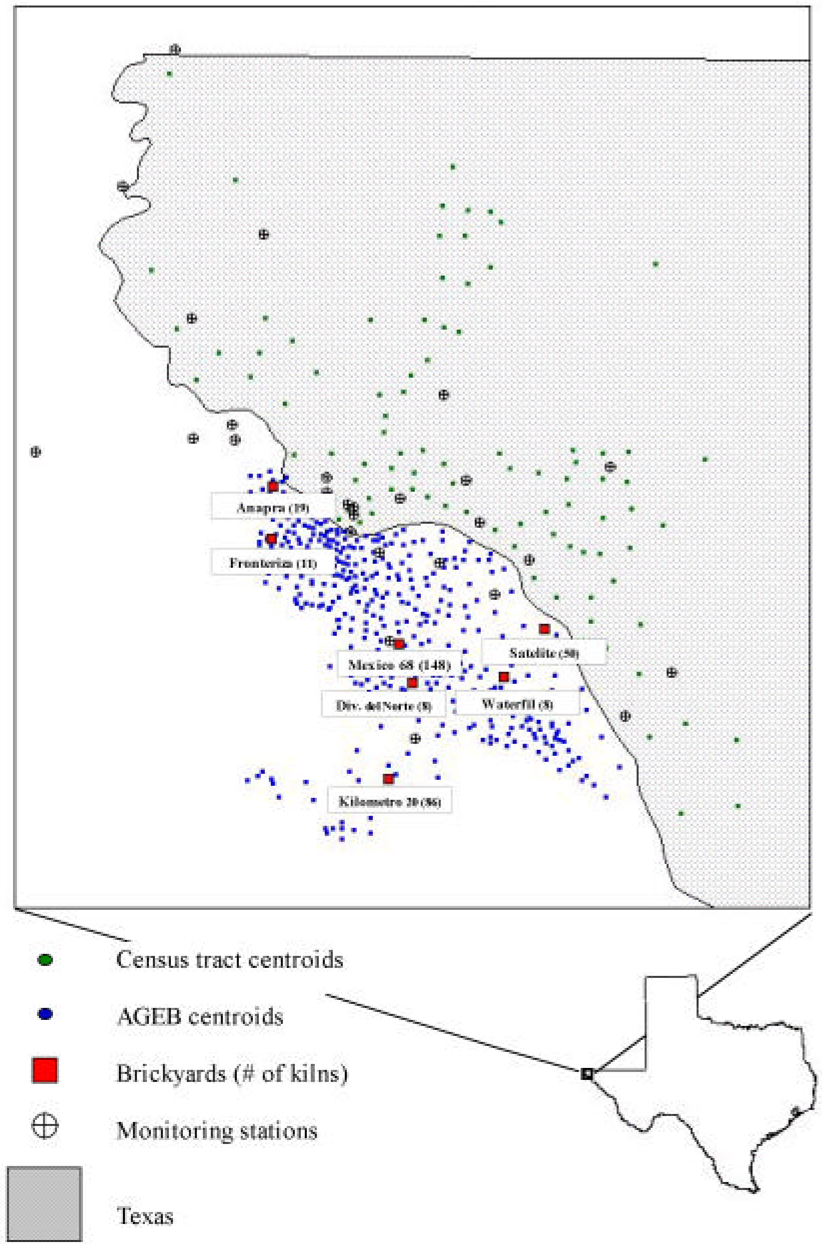
Among the emissions sources in Ciudad Juárez that have attracted attention from environmental regulators and advocates on both sides of the border are approximately 350 small-scale traditional brick kilns. Averaging 10 square meters, these primitive open-topped adobe kilns are principally fired with scrap wood and sawdust that is often impregnated with laminates and varnishes. Used tires, plastic containers and other types of refuse are often burned as well. The kilns are frequently cited as the third or fourth leading contributor to air pollution in Paso del Norte, a statistic that remains undocumented since no detailed emissions inventory is available for Ciudad Juárez. Brick kilns are primarily associated with carbon monoxide and particulate emissions but they may also emit volatile organic compounds, nitrogen oxide, sulfur dioxide, and heavy metals depending on the type of fuel burned (Johnson, Soto and Ward, 1994).

The location of the traditional brick kilns exacerbates their adverse impact on human health. They are clustered in seven poor *colonias* (neighborhoods) scattered throughout Ciudad Juárez: Anapra, Division del Norte, Fronteriza Baja, Kilometro 20, Mexico 68, Satelite, and Waterfill (Figure 1). When brickmakers squatted in these *colonias* 25 or 30 years ago, all were situated on the outskirts of the city. Today, however, most have been enveloped by urban sprawl.

Despite frequent complaints about emissions from traditional kilns, a number of factors make it politically difficult for environmental authorities to require brickmakers to bear the full costs of pollution control. First, brickmaking is a significant source of employment providing over 2,100 jobs (FEMAP, 1994). Second, it is an extremely small-scale, low-technology activity and, as a result, most brickmakers are impoverished. On average, each kiln employs six workers who perform all tasks by hand. Studies have put the brickmakers' monthly profit at between $60-$120 (RFF 1995, FEMAP 1991). Finally, the brickmakers are well-organized. Approximately two-thirds belong to trade associations or other local organizations (RFF survey).

There has only been one concerted effort to control emissions from Ciudad Juárez's brick kilns. In the early 1990s, a bi-national coalition led by a Mexican non-profit organization initiated an effort to convince brickmakers to substitute clean-burning propane for dirty traditional fuels (for a detailed account of the project, see Blackman and Bannister (1997)). The coalition used a number of carrots and sticks to encourage fuel switching: local propane companies provided free access to requisite equipment; universities supplied technical extension;

Figure 1. Locations of brickyards, survey unit centroids, and monitoring stations



the municipal government of Ciudad Juárez attempted to enforce a ban on dirty traditional fuels; and perhaps most important, local trade and community organizations pressured brickmakers to adopt propane. By October 1993, over half of the brickmakers in Ciudad Juárez were using propane exclusively. Unfortunately, within a year propane use was almost completely eliminated due to a nationwide economic liberalization program that cut long-standing subsidies on this fuel.

Today, traditional kilns continue to concern regulatory authorities in Ciudad Juárez and El Paso. The Joint Advisory Committee (JAC), an international institution created in 1996 to coordinate air pollution control policies in the sister cities, has identified the kilns as one among a number of potential targets for cooperative action (JAC 1999). The following four pollution control strategies have received considerable attention.

NMSU kilns. Researchers at New Mexico State University (NMSU) have designed a low-cost, low-technology pollution control technology that involves replacing traditional open-topped kilns with pairs of domed kilns connected by an underground tunnel fitted with clay-filled screens. Both kilns in each pair are loaded with uncooked bricks, but only one is fired at a time. During the firing, the screens in the underground tunnel and the uncooked bricks in the second kiln act as filtering system. NMSU kilns have been found to reduce emissions of particulate matter smaller than 10 microns (PM10)—the pollutant we focus on in this paper—by 99.5% (Avila et al. 1999). The reductions are quite high in part because the new kilns cut firing times by approximately 50%. This design is particularly promising because unlike other proposed kiln modifications it uses low-cost, readily-available materials.[1]

Natural gas. Natural gas burns as cleanly as propane but is considerably less expensive. Like propane, it can be used effectively in existing traditional kilns and requires minimal investments on the part of individual brickmakers. However, while propane can be distributed in mobile tanks, natural gas requires dedicated pipelines and decompressors, infrastructure that would have

[1] For a description of NMSU kilns, see Avila et al. (1999) and for a discussion of previous clean brick making technologies, see Blackman and Bannister (1997).

to be built to service the brickyards. We assume that switching to natural gas eliminates 99.9% of PM10 emissions.

Relocation. Moving kilns away from densely populated residential neighborhoods is frequently advocated as a means of reducing exposure to kiln emissions. There is some precedent for this strategy. In 1999, 16 brick kilns in a centrally located brickyard called Francisco Villa were relocated to Kilometro 20, the brickyard that is the furthest from Paso del Norte's population centers (Figure 1). Kilometro 20 is also the one brickyard in which land is plentiful. We model this scenario as a wholesale relocation of kilns to Kilometro 20.

No-Burn days. Since the transport of kiln emissions depends on weather conditions, requiring brickmakers to forego firing on certain days could significantly reduce exposure. El Paso currently has a "no-burn days" program that prohibits open-air fires on bad weather days, and there has been some discussion of enforcing such restrictions in Ciudad Juárez. We model this scenario as a prohibition on firing on days with low windspeed and high air stability—weather conditions correlated with high exposure (see the Appendix for a more detailed description of our methodology). Because enforcement of no-burn days is bound to be imperfect, we assume a compliance rate of 50%, that is, we assume only half of the kilns scheduled to fire on no-burn days actually forego firing.

3. Methods

Although brick kilns emit a variety of pollutants, we have chosen to focus on only on PM10 for several reasons. First, PM10 is generally thought to be responsible for a large proportion of the total non-carcinogenic adverse health impacts of air pollution (Pope et al. 1995). Also, data on the emissions of other types of air pollutants (e.g., toxics) from brick kilns is limited. Finally, the effects of PM10 on human health are relatively well-understood.

We have also chosen to focus only on one category of adverse effects of PM10—the effects on human morbidity and mortality. We do not consider the effects of PM10 on visibility, materials damages, or non-use values. Therefore, our estimates of the benefits of controlling brick kiln emissions are a lower bound on the total value of the benefits.

The benefit of each of the four pollution control strategies is the difference between the damages associated with uncontrolled PM10 emissions and the damages associated with controlled PM10 emissions. Estimating damages in each case entails the serial application of three models. First, we use a specially parameterized air dispersion model to estimate all brick kilns' combined contribution to annual average ambient levels of PM10 at several hundred receptor locations in Paso del Norte. Next, we use a health effects model to estimate the number of cases of human mortality and morbidity that result from this pollution each year. Finally, we use a valuation model to calculate the dollar values of these health impacts. Having estimated the annual benefits of each of our control strategies, we compare them to annualized costs. This section discusses each of these steps.[2]

3.1. Air dispersion model

We use the US EPA's Industrial Source Complex Short Term 3 (ISCST3) air dispersion model to estimate annual average concentrations of PM10 from brick kilns at a rectangular array of 5,546 receptor locations in the study area depicted in Figure 1. ISCST3 is a Gaussian plume model which uses data on emissions source characteristics, local meteorology and local topography to estimate hourly, daily, and annual concentrations of emissions in a defined study area. ISCST3 has been one of the US EPA's chief tools for investigating violations of ambient air quality standards (Riswadkar and Kumar, 1994).[3]

Table 1 details the data used to parameterize the ISCST3 model. Wherever possible we have used conservative parameters, that is, parameters that yield the lowest average annual ambient concentrations of PM10. As for meteorological data, the ISCST3 model uses one specific year's worth of hourly data on temperature, wind speed, wind direction, and mixing height. We use the most recent year of data available (1990) from the one local weather station that collects all of the requisite data (the El Paso International Airport). Note that we report probability distributions for several of the emissions source characteristics. As discussed in Section 3.4 below, these probability distributions are used to perform a Monte Carlo analysis that accounts for some of the uncertainty associated with source emissions characteristics.

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[2] Our health and valuation models draw on the Tracking and Analysis Framework (TAF), an integrated tool for benefit-cost analysis developed in part by Resources for the Future (Bloyd et al. 1996).

[3] Model selection was based on the Guideline on Air Quality Models (40 CFR 1997). More sophisticated 3-D models (such as CIT and UAM) require meteorological and emissions inventory data which are not available at this time. In validating a number of different air dispersion models, Patel and Kumar (1998) conclude that the ISCST model had the best overall performance.

Table 1. Air dispersion model inputs

Input	Unit	Parameter or distribution	Source
Emissions source characteristics			
Kiln radius (traditional kiln)	m.	1.75	Bruce (1999)
Kiln radius (modified kiln)	m.	0.37	Bruce (1999)
Kiln height	m.	3.0	Avila et al. (1999)
Emissions velocity (traditional kiln)	m./s.	0.5	Bruce (1999)
Emissions velocity (modified kiln)	m./s.	1.0	Bruce (1999)
Stack temperature (traditional kiln)	^oK	573	Bruce (1999)
Stack temperature (modified kiln)	^oK	333	Bruce (1999)
Peak emission rate total dry aerosols 0.5 to 20 microns in diameter[a]	g./sec	N(7.83, 2.89)	Bruce (1999)
Average emission rate / peak rate[a]	--	T(0.2, 0.3,0.4)	Bruce (1999)
Gr. PM10 / gr. total dry aerosols[a]	--	N(0.7, 0.1)	US EPA (1997)
Firings / month April to September[a]	--	2	Alfaro (2000a)
Firings / month October to May[a]	--	1	Alfaro (2000a)
Hours / firing (traditional kiln)[a]	hours	17	Alfaro (2000a)
Hours / firing (modified kiln)[a]	hours	8.5	Avila et al. (1999)
Number of kilns	--	See Fig. 1	Tarin (2000)
Location of kilns	^{o}	See Fig. 1	Valenzuella (2000)
Weather data			
Temperature (hourly)	^oK	--	NCDC (2000)
Wind speed	m./s.	--	NCDC (2000)
Random flow vector	^{o}	--	NCDC (2000)
Stability category	--	--	NCDC (2000)
Mixing height	m	--	US EPA (2000)
Topographical data	m	--	INEGI (1992);USGS (2000)

[a]Used to calculate the scaling factor defined in the Appendix.
$N(\mu, \sigma)$ = normal distribution with mean = μ and std = σ ; T(a,b,c)= triangular distribution.

3.2. Health effects model

To estimate exposure to the PM10 produced by brick kilns, we use population data at the survey unit level, that is, at the level of Areas Geoestadísticas Básicas (AGEBs) in Ciudad Juárez and census tracts in El Paso. We assign the inhabitants of each survey unit a distance-weighted average of PM10 concentrations predicted by the ISCST3 model at all receptor points within 800 meters of the survey unit centroid (see the Appendix for details). Next we estimate the health effects of this exposure using concentration-response (CR) coefficients reported in the epidemiological literature. CR coefficients indicate the expected change in the number of cases of some heath endpoint due to a marginal change in the ambient concentration of a particular air pollutant. We model nine different health endpoints:

- Mortalities (MORTs)
- Respiratory hospital admissions (RHAs)
- Emergency room visits (ERVs)
- Adult respiratory symptom days (ARSDs)
- Adult restricted activity days (ARADs)
- Asthma attacks (AAs)
- Child chronic bronchitis (CCBs)
- Child chronic cough cases (CCCs)
- Adult chronic bronchitis cases (ACBs)

We make the conventional assumption that these health effects are linear functions of PM10 exposure levels (see, e.g., US EPA 1999). This has the somewhat counterintuitive implication that every one ug/m^3 increase in concentration of PM10 has the same marginal health impact regardless of the baseline concentration of PM10. While some researchers have postulated that the baseline concentration of PM10 matters because the relationship between ambient levels of air pollution and human health entails thresholds, the evidence for such non-linearities is not very strong (Krupnick 1996). Hence, health effects are calculated as

$$H_i = CR_i \times \sum_{j=1}^{J} (\overline{C}_j \times Pop_j)$$

where,

H_i = number of cases of heath endpoint *i*
CR_i = the concentration response coefficient for health endpoint *i*
$\overline{C}_j$ = the estimated average annual concentration of PM10 from brick kilns for census unit *j*
Pop_j = the population in census unit *j*.

Table 2 presents the parameters used in the health effects model. Since, as discussed in the Section 3.3, mortality effects—not morbidity effects—dominate the total benefits estimates because of the relatively high monetary value assigned to the avoidance of premature mortality, by far the most important CR coefficient in Table 2 is that for mortality. There has been considerable controversy about the magnitude of this coefficient. In estimating the costs and benefits of the Clean Air Act, the US EPA chose to rely on a large-scale study that followed a sample population over time and found that a 10 ug/m^3 change in daily PM10 results in a 1.6% annual increase in the mortality rate (Pope et al. 1995; US EPA 1999). Other similar studies find much larger effects (e.g., Dockery et al. 1993). Nevertheless, we make the more conservative and more widely used assumption that a 10 ug/m^3 change in daily PM10 results in a 1% annual increase in the mortality rate. This CR coefficient is based on a host of US studies (Ostro 1994). A discussion of the remaining CR coefficients in Table 2 can be found in Chapter 8 of Bloyd et al. (1996).

A challenge in estimating morbidity damages is identifying a set of endpoints that reflects the full range of identified adverse health effects but that avoids double counting. For example, there is a potential for double counting if adult restricted activity days that result from relatively acute symptoms are also counted as adult respiratory symptom days which result from all types of symptoms. We have dealt with this issue in the conventional manner—by restricting some endpoints to subpopulations, subtracting potentially overlapping categories of endpoints, and carefully selecting how each endpoint is valued. The Appendix details our methodology.

3.3. Valuation model

To estimate the monetary values of health damages avoided by reducing PM10 emissions from brick kilns, we use a combination of the following: (i) willingness to pay (WTP) figures from the economics literature, i.e., a “benefits transfer” approach, (ii) estimates of the value of work loss days based on average daily wages in Ciudad Juárez and El Paso, and (iii) estimates of health care costs based on the value of work loss days (Table 3). Since over three-quarters of the total estimated benefits arise from premature mortalities avoided, by far the most important

Table 2. Health effects model inputs

	Parameter	Units	Value or distribution	Source
	CR coefficients			
$\mathbf{CR_{MORT}}$	Mortalities	%change mort. rate/(ug/m^3)	N(1m, 300u)	Krupnick (1996)
$\mathbf{CR_{RHA}}$	Respiratory hospital admissions	admiss./yr./(ug/m^3)/person	N(102u, 62.5u)	Plagiannakos and Parker (1988)[c]
$\mathbf{CR_{ERV}}$	Emergency room visits	visits/yr./(ug/m^3)/person	N(235.4u, 128.3u)	*Samet et al. (1981)*
$\mathbf{CR_{ARSD}}$	Adult symptom days	days/yr./(ug/m^3)/adult[a]	N(0.247, 0.059)	Krupnick, Harrington and Ostro (1990)
$\mathbf{CR_{ARAD}}$	Adult restricted activity days	days/yr./(ug/m^3)/non-asthmatic adult	N(0.0575, 0.0275)	Ostro (1987)
$\mathbf{CR_{AA}}$	Asthma attacks	attacks/day/(ug/m^3)/asthmatic person	N(912u,450u)	Ostro et al. (1991)
$\mathbf{CR_{CCB}}$	Child chronic bronchitis	cases/yr./(ug/m^3)/child[a]	N(1.59m, 805u)	Dockery, Speizer and Stram (1989)
$\mathbf{CR_{CCC}}$	Child chronic cough	cases/yr./(ug/m^3)/child[a]	N(1.84m, 924u)	Dockery, Speizer and Stram (1989)
$\mathbf{CR_{ACB}}$	Adult chronic bronchitis	cases/yr./(ug/m^3)/adult[a]	N(61.5u, 30.7u)	Abbey et al. (1993)
	Population data			
	El Paso 1995[b]	persons per census tract	--	US Census Bureau (1990)
	Ciudad Juárez 1995	persons per AGEB	--	INEGI (1995)
	Other parameters			
	Baseline mort. rate TX 1997	deaths/person/year	0.007345	CDC (1999)
	Baseline mort. rate Chih. 1997	deaths/person/year	0.005506	INEGI (2000)
	Fraction pop. asthmatic US		0.05	Bloyd et al. (1996)
	Fraction pop. asthmatic Mexico		0.05 (US rate)	Bloyd et al. (1996)
	Work loss days per ERV	days	1.28	Chestnut and Rowe (1988)
	Work loss days per RHA	days	9.30	Heart, Lung and Blood Institute (1992)

[a]Adults are defined as persons older than 17.
[b]Estimated for 1995 at the census tract level using the aggregate growth rate of El Paso's population based on 1995 Bureau of Census estimates of the total population of El Paso.
[c]We convert the relationship between annual RHAs and annual SO_4 concentrations in this study to PM10 using the "standard" ratio of SO_4 to PM10 reported in Lee et al. (1994, Part III).
m = 10^{-3}; u = 10^{-6}
$N(\mu,\sigma)$ = normal distribution with mean = μ and std = σ

parameter in the valuation model is the value of a statistical life. We use a discrete distribution—$1.9 million (33%), $3.8 million (34%), and $7.5 million (33%)—from Hagler Bailly, Inc. (1991). This distribution is relatively conservative. For example, the US EPA used a mean value of $4.8 million per mortality avoided to assess the benefits of the Clean Air Act (see US EPA 1999, Appendix H-8). The parameters used to value respiratory hospital admissions and emergency room visits are estimates of medical costs associated with each endpoint. These estimates are based on work-day-equivalent conversion factors taken from a study for Santiago, Chile (World Bank 1994). We also use conversion factors to estimate the value of child chronic cough. A discussion of the remaining valuation parameters can be found in Chapter 9 of Bloyd et al. (1996).

Table 3. Benefits valuation model inputs (1999 $US)

	Parameter (value of…)	Units	Distribution (μ,σ)	Source
VSL	statistical life	millions $/stat. Life	1.9 (33%); 3.8 (34%); 7.5 (33%)	Hagler Bailley, Inc. (1991)
Val_{RHA}	respiratory hospital admission	$/case	43.6 days*Val_{WLD}	World Bank (1994)
Val_{ERV}	emergency room visit	$/case	2.2 days*Val_{WLD}	World Bank (1994)
Val_{ARSD}	adult respiratory symptom day	$/case/individual	b(6)	Krupnick and Kopp (1988)
Val_{ARAD}	adult reduced activity day	$/case/individual	62.09	Krupnick and Kopp (1988)
Val_{AA}	asthma attack	$/case	U(13,62)	Rowe and Chestnut (1985)
Val_{CCB}	child chronic bronchitis	$/case	166	Krupnick and Cropper (1989)
Val_{CCC}	child chronic cough	$/case	0.12*Val_{WLD}	Ostro (1994)
Val_{ACB}	adult chronic bronchitis	$/case	N(9887,3666)	Krupnick and Cropper (1989)
Val_{WLD}	work loss day El Paso	$/day	89	US Bureau of Census (1998)
Val_{WLD}	work loss day Juárez	$/day	17	INEGI (1995)

$N(\mu,\sigma)$ = normal distribution with mean = μ and std = σ
U(a, b) = uniform distribution with lower bound a and upper bound b
b(a) = beta distribution with central value of a

Unfortunately, to our knowledge direct estimates of Mexican WTP for reductions in the health endpoints considered in this paper are not yet available. Therefore, we use WTP parameters (for adult respiratory symptom days, adult reduced activity days, asthma attacks, and chronic bronchitis) that are based on American studies. But given that average income adjusted for purchasing power parity is approximately four times higher in the United States than it is in

Mexico, Mexican WTP may be lower than American WTP.[4] Cultural factors may also cause WTP in the two countries to differ. To account for international differences in WTP, we use sensitivity analysis. For each health impact, we use three different values for Mexican WTP based on three different assumptions about the elasticity of WTP with respect to income, a parameter we will call E.[5] We assume alternatively that E = 1, E = 0.33 and E = 0. For example, E = 0.33 implies that if average per capita income adjusted for purchasing power parity is 10% lower in Ciudad Juárez than in El Paso, then WTP is 3.3% lower. An E between 0.2 and 0.5 is supported by some studies that look at differences in WTP across income groups (Alberini et al. 1997, Loehman et al. 1979). Thus, the middle value of the discrete probability distribution we use to value premature mortality in Mexico is \$3.80 million assuming E = 0, \$2.42 million assuming E = 0.33, and \$0.97 million assuming E = 1.

3.4. Uncertainty

We use Monte Carlo analysis to account for uncertainty associated with the parameterization of our air dispersion, health impacts, and benefits valuation models. That is, where data on probability distributions is available, we treat model parameters as distributions and we use these distributions to generate (95%) confidence intervals for model outputs. In our air dispersion model we have data on probability distributions of the peak emission rate of total dry aerosols, the ratio between peak and average emissions rates and the fraction of total aerosols that is PM10. All are parameters used to calculate the "scaling factor' described in the Appendix.[6] In our health impacts model, for each of our nine concentration response coefficients, we use mean values and standard deviations reported in the literature, and—unless the literature dictates otherwise—assume normal distributions. Finally, for our benefits valuation model, we have information on the probability distributions of the value of a statistical life, the value of an adult respiratory symptom day, the value of an asthma attack, and the value of adult chronic bronchitis.

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[4] In 1998, per capita gross national product adjusted for purchasing power parity was \$29,240 in the United States and \$7,450 in Mexico (World Bank 2000).

[5] The empirical foundations of this second-best approach to estimating international differences in WTP can be legitimately questioned. Evidence on the topic is sparse. Chestnut, Ostro and Vichit-Vadakan (1999) find that median WTP to avoid respiratory symptoms is higher in Thailand than one would expect from US studies. See also Alberini, et al. (1997).

[6] Including only parameters used to calculate the scaling factor avoids iterative runs of the ISCST3 model which would be extremely computationally intensive.

3.5. Costs of pollution control strategies.

Table 4 gives the annualized costs of each of the control strategies. We consider a uniform application of these strategies across brickyards and within brickyards. For example, for the natural gas strategy, we assume all kilns in all seven brickyards switch.[7] For the time being, we consider only capital costs and operations and maintenance costs and we neglect transactions costs such as the costs involved in convincing brickmakers to relocate or adopt new technologies. For the NMSU kilns strategy, the sole cost is that of building a pair of modified kilns. The ten-year lifetime is based on a 1995 RFF survey of 100 brickmakers, which found that the average age of kilns then in use was eleven years. For the natural gas strategy, capital costs per kiln are based on RFF survey data on the conversion of kilns to propane in the early 1990s. Like conversion to natural gas, conversion to propane requires investments in a burner and modifications to enable the kiln to withstand higher temperatures. Our assumption that the infrastructure has a 20-year useful life is quite conservative. For the relocation strategy, we assume that a conventional kiln costs $1,500 to build and has a ten-year lifetime and that relocation costs amount to an additional $4,500. Our costs for the no-burn days strategy are based on the administrative costs of a similar program in El Paso. This program entails labor costs only. Specifically, one person hour per day is devoted to monitoring weather data, and five person days are devoted to enforcement for each no-burn day declared. We assume Mexican regulatory labor costs $40,000 per person year.

[7] In theory, these strategies could be combined to maximize health benefits subject to a fixed budget for control costs (or equivalently to minimize the control costs given a health effects goal). However, allowing for this type of flexibility becomes quite involved as it entails complex feasibility constraints on various combinations of strategies.

Table 4. Annualized costs of pollution control strategies (1999 $US)

Cost category	NMSU kilns	Natural gas	Relocation	No-burn days
Capital				
Present value per kiln	3,000[a,b]	349[c]	6,000[a,b]	0
Present value infrastructure	0	1,002,005[a]	0	0
Lifetime of capital	10[e]	20	10[e]	--
Annualized costs (r = 12%)	175,214	149,553	350,429	0
O&M (annual)				
Per kiln	0	0	0	0
Infrastructure	0	100,000[d]	0	24,692[c]
Total annualized costs	175,214	249,553	350,429	24,692

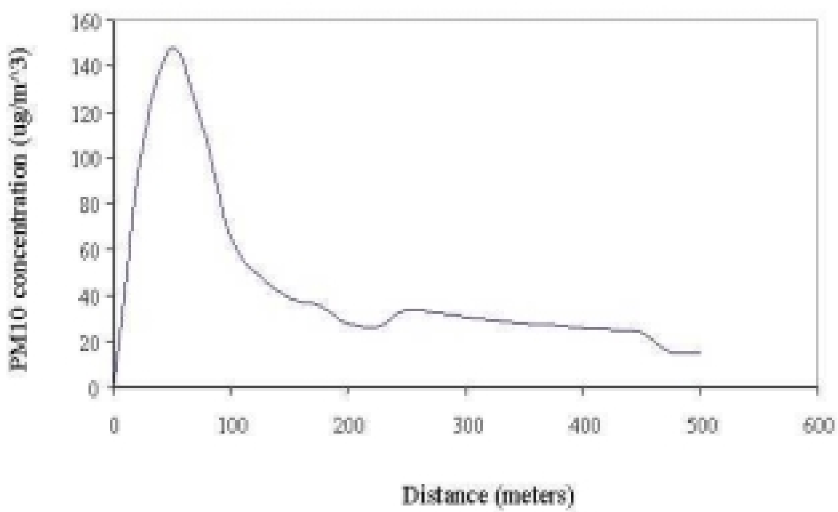
[a]Alfaro (1995) and (2000b).
[b]Marquez (2000).
[c]Reynoso (2000).
[d]Johnson(2000).
[e]RFF (1995).
Note**:** all cost estimates neglect transactions costs.

Note that we ignore the effects of two of the control strategies on variable production costs. The NMSU kilns strategy would reduce variable costs since it cuts firing times, and therefore fuel consumption, by approximately 50%. The natural gas strategy would raise variable costs since natural gas is more expensive than traditional fuel. We neglect these effects because we assume that given uniform application of these control strategies across all brickyards, prices in the highly competitive market for bricks would increase or decrease to offset the changes in variable costs.

4. Results

The air dispersion model results suggests that brick kilns' impacts are highly localized geographically. Figure 2, a concentration profile for PM10 emissions from the brickyard Mexico 68 on a due north transect, shows that virtually all of a brick kiln's PM10 emissions are deposited less than 500 meters away. This result stems from the fact that brick kilns have very low "stack-heights" and low emissions velocities.

Figure 2. Profile of brick kiln PM10 concentration downwind of Mexico 68



Combining the predictions of the air dispersion model with our population model to estimate annual human exposure to PM10 [individuals*ug/m^3/year] yields two interesting results. First, as the last column in Table 5 illustrates, even though El Paso and Ciudad Juárez are in close proximity, and even though the prevailing winds blow north for most of the year, total exposure to brick kiln PM10 emissions in El Paso is about one-seventh of total exposure in Ciudad Juárez. This result is not surprising given that most kiln emissions are deposited locally. Second, certain brickyards are more important sources of exposure than others. The main reasons are that some brickyards have more kilns and are located in more densely populated areas than other brickyards.

Table 5. Total annual exposure to uncontrolled brick kiln emissions (individuals*ug/m^3/year)

Brickyard	Cd. Juárez	El Paso
1-Kilometro 20	284,900	46,060
2-Division del Norte	88,490	7,370
3-Waterfill	78,940	8,832
4-Mexico 68	1,733,000	156,500
5-Satelite	262,600	105,800
6-Fronteriza Baja	38,460	9,506
7-Anapra	69,760	22,270
Total	2,556,000	356,400

(Source: RFF model)

The health effects model suggests that brick kilns have very significant impacts on mortality and morbidity (Table 6). Most important, the model's mean prediction is that brick kiln emissions are responsible for 14 premature mortalities per year in Ciudad Juárez and 3 premature mortalities per year in El Paso. The higher levels of morbidity and mortality in Ciudad Juárez stem from the fact that, as discussed above, most PM10 emissions from brick kilns are deposited locally. Note that the predicted health effects pass a reality check in the sense that they are related to each other in a reasonable way. For instance, the model predicts twice as many emergency room visits as respiratory hospital admissions, and over twice as many adult respiratory symptom days as adult restricted activity days.

Table 6. Annual health effects of uncontrolled PM10 emissions from brick kilns (mean values and 95% confidence intervals)

Health endpoint	Cd. Juárez			El Paso		
(number of cases)	Low	*Mean*	*High*	*Low*	*Mean*	*High*
Mortality	2.5	14.1	31	0.5	2.6	5.8
Respiratory hospital admissions	0	262	770	0	37	107
Emergency room visits	0	607	1,719	0	85	240
Work loss days	0	3,216	8,500	0	448	1,185
Adult respiratory symptom days	91,610	376,600	794,000	14,430	59,300	125,000
Adult restricted activity days	2,704	138,000	349,100	377	19,240	48,670
Asthma attacks	180	42,680	108,600	25	5,950	15,130
Child chronic bronchitis	0	1,637	4,416	0	184	497
Child chronic cough	0	1,878	5,017	0	211	564
Adult chronic bronchitis	0	93	242	0	15	38

(Source: RFF model)

Table 7 gives our estimates of the monetized annual benefits of completely eliminating PM10 emissions from brick kilns. For El Paso, the mean total benefit is approximately $13 million. For Ciudad Juárez, we report three sets of mean benefit estimates based on different assumptions about the elasticity of WTP with respect to income. Even when this elasticity is assumed to be quite high (E = 1.0), the mean total benefit estimate is over $19 million. Assuming a middle elasticity value (E = 0.33), the mean total benefit is $47 million. Assuming that Mexican and American have the same WTP (E = 0), the mean total benefit to Ciudad Juárez is over $74 million. Note that reduced mortality is by far the largest component of benefits, accounting for over 80% of the total. Assuming a middle value for E, total mean benefits are about four times greater in Ciudad Juárez than in El Paso.

Table 7. Annual value of eliminating uncontrolled PM10 emissions from brick kilns (1999 US $; mean values and 95% confidence intervals)

Health Endpoint	**Cd. Juárez**					**El Paso**		
	(E = 1.0)	**(E = 0.33)**			**(E = 0)**			
	Mean	*Low*	*Mean*	*High*	*Mean*	*Low*	*Mean*	*High*
Mortality	15,720,000	4,398,000	39,280,000	119,000,000	61,680,000	1,284,000	11,470,000	34,750,000
Respiratory hospital admissions	194,400	0	194,400	571,000	194,400	0	141,900	416,800
Emergency room visits	22,690	0	22,690	64,270	22,690	0	16,560	46,910
Work loss days	54,680	0	54,680	144,500	54,680	0	39,910	105,500
Adult respiratory symptom days	224,400	1,085	560,900	2,158,000	880,700	1,631	148,100	559,700
Adult restricted activity days	2,183,000	106,900	5,458,000	13,800,000	8,570,000	23,400	1,195,000	3,022,000
Asthma attacks	408,200	5,244	1,020,000	3,114,000	1,602,000	1,148	223,400	681,600
Children's chronic bronchitis	69,250	0	173,100	466,800	271,800	0	30,570	82,430
Children's chronic cough	3,830	0	3,830	10,230	3,830	0	2,255	6,026
Adult chronic bronchitis	236,000	0	589,900	1,820,000	926,200	0	145,800	450,100
Total benefits	19,110,000	4,460,000	47,360,000	141,200,000	74,210,000	1,282,000	13,410,000	40,130,000

E = the elasticity of WTP (willingness to pay) with respect to income adjusted for purchasing power parity.
(Source: RFF model)

Table 8 presents estimates of the annual net benefits of each of the four pollution control strategies for Ciudad Juárez, El Paso and the combined metropolitan area. For Ciudad Juárez, we allow for three different values of E. For each scenario we consider—that is, for each combination of target location and E—the ranking of the control strategies is the same: the net benefits of natural gas and NMSU kilns are virtually the same, the net benefit of relocation is about half that of natural gas, and the net benefit of no-burn days is about one fiftieth that of natural gas. Natural gas and NMSU kilns have the highest net benefits because they are most effective at reducing PM10 emissions. Even though natural gas entails higher control costs than NMSU kilns (owing mainly to operation and maintenance costs), the two strategies yield approximately the same net benefits because natural gas is more efficient at eliminating PM10 emissions. Relocation is ranked third because the benefits of this strategy are about half those for natural gas, while the costs are the highest of any of the four strategies. Recall that most of the costs for this strategy arise from relocating brickmakers' homes. The no-burn days strategy is ranked last because it generates the lowest benefits—less than one hundredth of those associated with natural gas.

For Ciudad Juárez, the mean annual net benefits of all four strategies are quite large for each of the three assumptions about the value of E. Assuming the middle value of E = 0.33, the mean annual net benefits range from $46,810,000 for natural gas to $905,000 for no-burn days. Even the lowest estimates—those at the low end of the 95% confidence interval given E = 1—are positive for all of the control strategies.

For El Paso, the mean annual net benefits for three of the control strategies—natural gas, NMSU kilns and relocation—are quite significant, ranging from $13,151,000 to $6,407,000. Even the low-end estimates for these strategies are positive. However, the mean annual net benefit of no-burn days is just $15,000 and the low-end estimate is negative.

For the combined metropolitan area, assuming E = 0.33, the mean annual net benefits of all four strategies are large, ranging from $60,137,000 to $945,000. Even the low-end net benefits are well above zero.

Table 8. Annual net benefits of pollution control strategies for traditional brick kilns (benefits less costs in 1999 $US)

Location	E	Scenario	Natural gas	NMSU kilns	Relocation	No-burn days
			Cost = 249,553	Cost = 175,214	Cost = 350,429	Cost = 24,692
Cd. Juárez	0	High	220,729,247	219,532,786	126,279,571	4,275,308
		Mean	**73,886,237**	**73,447,986**	**42,139,571**	**1,435,308**
		Low	6,765,425	6,745,386	3,668,571	114,308
	0.33	High	140,809,247	140,072,486	80,499,571	2,775,308
		Mean	**47,063,087**	**46,810,286**	**26,759,571**	**905,308**
		Low	4,205,987	4,220,326	2,202,571	63,308
	1	High	56,643,497	56,390,686	32,249,571	1,095,308
		Mean	**18,841,337**	**18,783,686**	**10,588,571**	**345,308**
		Low	1,512,683	1,563,176	659,071	9,308
El Paso	--	High	39,840,317	39,741,286	19,879,571	125,308
		Mean	**13,147,037**	**13,151,066**	**6,406,571**	**15,308**
		Low	1,031,165	1,092,206	295,571	-20,692
All	0.33	High	180,899,117	179,988,986	100,729,571	2,925,308
		Mean	**60,459,677**	**60,136,566**	**33,516,571**	**945,308**
		Low	5,486,705	5,487,746	2,848,571	67,308

E = the elasticity of WTP (willingness to pay) with respect to income adjusted for purchasing power parity. (Source: RFF model)

Hence from the point of view of a social planner in Ciudad Juárez only concerned about Mexican human health, the benefits of each of the control strategies are much larger than the costs. From the point of view of a social planner in El Paso only concerned about American human health, the benefits of each of the control strategies except no-burn days are much larger than the costs. This implies that it is worthwhile for Americans to subsidize pollution control efforts in Ciudad Juárez.

5. Conclusion

We have used a specially parameterized air dispersion model in combination with benefits transfer methods to estimate the net benefits of four pollution control strategies for informal brick kilns in Ciudad Juárez. We found that given a wide range of modeling assumptions, the benefits of three control strategies—NMSU kilns, natural gas and relocation—are considerably higher than the costs. We also found that even though brick kilns have significant health impacts in El Paso, the lion's share of the damages they inflict are concentrated in neighborhoods surrounding the brickyards. What are the policy implications of these findings?

If pollution control resources were allocated purely on the basis of whether net benefits are positive, targeting informal brick kilns would clearly be advisable. But given that regulatory resources are scarce, policy-makers cannot use this decision criteria. Rather, they must compare the net benefits of pollution control efforts across different types of emissions sources, including large industrial sources that are the traditional focus of pollution control efforts. While such a comparison is beyond the scope of this paper, the magnitude of the net benefits associated with control strategies for brick kilns strongly suggests that these sources should be a high priority for local policy-makers. Future research directly comparing the net benefits of controlling emissions from informal sources and large industrial sources would likely make a definitive case for amending current regulatory practice.

Our finding that exposure to brick kiln emissions is spatially concentrated suggests that, in general, informal polluters may have disproportionate impacts in the low-income neighborhoods where they are typically found. Finally, as noted above, our finding that brick kilns have significant adverse health effects in El Paso indicates that it may be worthwhile for Americans to subsidize pollution control efforts in Ciudad Juárez, a policy that Mexico should find especially attractive given that its citizens would be the main beneficiaries.

Appendix

A. Selection of no-burn days

To select no-burn days, we first identified weather conditions that were correlated with high exposure to kiln emissions in 1990, our sample weather year. We did this by regressing total daily exposure from brick kilns predicted by our air dispersion model onto actual daily averages of five weather variables. We found that windspeed was significantly negatively correlated with exposure, and stability was significantly positively correlated with exposure. Next we identified all those days in our sample on which windspeed was at least one standard deviation below the annual average *and* stability was at least one standard deviation above the annual average. Using this criteria, we selected 23 days.

B. Scaling factor used in air dispersion model

We used an arbitrary emissions rate to estimate average annual PM10 concentrations at designated receptor locations and then used a scaling factor to adjust these concentrations to reflect real emission rates. The scaling factor was calculated as

emissions rate of PM10 per kiln (g/sec) =

* [peak emissions rate total dry aerosols (g./sec)]

* [average rate / peak rate]

* [g. PM10 / g. total dry aerosols]

* [firings / month]

* [hours / firing]

C. Distance-weighted average used in exposure model

The PM-10 concentration assigned to the inhabitants of each survey unit was calculated

as

$$\overline{C}_j = \frac{\sum_r \frac{C_r}{d_{j,r}}}{\sum_r \frac{1}{d_{j,r}}} \quad \forall \ r \text{ where } d_{j,r} \leq 800 \text{ meters},$$

where

$\overline{C}_j$ = a distance-weighted average annual concentration of PM10 from brick kilns for survey unit *j*

C_r = the average annual concentration of PM10 from brick kilns at receptor location *r* predicted by the ISCST3 model, and

$d_{r,j}$ = the distance between receptor location *r* and the centroid of survey unit *j*.

D. Methods used to avoid double counting health damages

To avoid double counting adult restricted activity days and asthma attacks, we restrict the former to non-asthmatic adults. To avoid double counting adult respiratory symptom days, adult restricted activity days and asthma attacks, we define adult respiratory symptom days as the difference between: (i) total adult respiratory symptom days estimated by the coefficient reported in Table 2, and (ii) the sum of adult restricted activity days and asthma attacks days experienced by adults. To avoid double counting respiratory hospital admissions and emergency room visits we use estimated medical costs to value these endpoints (a necessity in any case since WTP estimates for these endpoints do not exist). Note that most emergency room visits do not actually result in respiratory hospital admissions, and hospitals typically charge for each separately. We avoid double counting adult restricted activity days and hospital visits because the data base used to estimate the CR coefficient for restricted activity days omits hospital and emergency room days. Finally, note that the potential for double counting asthma attacks and hospital visits is small since an estimated 0.5% of asthma attacks actually result in hospitalization (Bloyd et al. 1996, 9-18).

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