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The benefits and costs of informal sector pollution control: Mexican brick kilns

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ABSTRACT. In developing countries, the rapid proliferation of informal firms – low-technology unlicensed micro-enterprises – is having significant environmental impacts. Yet environmental management authorities typically ignore such firms. This paper estimates the annual net benefits (benefits minus costs) of controlling particulate emissions from a collection of informal brick kilns in Ciudad Juárez, Mexico and from two of the city's leading formal industrial polluters. We find that the annual net benefits of controlling brick kiln emissions are substantial – in the tens of millions of dollars – and exceed those for the two formal industrial facilities by a significant margin. These results suggest that, in some cases, the conventional allocation of pollution control resources across formal and informal polluters may be suboptimal.

1. Introduction

In developing countries, population growth, rural–urban migration, and regulation have spurred the rapid expansion of an urban informal sector comprised of low-technology micro-enterprises largely operating outside the purview of the state. Today, the informal sector accounts for over half of non-agricultural employment and contributes between a quarter and three-quarters of gross domestic product in both Latin America and Africa (Ranis

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and Stewart, 1994; Schneider and Enste, 2000). Once popularly viewed as an economic backwater – a collection of street merchants transitioning to salaried jobs – the informal sector is now recognized as a hotbed of entrepreneurship and innovation (De Soto, 1989).

Unfortunately, for all its economic benefits, there are good reasons to suspect that the expansion of the informal sector is having significant environmental impacts. The main reason is that informal firms, although tiny, are exceptionally numerous, and a significant percentage are involved in highly polluting activities such as leather tanning, ceramics, metalworking, electroplating, and mining. For example, in Mexico, 38 per cent of informal firms are classified as industrial (STPS/DOL, 1992). Also, informal firms generally lack pollution control equipment, and often access to basic sanitation services. Finally, such firms are typically labor intensive and situated in poor residential areas. Therefore, their emissions directly affect a considerable population of employees and neighbors.

Nevertheless, policy makers have thus far paid little attention to informal polluters. Industrial environmental management efforts in developing countries have generally focused exclusively on large formal facilities. In part, this bias stems from the perceived difficulty of regulating numerous small firms. But a second reason may simply be that the problem is not well-understood – policy-makers lack information on the magnitude and incidence of the environmental damages caused by informal firms, and on the costs of mitigating these damages. Little research has been conducted to fill this gap. Collecting the requisite data is difficult since informal firms are, by their nature, wary of record keeping and monitoring.

This paper presents a benefit–cost analysis of four strategies for reducing air pollution from a collection of approximately 330 informal brick kilns in Ciudad Juárez, Mexico. Our analysis takes advantage of data on US–Mexico border environmental problems collected in the wake of the 1992 North American Free Trade Agreement. We also compare the net benefits (benefits minus costs) of controlling brick kiln emissions to estimates of the net benefits of controlling emissions from two of the city's leading formal industrial polluters.

We find that brick kilns emissions are responsible for serious health damages including over a dozen premature mortalities per year. As a result, the annual net benefits of three of the four emissions control strategies for brick kilns are positive and quite large – in the tens of millions of dollars. We also find that the net benefits of controlling emissions from brick kilns exceed net benefits of controlling emissions from formal factories by a considerable margin, although the size of this margin depends critically on the actual level of pollution abatement in formal factories. These findings suggest that, in some cases, the conventional allocation of regulatory resources across formal and informal polluters may be suboptimal.

A number of caveats are in order. First, to make the analysis tractable, we neglect several factors. Although the emissions sources we analyze generate a variety of pollutants, we focus solely on particulate matter smaller than 10 microns (PM10) because it is thought to be responsible for a large proportion of the total non-carcinogenic adverse health impacts of air pollution (Pope *et al.*, 1995), and because its effects on human health are relatively

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well-understood. In addition, we neglect both secondary PM10 (formed when pollutants such as nitrogen oxides and sulfur dioxide undergo chemical reactions in the atmosphere) and long-range transport of PM because, as discussed in Section 5.1, neither is likely to be important in our case study. Also, we limit our attention to the effects of PM10 on human morbidity and mortality. We do not consider the effects on visibility, materials damages, or non-use values. Finally, we restrict attention to human morbidity and mortality in Ciudad Juárez and omit estimates of the (far less severe) damages in El Paso, Texas – Juárez's nearby sister city.[1] Given that we restrict attention to one type of pollutant, one category of adverse impacts, and a subset of the affected population, our estimates of the net benefits of controlling air pollution are a lower bound on the total value of the net benefits.

Second, data limitations constrain our analysis somewhat. Most importantly, data on Mexican formal industrial facilities are extremely tightly held. Therefore, as discussed below, PM10 emissions and abatement costs for the two formal factories in our sample are estimated rather than measured. These estimated data are adequate for our purpose – to convey a general sense of how net benefits of controlling emissions from formal factories compare with the (more precisely measured) net benefits of controlling emissions from informal firms.

Third, our analysis of abatement costs ignores implementation costs – the costs regulators pay to compel firms to abate pollution – because these costs are highly uncertain and difficult to measure. In the informal sector, implementation costs are likely to be significant. We return to this issue in the conclusion.

To our knowledge, ours is the first rigorous analysis of the benefits and costs of informal sector pollution control. The literature on informal sector pollution problems is quite thin. Most of it relies on case studies and focuses either on distinguishing between successful and unsuccessful environmental management strategies (Blackman, 2000; Biller and Quintero, 1995; Perera and Amin, 1996) or on identifying the drivers of informal firms' environmental performance (Blackman and Bannister, 1998). The literature on the link between small and medium enterprises (SMEs) and the environment is somewhat more robust. Policy levers are, again, the main focus (Kennedy, 1999; World Bank, 1998). Although we know of no rigorous benefit–cost analyses, a number of studies assess the magnitude and incidence of SME pollution. For example, Lanjouw (1997) finds that SMEs are responsible for over 90 per cent of total water pollution associated with six important economic sectors in Ecuador (see also, Bartone and Benavides, 1997 and Dasgupta *et al.*, 2002).

To understand the organization of the paper, it is helpful to provide a brief overview of the four-step method used to estimate the net benefits of investing in a specific 'control strategy' to cut PM10 pollution from an emissions source. First, we use a specially parameterized air dispersion

[1] Levels of morbidity and mortality due to brick kiln PM10 in El Paso are about a seventh of levels in Ciudad Juárez. For a longer version of the paper, which includes El Paso in estimates of health damages and discusses the cross-border implications of the benefit–cost analysis, see Blackman *et al.* (2003).

Table 1. *Sectoral contributions to total anthropogenic air pollution in Ciudad Juárez*

Sector	*Pollutant*					
	PM	*SO_2*	*CO*	*NO_x*	*HC*	*All*
Informal brickmaking (%)	16	43	0	0	2	1
Other industry (%)	14	17	0	5	3	1
Services (%)	3	2	0	3	23	4
Transportation (%)	68	38	99	92	72	95
All (%)	100	100	100	100	100	100
Total (tons)	1,510	4,144	452,762	26,115	76,134	560,667

Notes: PM = particulate matter; SO_2 = sulfur dioxide; CO = carbon monoxide; NO_x = nitrogen oxides; HC = hydrocarbons.
Source: Gobierno del Estado de Chihuahua (1998).

model to gauge the extent to which the control strategy improves air quality in Ciudad Juárez. Second, we use a health effects model to estimate how many cases of human mortality and morbidity are avoided each year as a result of this improved air quality. Third, we use a valuation model to calculate the dollar value of the avoided mortality and morbidity. Finally, to arrive at net benefits, we calculate the annualized costs of the control strategy and subtract them from our estimate of the dollar value of benefits.

The remainder of the paper is organized as follows. Section 2 briefly discusses air pollution in Ciudad Juárez and presents aggregated emissions inventory data to demonstrate that the city's informal brick kilns are a leading source of air pollution. Section 3 provides background on informal brick kilns and presents data needed to estimate the annual net benefits of controlling PM10 emissions from these sources, including emissions characteristics given four different control strategies and the annualized costs of each strategy. Section 4 provides similar information for two formal sector facilities. Section 5 discusses the air dispersion, health impacts, and valuation models used to estimate the benefits of pollution abatement. Section 6 presents our results, and the last section considers the policy implications of our findings.

2. Air pollution in Ciudad Juárez

A sprawling industrial city with a population of over 1.2 million, Ciudad Juárez, has the worst air pollution on the US–Mexico border. The city violates Mexican federal ambient air quality standards for PM10, ozone, and carbon monoxide. According to official 1996 emissions inventory data – the Sistema Nacional de Información de Fuentes Fijas (SNIFF) for the state of Chihuahua (Gobierno del Estado de Chihuahua, 1998) – automobiles and trucks are the leading source of anthropogenic air pollution in Ciudad Juárez (table 1).[2] Unfortunately, for a variety of technological and political

[2] Naturally occurring particulates from wind erosion and unpaved roads are an important source of particulate matter. However, such particulate matter

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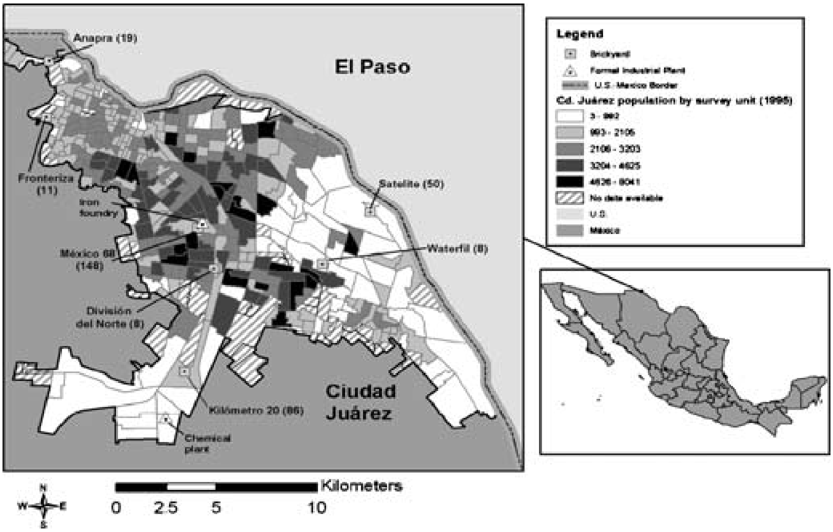

Figure 1. *Brickyards, formal industrial facilities, and population in Ciudad Juárez*

reasons, such sources are notoriously difficult to control (Harrington and McConnell, 2003). Industry is also a leading source of air pollution. Although Ciudad Juárez is home to over 250 maquiladoras – foreign-owned plants that have located in the city to reduce labor costs – surprisingly, a collection of 330 informal brick kilns are the city's leading source of industrial air pollution. They contribute 16 per cent of all particulate matter pollution, and 43 per cent of all sulfur dioxide. These statistics alone suggest that in Ciudad Juárez, the informal sector deserves serious consideration as a potential target for pollution control efforts.

3. Informal brick kilns

3.1 Description

Ciudad Juárez's informal brick kilns mainly supply construction companies specializing in low-cost housing. The typical kiln is a 10-meter-square primitive adobe structure that holds 10,000 bricks, employs five or six people, generates profits on the order of $100 per month, and is fired two times a month with scrap wood, sawdust, and other rubbish. Brick kilns use no pollution control devices whatsoever (Blackman and Bannister, 1997). The location of the kilns within the city exacerbates their adverse impact on human health; they are clustered in seven poor neighborhoods most of which are residential (figure 1).[3] Past efforts to control emissions

is principally comprised of large particles, which are relatively benign epidemiologically. Smaller particulates related to combustion are much more dangerous because they are inhaled deeply into the lungs.

[3] When brickmakers squatted in these neighborhoods 25 or 30 years ago, all were situated on the outskirts of the city. Today, however, most have been enveloped by urban sprawl.

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Table 2. *Brick kiln emissions characteristics*

Characteristic	*Unit*	*Parameter or distribution*	*Source*
Kiln radius (traditional kiln)	m	1.75	Bruce (1999)
Kiln radius (NMSU 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 (NMSU kiln)	m/s	1.0	Bruce (1999)
Plume temperature (traditional kiln)	°K	573	Bruce (1999)
Plume temperature (NMSU kiln)	°K	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 (1995)
Firings/month April to September[a]	–	2	Alfaro (p.c., 2000)[b]
Firings/month October to May[a]	–	1	Alfaro (p.c., 2000)[b]
Hours/firing (traditional kiln)[a]	hours	17	Alfaro (p.c., 2000)[b]
Hours/firing (NMSU kiln)[a]	hours	8.5	Avila *et al.* (1999)
Number of kilns	–	See Fig. 1	Tarin (p.c., 2000)[b]
Location of kilns	°	See Fig. 1	Valenzuela (p.c., 2000)[b]

Notes: [a]Used to calculate total emissions.
[b]Personal communication with authors.
$N(\mu,\sigma)$ = normal distribution where μ= mean and σ= standard deviation;
$T(m_1,m_2,m_3)$ = triangular distribution with m_1 = minimum, m_2 = mode, m_3 = maximum.

from Ciudad Juárez's brick kilns have mainly focused on encouraging brickmakers to substitute clean-burning propane for dirty traditional fuels. Unfortunately, these efforts have been undermined by rising propane prices (see Blackman and Bannister, 1997).

3.2 Data

Two types of data are required to estimate the net benefits of reducing PM10 emissions from informal brick kilns: data on the emissions characteristics of brick kilns (e.g., dimensions, locations, emissions rates, emissions velocities, and plume temperatures), and data on the efficacy and costs of appropriate emissions control strategies. As a general rule, whenever possible, we have chosen relatively conservative data that avoids any upwards bias in net benefits; that is, data that yield the lowest average annual ambient concentrations of PM10 and the highest annualized costs. Data on brick kiln emissions characteristics were obtained from studies conducted by local universities and from interviews with local stakeholders (table 2).

Note that we report probability distributions for several parameters. As discussed in section 5.4 below, these probability distributions are used to perform Monte Carlo simulations that account for some of the uncertainty associated with source emissions characteristics.

As for data on emissions abatement, we model the four control strategies that have received considerable attention from local stakeholders (see, e.g., JAC, 1999).

NMSU kilns. Researchers at New Mexico State University (NMSU) have designed a low-cost, low-technology pollution control strategy that involves replacing traditional open-topped kilns with pairs of domed kilns connected by an underground tunnel fitted with clay-filled screens. NMSU kilns have been found to reduce emissions of PM10 by 99.5 per cent (Avila *et al.*, 1999). This design is particularly promising because it cuts fuel costs by approximately 50 per cent.

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 a minimal investment on the part of individual brickmakers. However, while propane can be distributed in portable tanks, natural gas requires dedicated pipelines and decompressors – infrastructure that would have to be built to service the brickyards. We assume that switching to natural gas eliminates 99.9 per cent of PM10 emissions.

Relocation. Moving kilns away from densely populated residential neighborhoods is frequently advocated as a means of reducing exposure to kiln emissions. In 1999, 16 brick kilns in a centrally located brickyard called Francisco Villa were moved to Kilometro 20, the brickyard furthest from Ciudad Juárez's population centers and the one brickyard in which land is plentiful (figure 1). We model this scenario as a wholesale relocation of all the kilns in Ciudad Juárez to Kilometro 20.

No-burn days. Since the transport of kiln emissions depends on weather conditions, requiring brickmakers to forego firing on certain days can significantly reduce exposure to these emissions. El Paso currently has a 'no-burn days' program that prohibits open-air fires during certain weather conditions, 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 wind speed and high air stability – conditions correlated with high particulate exposure (see Blackman *et al.*, 2003 for a more detailed description of the methodology we use to select these days). Because enforcement of no-burn days is bound to be imperfect, we assume only half of the kilns scheduled to fire on no-burn days actually forego firing.

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. Thus, for policy purposes, we

Table 3. *Annualized costs of pollution control strategies for brick kilns (1999 $US unless otherwise noted)*

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

Notes: [a]Alfaro (personal communications, 1995 and 2000).
[b]Marquez (personal communication, 2000).
[c]Reynoso (personal communication, 2000).
[d]Johnson (personal communication, 2000).
[e]RFF (1995).

effectively treat all brickmakers in Ciudad Juárez as a single emissions source.[4]

Table 3 gives the annualized costs of each of the control strategies. For the NMSU kilns strategy, the sole cost is that of building modified kilns. 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 propane, conversion to natural gas requires investments in a burner and modifications that enable the kiln to withstand higher temperatures.[5] For the relocation strategy, we assume that capital costs are comprised of two elements: the financing needed to relocate brickmakers' homes ($4,500), and that needed to build a new kiln ($1,500). Our costs for the no-burn days strategy are based on the administrative costs of a similar program in

[4] The principal reason is that brick making – like most informal activities with low barriers to entry and slim profit margins – is intensely competitive. Therefore, as the propane project in the early 1990s demonstrated, policy scenarios in which only a portion of the city's brickmakers adopt a cost-increasing or cost-decreasing pollution control strategy are not sustainable, as the adopters are at a cost disadvantage or cost advantage (Blackman, 2000).

[5] Note that we ignore the effects of NMSU kilns and natural gas on variable production costs. The former strategy would reduce variable costs since it cuts fuel costs by approximately 50 per cent, while the latter strategy would raise variable costs since natural gas is more expensive than traditional fuel. We assume that given uniform application of these control strategies across all brickyards, prices in the highly competitive market for bricks would adjust to offset the changes in variable costs. A more rigorous evaluation of these market effects is beyond the scope of this study.

El Paso. This program entails labor costs only: 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 of $18,000 per person-year.

4. Formal industrial sources

4.1 Description

Ideally, in order to assess the relative importance of controlling emissions from informal sources, we would simply compare estimates of the net benefits of cutting brick kiln PM10 with estimates of the net benefits of controlling emissions from an array of formal sector sources in the city. Unfortunately, however, emissions and abatement cost data for formal sources in Ciudad Juárez either do not exist, or are extremely tightly held. Therefore, we have developed net benefits estimates for a small sample of (two) formal sources using US Environmental Protection Agency models. In doing so, we have taken care to minimize the likelihood that our methods bias these estimates downwards, and thereby inappropriately lead to a finding that net benefits of controlling brick kiln PM10 are relatively high by comparison. Specifically, we have purposely chosen to estimate emissions from formal sector sources that have much higher PM10 emissions than most formal facilities in the airshed.[6] In addition, given uncertainty about abatement investments at the two plants, we present a variety of net benefits estimates based on different assumptions about such investments.

Although questions have been raised about the reliability of the facility-level data in the SNIFF emissions inventory – indeed, this is the reason we use US Environmental Protection Agency models instead of these data to estimate PM10 emissions – an alternative ranking of local emissions sources is lacking and, therefore, we rely upon the SNIFF to identify the city's top seven industrial sources of PM10. In the summer of 2001, we interviewed managers and engineers of these plants (both in person and by telephone) in an attempt to obtain production data needed to estimate PM10 emission and abatement costs. Two of these seven facilities provided us with the requisite data: a US-owned gray iron foundry and a Belgian-owned chemical plant. The iron foundry produces table bases. It employs about 140 workers and is located in an industrial park in a densely populated central section of Ciudad Juárez (figure 1). The chemical plant mainly produces hydrofluoric acid. It employs about 150 workers and is located in the sparsely populated southern section of the city.

We used US Environmental Protection Agency emissions factors to estimate emissions from each of the particulate-intensive production processes in the iron foundry and chemical plant (US EPA, 1995).[7] Based on

[6] Although we know little about how abatement costs at our two sample plants compare to costs at other plants, we do know that abatement costs are relatively unimportant in estimating net benefits since benefits of controlling PM10 pollution are two to three orders of magnitude higher than the costs (tables 5 and 10).

[7] In the gray iron foundry, the principal sources of PM10 emissions are, in order of magnitude: pouring and cooling of molten iron, handling of sand used to make

Table 4. *Estimated annual PM10 emissions from formal and informal sources (tons)*

	Emissions	
Facility/process	*No controls*	*US controls*
Iron foundry		
Induction furnace	4.43	0.89
Pouring/cooling	11.28	0.56
Shakeout	12.26	0.61
Sand handling	13.78	0.69
Cleaning/finishing	0.37	0.02
Total	42.12	2.77
Chemical plant		
Fluorspar drying	748.70	7.49
Fluorspar handling	598.00	5.985
Fluorspar transfer	60.00	11.97
Total	1,407.60	25.45
Brick kilns		
Total	596.19	n/a

Sources: US EPA (1995) for industrial facilities; Parameters in table 2 for brick kilns.

engineering estimates and historical emissions data, these emissions factors are widely used by regulatory agencies around the world to estimate plant-level emissions. The data required to use them include: the type of output, the scale of production, and the type of production technology. We obtained these data from plant managers.

An important element of uncertainty in estimating emissions at the two formal plants concerns their actual levels of pollution abatement. In Mexico, as in many developing countries, pollution control regulations are fairly stringent. Non-compliance is widespread, however (Dasgupta *et al.*, 2000). Both the iron foundry and chemical plant claim to fully comply with pollution control regulations and – given that maquiladoras on the US–Mexico border are subject to considerable scrutiny – these claims may well be valid. Unfortunately, we are not able to verify these claims: reliable plant-level data on pollution abatement are tightly held, as are monitoring data collected by Mexican regulatory authorities. Given this uncertainty – and given that our goal is to estimate net benefits for *typical* formal industrial facilities which may not comply with pollution control regulations – we estimate emissions for the iron foundry and chemical plant assuming alternatively that they have installed: (i) all of the pollution control equipment standard in similar US plants, and (ii) no pollution control equipment whatsoever. Table 4 presents estimated annual PM10

molds, shaking sand from the molds, cleaning and finishing of cast iron, and operating induction furnaces. The bulk of the chemical plant's PM10 emissions come from the use of fluorspar, the principal material used in the manufacture of hydrofluoric acid. In particular, PM10 is emitted in drying, handling, and transferring fluorspar.

Table 5. *Annualized costs of installing PM-10 abatement equipment standard at US plants in Mexican formal industrial facilities (1999 $US)*

Facility/process	*Abatement device*	*Annual abatement cost*
Iron foundry		
Induction furnace	pulse jet bag house	13,395
Pouring/cooling	pulse jet bag house	34,109
Shakeout	shaker bag house	1,152
Sand handling	pulse jet bag house	1,164
Cleaning/finishing	shaker bag house	54
Total		49,874
Chemical plant		
Fluorspar drying	pulse jet bag house	10,955
Fluorspar handling	pulse jet bag house	8,750
Fluorspar transfer	pulse jet bag house	8,281
Total		27,986

Sources: US EPA (1998a, 1998b, and 1999b).

emissions for the two plants and for the brick kilns. Note that particulate emissions from the chemical plant are significantly higher than emissions from the iron foundry.

In addition to PM10 emissions, our air dispersion model requires detailed source-specific data on stack heights, emissions velocities, and plume temperatures. Managers of the two formal plants did not provide these parameters. Therefore, we estimated them using publicly available data on US gray iron foundries and hydrofluoric acid plants (US EPA, 2001).

We used US Environmental Protection Agency (EPA) spreadsheet models (US EPA, 1998a; US EPA, 1999b) along with control technique documentation (US EPA, 1998b) to estimate the costs of particulate control equipment likely to be found at the iron foundry and chemical plant (table 5). Based on vendor quotes, these spreadsheet models estimate the annual costs of installing and operating various types of abatement equipment to control emissions from specified production processes at specified types of plants. The data inputs required for these models (including engineering design values, operating statistics, electricity prices, and waste disposal costs) were obtained from a variety of sources including: interviews with plant managers, operating permits for US plants with identical outputs and similar scales, INEGI, the International Energy Agency, and interviews with waste disposal officials in Ciudad Juárez.

5. Benefits estimates

The benefits of pollution control for an emissions source, or collection of sources, are the difference between the damages associated with uncontrolled PM10 emissions and the damages associated with controlled PM10 emissions. As discussed in section 1, we use three models to estimate damages for each scenario: an air dispersion model, a health effects model, and a valuation model. This section discusses each of these models.

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Table 6. *Meteorological and topographical data*

Data	*Unit*	*Source*
Meteorology		
Temperature (hourly)	°K	NCDC (2000)
Wind speed	m/s	NCDC (2000)
Random flow vector	°	NCDC (2000)
Stability category	–	NCDC (2000)
Mixing height	m	US EPA (2000)
Topography	m	INEGI (1992)

5.1 Air dispersion model

The US EPA's Industrial Source Complex Short Term 3 (ISCST3) Gaussian plume air dispersion model uses data on emissions source characteristics, local meteorology, and topography to estimate average hourly, daily, and annual concentrations of primary PM10 due to a specific source inside a defined study area.[8] We use this model to estimate annual average concentrations of primary PM10 due to each of our study sources – brick kilns, the iron foundry, and the chemical plant – at 4,026 arbitrarily chosen 'receptor locations' in the study area.[9] Table 6 details the meteorological and topographical data used to parameterize the ISCST3 model. Regarding the former, the ISCST3 model uses one specific year's worth of hourly data on temperature, wind speed, wind direction, and mixing height.

5.2 Health effects model[10]

To estimate exposure to the PM10 emitted by the various sources, we use INEGI population data at the level of survey units called areas geoestadísticas básicas (AGEBs). Like census tracts in the United

[8] The ISCST model has been one of the US EPA's chief tools for investigating violations of ambient air quality standards (Riswadkar and Kumar, 1994; Patel and Kumar, 1998). Note that the ISCST3 model does not have the capability to handle long-range transport of PM10 or secondary PM10 formed when pollutants such as nitrogen oxides and sulfur dioxide undergo chemical reactions in the atmosphere. Neither phenomenon is likely to be important in our case study, however. Brick kilns in Ciudad Juárez are unlikely to contribute to long-range transport of PM10 because they do not have smoke stacks and are unlikely to generate significant secondary PM10 because they emit little sulfur oxide or nitrogen oxide. Our two formal sector sources are also unlikely to generate significant secondary PM10 or long-range transport. As table 4 demonstrates, over 99 per cent of PM10 emitted by both facilities results from handling of various materials at ground level, not combustion.

[9] The receptor locations were chosen by mapping a 365 meter rectangular grid on to the study area; that is, grid points are 365 meters apart. As discussed in the next subsection, this spatial detail is needed in order to estimate human exposure to this air pollution.

[10] Our health effects model draws on Cesar *et al.* (2002) and 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).

States, AGEBs vary in both size and population (figure 1).[11] We assign the inhabitants of each AGEB a distance-weighted average of PM10 concentrations predicted by the ISCST3 model at all receptor locations within 800 meters of the AGEB centroid.[12] 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 health endpoint due to a marginal change in exposure to a particular air pollutant. We model 11 different health endpoints, which are listed in table 7. We assume that these health effects are linear functions of PM10 exposure levels, a common approach in the literature (see, e.g., US EPA 1999a).[13]

Since, as discussed in section 5.3, 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 7 is that for mortality. We make the relatively conservative assumption – based on a number of US studies (Ostro, 1994) – that a 10 ug/m^3 change in daily PM10 results in a 1 per cent annual increase in the mortality rate.[14] A discussion of the remaining CR coefficients in table 7 can be found in chapter 8 of Bloyd *et al.* (1996) and in Cesar *et al.* (2002).

[11] Within Ciudad Juárez, AGEB size averages 564,572 m^2 with a standard deviation of 783,681 m^2. The smallest AGEB is 5,980 m^2 and the largest is 8,082,810 m^2. AGEB population averages 2,216 with a standard deviation of 1,387. The smallest AGEB has just three inhabitants and the largest 8,041.

[12] We do not assign the inhabitants of each AGEB the average of PM10 concentrations predicted by the ISCST3 model at all receptor locations within the AGEB, because this method would generate unrealistic spikes in exposure in several small AGEBs where there are a limited number of (arbitrarily located) receptor points and one of these receptors happens to be in very close proximity to an emissions source. Assigning the inhabitants of each AGEB a distance-weighted average of PM10 concentrations predicted by the ISCST3 model at all receptor locations within 800 meters of the AGEB centroid results in more conservative exposure estimates.

[13] 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).

[14] The CR mortality coefficient we use is virtually identical to a weighted average drawn from 40 studies from all over the world that was used by Cesar *et al.* (2002) to estimate mortality due to PM10 in Mexico City. Our CR mortality coefficient is relatively conservative. For example, in estimating the costs and benefits of the Clean Air Act, the US EPA relied on a large-scale study that followed a sample population over time and found that a 10 ug/m^3 change in PM10 results in a 3.6 per cent annual increase in the mortality rate (Pope *et al.*, 1995; US EPA, 1999a). Other US 'cohort' studies find even larger effects (e.g., Dockery *et al.*, 1993). A weighted average of Mexican studies used by Evans *et al.* (2000) to estimate mortality due to PM10 in Mexico City (1.4 per cent) is also larger than the mortality CR coefficient we use.

Table 7. *Health effects model inputs*

	Parameter	*Value or distribution*	*Units*	*Source*
	CR coefficients			
CR_{MORT}	Mortalities	$N(1m, 300u)$	% change mortality rate/(ug/m^3)	Krupnick (1996)
CR_{RHA}	Respiratory hospital admissions	$N(1.39m, 105u)$	% change RHA rate/(ug/m^3)	Cesar *et al.* (2002), pooled estimate
CR_{CHA}	Cardiocerebrovascular h. admis.	$N(600u, 93u)$	% change CHA rate/(ug/m^3)	Cesar *et al.* (2002), pooled estimate
CR_{ERV}	Emergency room visits	$N(311m, 383u)$	% change ERV rate/(ug/m^3)	Cesar *et al.* (2002), pooled estimate
CR_{ARSD}	Adult symptom days	$N(247m, 59m)$	days/yr./(ug/m^3)/adult[a]	Krupnick *et al.* (1990)
CR_{ARAD}	Adult restricted activity days	$N(57.5m, 27.5m)$	days/yr./(ug/m^3)/non-asthmatic adult	Ostro (1987)
CR_{CRAD}	Child restricted activity days	$N(57.5m, 27.5m)$	days/yr./(ug/m^3)/non-asthmatic child	Ostro (1987)[b]
CR_{AA}	Asthma attacks	$N(6.1m, 3.1m)$	attacks/(ug/m^3)/asthmatic person	Ostro *et al.* (1991)
CR_{CCB}	Child chronic bronchitis	$N(1.59m, 805u)$	cases/yr./(ug/m^3)/child[a]	Dockery *et al.* (1989)
CR_{CCC}	Child chronic cough	$N(1.84m, 924u)$	cases/yr./(ug/m^3)/child[a]	Dockery *et al.* (1989)
CR_{ACB}	Adult chronic bronchitis	$N(61.5u, 30.7u)$	cases/yr./(ug/m^3)/adult[a]	Abbey *et al.* (1993)
	Population data			
	Ciudad Juárez 1995	–	persons per survey unit	INEGI (1995)
	Other parameters			
	Baseline mort. rate Chih. 1997	5.506	deaths/1000 persons/year	INEGI (2000)
	Background RHA rate	4.11	RHA/1000 persons/year	Cesar *et al.* (2002), citing SSA (1996)
	Background CHA rate	4.03	CHA/1000 persons/year	Cesar *et al.* (2002), citing SSA (1996)
	Background ERV rate	31.68	ERV/1000 persons year	Cesar *et al.* (2002), citing SSA (1996)
	Fraction pop. asthmatic	0.05 (US rate)		Bloyd *et al.* (1996)

Notes: [a]Adults are defined as persons older than 17.
[b]Following Cesar *et al.* (2002), CR functions for CRAD and ARAD are assumed to be the same.
$m = 10^{-3}$.
$u = 10^{-6}$.
$N(\mu,\sigma)$ = normal distribution with μ= mean and σ= standard deviation.

Table 8. *Benefits valuation model inputs (1999 $US/case unless otherwise noted)*

	Parameter (value of...)	*Work-loss days*[a] *(days/case)*	*Cost of illness*[b] *($/case)*	*WTP*[c] *($/case or death)*
VSL	statistical life			[Prtb(1.9M,3.8M,7.5M) (0.33,0.34,0.33)]
Val_{RHA}	respiratory hospital admission	8	1,870	
Val_{CHA}	cardio. hospital admission	45	5,611	
Val_{ERV}	emergency room visit	5	91	
Val_{ARSD}	adult respiratory symptom day	1	10	
Val_{ARAD}	adult restricted activity day	1	10	
Val_{CRAD}	child restricted activity day	1	10	
Val_{AA}	asthma attack	1	337	
Val_{CCB}	child chronic bronchitis	7	190	
Val_{CCC}	child chronic cough	7	190	
Val_{ACB}	adult chronic bronchitis	7	218	

Notes: [a]Source: Cesar *et al*. (2002). Work loss days are valued at $17/day, the average manufacturing wage for Ciudad Juárez (INEGI, 1995).
[b]*Source*: Hernández-Ávila *et al*. (1995).
[c]*Source*: Rowe *et al*. (1995) adjusted.
Prtb(a)(p) = discrete probability distribution: a = vector of outcomes; p = vector of probabilities for each.
$M = 10^6$.

A challenge in estimating morbidity damages is identifying a set of endpoints that reflects the full range of 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 that 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 (see Blackman *et al.*, 2003 for details).

5.3 Valuation model

To estimate the monetary values of health damages avoided by reducing PM10 emissions from our sample sources, we use two different approaches (table 8). To value premature mortalities, we use willingness to pay (WTP) figures obtained from the economics literature. To value morbidity, we use

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the sum of: (i) estimates of the value of work loss days (WLD) based on average daily wages in Ciudad Juárez, and (ii) estimates of the cost of illness (COI) based on a study of health care costs in Mexico (Hernández-Ávila *et al.*, 1995).[15] Since, as noted above, the lion's share of total estimated benefits arise from premature mortalities avoided, by far the most important parameter in the valuation model is the value of a statistical life (VSL). We use a discrete distribution – \$1.9 million (33 per cent), \$3.8 million (34 per cent), and \$7.5 million (33 per cent) – adjusted from Rowe *et al.* (1995) to value this endpoint. 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 1999a, Appendix H-8).

Unfortunately, to our knowledge, direct estimates of Mexican WTP for reductions in mortality risk are not yet available. Therefore, we use a WTP parameter based on US studies. But given that average income adjusted for purchasing power parity is approximately four times as high in the United States as in Mexico, Mexican WTP may be lower than US WTP.[16] Cultural factors may also cause WTP in the two countries to differ. To account for international differences in WTP, we use three different values for Mexican WTP based on three different assumptions about the elasticity of WTP with respect to income (E).[17] 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 per cent lower in Mexico than in the US, then WTP is 3.3 per cent 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 our WTP distribution is \$3.80 million assuming $E = 0$, \$2.42 million assuming $E = 0.33$, and \$0.97 million assuming $E = 1$.

5.4 Uncertainty

We use Monte Carlo simulation to account for uncertainty associated with the parameterization of the air dispersion, health impacts, and benefits valuation models. That is, where data on probability distributions is available (see tables 2, 7, and 8), we treat model parameters as distributions

[15] We use the sum of WLD and COI instead of WTP for two reasons. First, Mexican WTP estimates for morbidity endpoints do not exist. Second, excluding WTP yields more conservative estimates of net benefits. Using [COI + WLD + WTP] typically overestimates the true social value of morbidity since WTP includes both COI and WLD when health care costs are borne by individuals. As discussed above, our overall strategy is to avoid any upward bias in net benefits estimates.

[16] 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).

[17] 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 *et al.* (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).

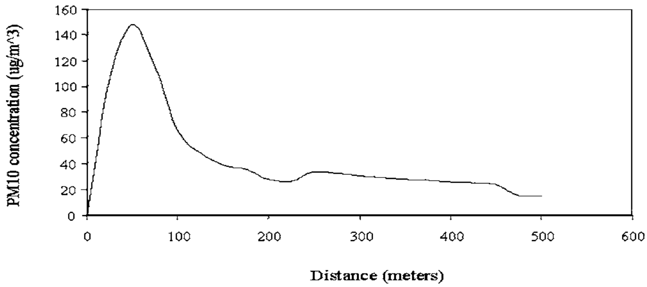

Figure 2. *Profile of brick kiln PM10 concentration downwind of Mexico 68*

Table 9. *Annual health effects of uncontrolled PM10 emissions from brick kilns (mean values and 95 per cent confidence intervals)*

	Number of cases		
Health endpoint	*Low*	*Mean*	*High*
Mortality	3	14	33
Respiratory hospital admissions	4	15	29
Cardio. hospital admissions	2	6	13
Emergency room visits	0	605	1,690
Adult respiratory symptom days	79,900	379,400	828,600
Work loss days	122,100	549,900	1,173,000
Adult restricted activity days	1,854	84,620	225,500
Child restricted activity days	1,240	56,650	150,900
Asthma attacks	8	782	2,075
Child chronic bronchitis	0	1,632	4,248
Child chronic cough	0	1,901	5,087
Adult chronic bronchitis	0	95	242

Source: RFF model.

and we use these distributions to generate 95 per cent confidence intervals for model outputs.

6. Results

6.1 Brick kilns

The air dispersion model results suggest 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 brick kilns primarily affect PM10 concentrations within 500 meters. This result stems from the fact that brick kilns have very low stack-heights and emissions velocities.

The health effects model suggests that brick kilns have significant impacts on mortality and morbidity (table 9). Most important, the model's mean

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Table 10. *Value of annual morbidity and mortality due to (i) uncontrolled PM10 emissions from brick kilns and (ii) controlled and uncontrolled PM10 emissions from formal industrial facilities (millions of 1999 $US; mean values and 95 per cent confidence intervals)*

	Value				
Source	*(E = 1.0) Mean*	*Low*	*(E = 0.33) Mean*	*High*	*(E = 0) Mean*
Brick kilns	29.25	6.50	53.11	160.10	75.79
Iron foundry					
US-level controls	0.18	0.07	0.32	0.76	0.45
No controls	2.76	1.09	5.00	11.94	7.13
Average[a]	1.47	0.58	2.66	6.35	3.79
Chemical plant					
US-level controls	0.26	0.08	0.47	1.24	0.68
No controls	13.15	4.25	23.91	63.26	34.13
Average[a]	6.71	2.17	12.19	32.25	17.40

Notes: [a]Average of 'no controls' and 'US-level controls' scenarios.
E = the elasticity of WTP (willingness to pay) with respect to income adjusted for purchasing power parity.
Source: RFF model.

prediction is that brick kiln emissions are responsible for 14 premature mortalities per year.

The first row of table 10 presents our estimates of the value of annual morbidity and mortality due to uncontrolled brick kiln emissions. (We discuss the remainder of table 10 in section 6.2). We report three sets of mean estimates based on different assumptions about the elasticity of WTP with respect to income. To illustrate the variation in our estimates, we also present the 95 per cent confidence interval for middle elasticity case. Even when this elasticity is assumed to be quite high ($E = 1.0$), the mean estimate is nearly $30 million. Assuming a middle elasticity value ($E = 0.33$), the mean estimate is $53 million. Assuming that Mexicans and Americans have the same WTP ($E = 0$), the mean estimate is $76 million. Note that reduced mortality is by far the largest component of benefits, accounting for over three-quarters of the total.

Table 11 presents estimates of the annual net benefits of each of the four brick kiln pollution control strategies allowing for three different values of E. For each value of 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 benefits of relocation are about half that of natural gas, and the net benefits of no-burn days are 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. Relocation is ranked third because the total benefits of this strategy are about half those for natural gas, while the costs are the highest of any of the four strategies. The no-burn days

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Table 11. *Annual net benefits of pollution control strategies for brick kilns and formal industrial facilities (benefits less costs in millions of 1999 $US; mean values and 95 per cent confidence intervals)*

		Brick kilns				*Iron foundry*		*Chemical plant*	
E	*Scenario*	*Natural gas*	*NMSU kilns*	*Relocation*	*No-burn days*	*Baseline = no controls*[a]	*Baseline = 50% US controls*[b]	*Baseline = no controls*[a]	*Baseline = 50% US controls*[b]
0	High	232.22	231.01	132.05	4.59	16.23	8.12	90.27	45.13
	Mean	**75.46**	**75.02**	**43.10**	**1.49**	**6.63**	**3.31**	**33.42**	**16.71**
	Low	9.12	9.08	5.20	0.17	1.42	0.71	5.94	2.97
0.33	High	159.69	158.88	90.77	3.09	11.13	5.56	61.99	31.00
	Mean	**52.81**	**52.52**	**30.11**	**1.04**	**4.63**	**2.31**	**23.41**	**11.70**
	Low	6.24	6.24	3.50	0.12	0.97	0.49	4.14	2.07
1	High	83.53	83.14	47.54	1.64	5.75	2.88	32.25	16.13
	Mean	**28.97**	**28.85**	**16.44**	**0.56**	**2.53**	**1.26**	**12.86**	**6.43**
	Low	3.21	3.24	1.70	0.06	0.51	0.25	2.24	1.12

Notes: [a]Net benefit of installing and operating PM10 control equipment standard in a similar US plant, assuming baseline has no controls.
[b]Net benefit of installing and operating PM10 control equipment standard in a similar US plant, assuming baseline has equipment sufficient to achieve 50 per cent of abatement that would result from US-level controls.
E = the elasticity of willingness to pay with respect to income adjusted for purchasing power parity.
Source: RFF model.

strategy is ranked last because it generates the lowest total benefits – less than one-hundredth of those associated with natural gas.

The mean annual net benefits of all four strategies are quite large for each of the three assumptions about the value of E. Assuming $E = 0.33$, the mean annual net benefits range from $53 million for natural gas to $1 million for no-burn days. Even the lowest estimates – those at the low end of the 95 per cent confidence interval given $E = 1$ – are positive for all of the control strategies.

6.2 Informal vs. formal sources

Table 10 presents estimates of the annual value of health damages due to PM10 emissions from the formal plants given two alternative emissions control scenarios: (i) absolutely no emissions controls, (ii) US-level controls. In addition, the table presents the average for these two scenarios.

We begin with two points that are fairly obvious, but important nonetheless. First, the health damages associated with the chemical plant are much higher than those associated with the iron foundry. The main reason is simply that the chemical plant emits far more PM10 (table 4). Second, the health damages from uncontrolled emissions are much higher than for controlled emissions – 16 times higher for the iron foundry and 50 times higher for the chemical plant. Thus, the magnitude of the health damages the formal plants generate depends critically on their investments in pollution control.

A comparison of the damages from the formal plants and from the brick kilns reveals that the former are significantly smaller than the latter, regardless of the assumption made about the level of pollution abatement at the two formal plants (table 10). If we assume that the formal plants have installed abatement devices standard in the US, and if $E = 0.33$, then the mean value of total damages generated by the chemical plant ($0.5 million) and the iron foundry ($0.3 million) are each less than 1 per cent of damages due to brick kilns ($53 million). If we assume, alternatively, that emissions from the formal plants are completely uncontrolled, then these percentages are 45 per cent and 9 per cent.

Given this disparity in damages – and the fact that the costs of PM10 emissions controls are relatively small in comparison to the benefits – it is not surprising that the net benefits of most of the emissions control strategies for brick kilns are significantly higher than the net benefits of installing US-levels of emissions controls at the formal plants. Table 11 presents estimates of the net benefits of installing emissions controls standard in the US at the two formal plants given two alternative assumptions about the 'baseline' level of pollution control (i.e., the existing level of abatement prior to installing any new equipment): (i) absolutely no controls, and (ii) 'average' controls, i.e. the investment needed to eliminate 50 per cent of the emissions eliminated by US-level controls. We assume that the annualized costs in the second scenario are half of those in the first. To simplify the comparison, we restrict attention to the mean values of total benefits and assume $E = 0.33$.

Of the various pollution control scenarios for formal plants included in table 11, the greatest net benefits – $23 million – result from installing

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US-level controls at the chemical plant assuming a baseline of absolutely no emissions controls. These net benefits are just 44 per cent of the net benefits of converting traditional brick kilns to natural gas or replacing them with NMSU kilns. Assuming a more realistic average baseline level of control, the net benefits of installing US-level controls in the chemical plant are $12 million, just 22 per cent of the net benefits of converting traditional brick kilns to natural gas or replacing them with NMSU kilns.

The net benefits of installing US-level controls at the iron foundry are an order of magnitude lower than the net benefits of installing the same level of controls at the chemical plant. Assuming a baseline of absolutely no emissions control, the net benefits of installing US-level controls at the iron foundry are $5 million, just 9 per cent of the net benefits of converting traditional brick kilns to natural gas or replacing them with NMSU kilns. Assuming a more realistic average baseline, the net benefits of installing US-level controls at the iron foundry are $2 million, just 4 per cent of the net benefits of converting traditional brick kilns to natural gas or replacing them with NMSU kilns.

7. Conclusion

We have used a specially parameterized air dispersion model in combination with concentration response coefficients and benefits transfer methods to estimate the net benefits of controlling PM10 from a collection of informal brick kilns and two formal industrial facilities in Ciudad Juárez. Our two principal findings are as follows. First, given a wide range of modeling assumptions, the benefits of three of four control strategies for brick kilns – NMSU kilns, natural gas, and relocation – are considerably higher than the costs. Second, the net benefits of these three strategies exceed the net benefits of controlling emissions from the two formal facilities by a considerable margin. However, the size of this margin depends critically on the baseline level of pollution abatement in formal factories. In addition, we found that health damages from brick kiln emissions are spatially concentrated in the poor residential neighborhoods surrounding the largest brickyards. What are the policy implications of these findings?

The fact that our estimates of the net benefits of controlling brick kiln emissions are positive and quite large strongly suggests that, in general, policy makers should at very least include clusters of informal polluters among those industrial emissions sources they consider to be potential targets for pollution control initiatives. However, the fact that the net benefits of controlling brick kiln emissions exceed those for two leading formal industrial sources of air pollution does not necessarily suggest that, in general, policy makers should shift scarce pollution control resources away from industrial facilities and towards clusters of informal polluters.

In deciding how to allocate pollution control resources, policy makers must consider not only health benefits and abatement costs – the two components of our net benefits estimates – but also implementation costs – the costs environmental management authorities incur in monitoring environmental performance, prosecuting non-compliance and, in certain cases, subsidizing abatement costs by, for example, educating managers of informal firms about the requirements and options for abatement and providing equipment and financial assistance. As noted in section 1, developing prospective estimates of implementation costs is difficult and is outside the scope of this study. However, there is good reason to expect these costs to be higher for clusters of informal polluters than for formal industrial facilities. Often informal firms are difficult to identify. Even if they can be identified, monitoring numerous tiny firms may simply be impractical for chronically under-manned and under-funded municipal regulatory agencies. In addition, requiring cash-strapped informal firms to bear the costs of pollution control may be unrealistic. Finally, and perhaps most important, it may be difficult to generate the political will to impose costs on firms which, as an important employer of the urban poor, are seen to perform a vital distributional function.

That said, there is little reason to believe that implementation costs for informal sources are, in general, prohibitive. Recent case studies – including companion research on brick kilns in northern Mexico – have demonstrated that a number of strategies can be used to lower implementation costs in the informal sector (Blackman, 2000; World Bank, 1998; Kennedy, 1999; and Wheeler *et al.*, 1999). A common element of many of these strategies is recruiting local stakeholders – including communities affected by informal sector pollution, organizations representing the informal firms, competitors, and upstream and downstream business contacts – to pressure polluters to cut emissions, to monitor their environmental performance and, in some cases, to subsidize investments in pollution abatement – in effect to undertake many of the functions traditionally performed by public-sector regulators. For example, in past efforts to convert brick kilns in Ciudad Juárez to clean-burning propane, neighborhood organizations and trade unions pressured brickmakers to switch and monitored their progress. Also, local propane companies subsidized investments in propane burners. The key to motivating stakeholders to undertake these tasks is often improving public awareness of, and appreciation for, informal sector pollution problems. Studies like this one may be able to contribute to this effort by demonstrating that informal polluters can generate very severe health damages, and that these damages may be disproportionately visited on those living nearby.

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