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Abstract

In the past two decades, rapid population and economic growth on the U.S.–Mexico border has spurred a dramatic increase in electricity demand. In response, American energy multinationals have built power plants just south of the border that export most of their electricity to the United States. This development has stirred considerable controversy because these plants effectively skirt U.S. environmental air pollution regulations in a severely degraded international airshed. Yet to our knowledge, this concern has not been subjected to rigorous scrutiny. This paper uses a suite of air dispersion, health impacts, and valuation models to assess the human health damages in the United States and Mexico caused by air emissions from two power-exporting plants in Mexicali, Baja California. We find that these emissions have limited but nontrivial health impacts, mostly by exacerbating particulate pollution in the United States, and we value these damages at more than half a million dollars per year. These findings demonstrate that power-exporting plants can have cross-border health effects and bolster the case for systematically evaluating their environmental impacts.

Key Words: electricity, air pollution, Mexico

JEL Classification Numbers: Q48, Q51, Q53

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Health Impacts of Power-Exporting Plants in Northern Mexico

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1. Introduction

Rapid population and economic growth on the U.S.–Mexico border during the past two decades has spurred a dramatic increase in the region's demand for electricity, causing intermittent excess demand (Romero 2007; Sweedler et al. 2002). In response, American energy multinationals have built power plants on the Mexican side of the border that sell most of their electricity to the United States. In 2003, two such power-exporting plants began operation three miles south of the border near Mexicali, Baja California, and Imperial County, California. Owned by Intergen and Sempra Energy and fueled with natural gas imported from the United States, the two plants sell three-quarters of their power to the U.S. grid. Various multinational companies are reportedly considering building similar power-exporting plants (Powers 2010; Barron 2005).

The Intergen and Sempra plants, and the prospect of more like them, have stirred considerable controversy (Carruthers 2007; Spagat 2003; Weiner 2002). Critics contend that they will degrade the binational area's already-poor air quality. By locating just south of the border, the plants skirt U.S. federal and state air pollution regulations, including those requiring all new facilities in degraded airsheds to offset their emissions (i.e., to pay for more-than-equivalent emissions reductions from other sources). Indeed, some argue that a major reason the new power plants were built in Mexico was to avoid U.S. environmental restrictions. In response, representatives and supporters of the power companies have pointed out that imposing new

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environmental restrictions on Mexican power-exporting plants would slow investment in new generating capacity and raise the cost of energy in the United States.

These issues have spurred legal and legislative activity as well as popular debate. In 2002, a coalition of Mexican and U.S. environmentalists brought an (ultimately) unsuccessful suit in California state court seeking an injunction barring the plants from exporting to the United States (Barron 2005; Tedford 2003). In 2005, the U.S. Congress commissioned an independent report on the matter by the U.S. Government Accountability Office (GAO 2005). And in 2010, California, alone among the four U.S. states on the Mexico border, passed legislation requiring new electricity-generating units in Mexico selling power to the state to comply with its air pollution control regulations (California Senate Bill 2037).

Despite all that activity, the central question of whether the Intergen and Sempra plants actually have significant human health impacts has not been subjected to rigorous scientific scrutiny. Indeed, the U.S. Government Accountability Office report concludes that "although emissions generated from the Sempra and Intergen plants may contribute to various adverse health impacts … the extent of such impacts is unknown" (GAO 2005).

To help fill that gap, this paper uses a suite of air dispersion, health impacts, and valuation models to assess the benefits of reducing—and/or offsetting—polluting emissions from the Intergen and Sempra power plants in Mexicali and, based on this assessment, to distill recommendations for regional energy and environmental policy. Specifically, we address the following four questions: What effect do emissions from the Intergen and Sempra plants have on ambient concentrations of air pollutants in the Mexicali–Imperial Valley airshed as well as more distant downwind areas? What effect do these changes in ambient pollutant concentrations have on human morbidity and mortality? What is the economic value of this morbidity and mortality? And what are the implications for regional energy and environmental policy?

Although our analysis focuses on a particular environmental issue in North America, it is relevant to air pollution problems on borders between industrialized countries with relatively robust environmental regulatory regimes and developing or transitioning countries with weaker ones. In such situations, each country may suffer damages from air pollution generated by its neighbor. In addition, dirty industries in the industrialized country could, in theory, migrate to neighboring "pollution havens" to cut environmental regulatory costs, a phenomenon sometimes referred to as a race to the bottom (Fullerton 2006; Spar and Yoffie 2000). Concerns about this phenomenon have been voiced in Europe where industrialized countries share airsheds with nearby transitioning countries (Lynch 2000; Kaldellis et al. 2007). The electricity sector is of particular concern—in 2010, the European members of the Organisation for Economic Co-operation and Development (OECD) imported 10 percent of their total consumption of 3.5 TWh (versus 1 percent for the United States, which consumed a total of 4.2 TWh) (U.S. EIA 2010).

The remainder of the paper is organized as follows. Section 2 provides information on the regulatory context for our own case study and for Europe. Section 3 discusses the history and technical characteristics of the Intergen and Sempra plants. Section 4 provides a brief overview of our modeling strategy. Section 5 describes our emissions data. Sections 6, 7, and 8 discuss the three principal components of our analysis: air quality modeling, health impacts analysis, and valuation. Finally, Section 9 summarizes our results and considers policy implications.

2. Regulatory Context

This section provides background on environmental regulatory context for transborder pollution control in our study countries and, for comparison's sake, in Europe.

2.1. The U.S.–Mexico Border

The federal Clean Air Act (CAA), passed in 1970 and amended in 1977 and 1990, provides the foundation for the United States' decentralized system of air pollution regulation (Erickson et al. 2004; KEMA et al. 2007). The CAA establishes broad guidelines for environmental management and sets National Ambient Air Quality Standards (NAAQS) for six criteria air pollutants—ozone (O3), nitrogen dioxide (NO2), carbon monoxide (CO), particulate matter (PM), sulfur dioxide (SO2), and lead (Pb)—as well as guidelines for hazardous air pollutants and two O3 precursors, volatile organic compounds (VOCs) and nitrogen dioxides (NOx). It assigns to states responsibility for developing and enforcing specific regulations to meet NAAQS. They have the option of establishing ambient standards that are more, but not less, stringent than those in the CAA. Of the four states on the U.S.–Mexico border, California and New Mexico have more stringent ambient standards, while Texas and New Mexico rely on federal standards. As mandated under the CAA, two of the principal tools that states use to achieve NAAQS are permits and offsets. Granted to individual facilities, permits are licenses to emit pollution that typically include sector- or source-specific emissions and/or technology standards based on the availability and cost of abatement and pollution prevention technologies. In nonattainment areas—those that fail to achieve compliance with NAAQS—sources are generally required to offset their emissions by more than 100 percent and to meet stricter emissions and technology standards.

Mexico has a more centralized system of air pollution regulation (Erickson et al. 2004; KEMA et al. 2007). Its framework federal environmental law is the 1988 General Law on Ecological Equilibrium and Environmental Protection (*Ley General de Equilibrio Ecológico y la Protección al Ambiente*, LGEEPA), which is complemented by numerous more specific Official Mexican Norms (*Normas Oficiales Mexicanas*, NOMs). The Environmental Ministry (*Secretaría de Medio Ambiente y Recursos Naturales*, SEMARNAT) is responsible for establishing and enforcing most environmental regulations, including air pollution regulations in the U.S.–Mexico border zone.[1] LGEEPA sets ambient air quality standards that are quite similar to U.S. NAAQS. To meet these standards, federal NOMs set sector-specific emissions standards for stationary sources of air pollution. In the case of the electricity sector, emissions standards depend on the size and type of the generating facility (KEMA et al. 2007; Johnson and Alvarez 2003).

In general, air pollution control regulation is significantly more stringent on the U.S. side of the border, for at least three reasons. First, some ambient air quality standards are more stringent. Although Mexican federal ambient standards are generally comparable to U.S. NAAQS, as noted above, California's and New Mexico's standards are more exacting than NAAQS (Erickson et al. 2004). Second, regulations aimed at meeting these ambient air quality standards are more stringent. Several U.S. border counties and cities are NAAQS nonattainment areas, where new and expanding sources are required to offset their emissions. For example, Imperial County, just north of Mexicali, is a nonattainment area for O3 and PM. By contrast, offsets are not required in Baja California (Erickson et al. 2004; KEMA et al. 2007).[2] This implies that if the Intergen and Sempra plants had been built in U.S. territory north of Mexicali, they would have been required to offset their PM, NOx, and VOC emissions. Third,

[1] Within the Environment Ministry, responsibility for standard setting falls to the National Ecology Institute (*Instituto Nacional de Ecología*, INE), while responsibility for monitoring and enforcement is assigned to the Environmental Attorney General's Office (*Procuraduría Federal de Protección al Ambiente*, PROFEPA). Over the past several decades, in accordance with new constitutional guidance, some authority for environmental regulation has been devolved to states. However, in practice, SEMARNAT retains most authority (Blackman and Sisto 2006; Lybecker and Mumme 2003).

[2] It is less clear that California's technology and emissions standards for air pollution are more stringent than Mexico's. These standards are difficult to define because they are established on a case-by-case basis depending on best available control technology and on facility- and site-specific factors such as the economic impacts of the standard on the polluter. That said, GAO (2005) concluded that California's technology and emissions standards for new gas-fired power plants are more stringent than Mexico's standards. Specifically, the study determined that if the Intergen plant were located in California, it likely would have been required to install additional CO control equipment and to lower NOx from one of its turbines (the EBC unit) from 3.5 part per million (ppm) to 2.5 ppm. By contrast, GAO (2005) determined that as built, the Sempra plant would have met California standards.

environmental regulatory enforcement is generally stronger in the United States than in Mexico, where the Environmental Ministry's enforcement branch (*Procuraduría Federal de Protección al Ambiente*, PROFEPA) is widely acknowledged to be understaffed and underfunded (Gilbreath 2003; OECD 2003).

In principle, international cooperation on air quality management could mitigate problems arising from the cross-border disparity in air pollution regulation. Unfortunately, however, the legal and institutional basis for such cooperation is limited. With the exception of a new California law requiring new power-exporting plants to meet state air pollution regulations, U.S. federal and state air pollution control laws and regulations apply only to emissions sources located in the United States. The 1983 La Paz Agreement on binational environmental and natural resource issues creates only a vague, nonbinding framework for U.S.–Mexican cooperation on cross-border air pollution problems. And the environmental side agreements to the 1993 North American Free Trade Agreement require each country to enforce only its own pollution control laws. Given this legal and institutional context, it is perhaps not surprising that actual efforts to control U.S.–Mexico transborder air pollution have been ad hoc, fragmented, and weak (GAO 2005; Sánchez-Rodríguez 2002).

The regulatory framework for cooperation between the United States and Canada on transborder air pollution policy is better developed. In 1991, spurred by increasing concern about acid rain, the two countries signed a framework for cooperation on transfrontier pollution that included specific commitments to reduce sulfur dioxide. They have since negotiated an annex focusing on ground-level O3 and implemented binational pollution control projects focusing on specific airsheds, namely the Great Lakes and Puget Sound regions (Van Nijnatten 2003; Sánchez-Rodríguez et al. 1998).

2.2. Europe

On paper, disparities in the stringency of air pollution regulation between industrialized and transitioning countries in the European Union (EU) are minimal. To address concerns about competitive advantages created by differences in environmental management among member countries, the EU has a longstanding policy of "harmonizing" regulations across member sates. This policy assumed new importance with the beginning of the process for the accession of East and Central European candidate member states (Czech Republic, Estonia, Hungary, Latvia, Lithuania, Poland, Slovakia, and Slovenia) to the EU in the early 1990s. To ensure harmonization, an EU directive required new members to adopt an entire body of EU environmental legislation (Lynch 2000; Milieu Ltd. 2004). Notwithstanding progress on harmonization, however, observers have raised concerns about persistent differences in monitoring and enforcement in Central and Eastern European candidate member states (Lynch 2000; Jacoby 1999).

In addition to harmonizing regulations, EU countries also have established a legal and institutional framework for managing transborder pollution. In 1979, the countries signed the Convention on Long Range Transboundary Air Pollution, which established a framework for subsequent specific agreements, including the 1985 Sulfur Protocol committing all signatories to reduce sulfur dioxide emissions by 30 percent from a 1980 baseline over eight years; the 1988 Nitrogen Protocol, which committed signatories to maintain nitrogen oxides below 1987 levels until 1994; and a 1999 second Sulfur Protocol, which committed all signatories—including eight new East and Central European candidate members—to binding country-specific emissions reductions (Kaldellis et al. 2007; ApSimon and Warren 1996).

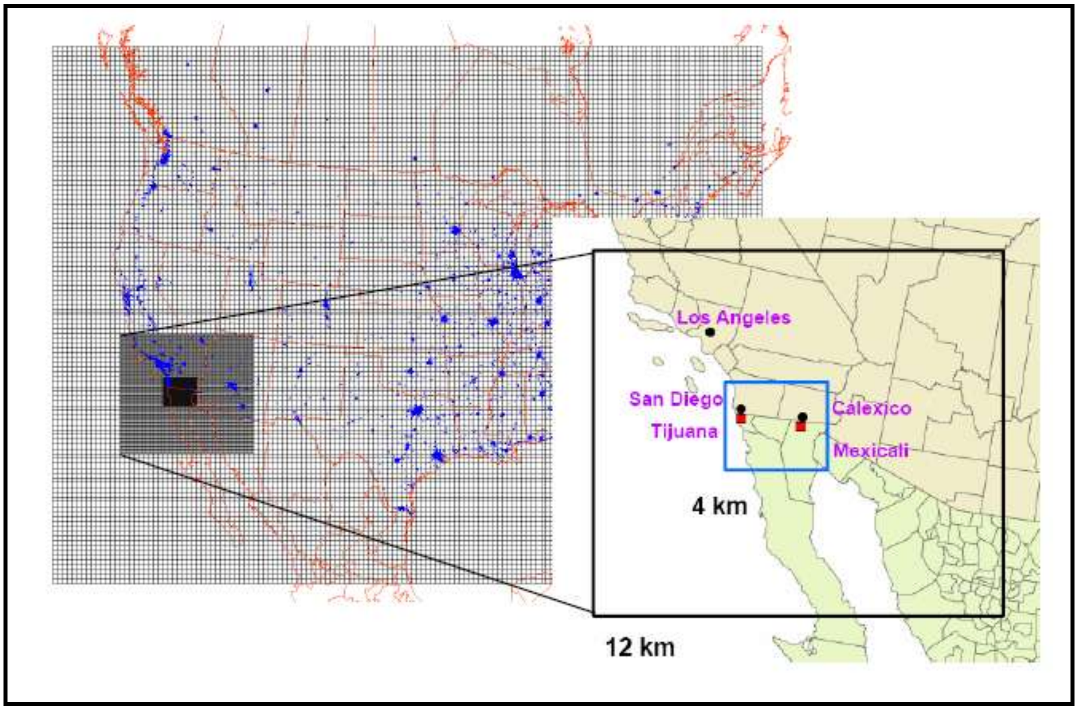
3. Power Plants

The Intergen and Sempra power plants are located in Mexicali, Baja California, approximately three miles south of the U.S.–Mexico border. Mexicali's U.S. sister city is Calexico, Imperial County, California (Figure 1). Both plants are combined-cycle natural gas–fired facilities, and both use fuel imported from the United States through a pipeline constructed by Sempra. Construction of the Intergen and Sempra plants, along with associated transmission lines, began in 2001, and both plants began operating in July 2003.

By all accounts, excess demand for electricity in California in 2000 and 2001 spurred Intergen's and Sempra's investment in the plants. Most analysts also agree that Intergen's and Sempra's decisions to build in Mexico were mainly driven by its shorter wait-times for regulatory permits (six months versus two years in California) and lower capital and labor costs. That said, most observers also agree that less stringent environmental regulation in Mexico may have played a role (Barron 2005; Tedford 2003).

The only international environmental regulatory hurdle that Intergen and Sempra plants in Mexicali faced was a U.S. regulation requiring foreign power plants using international transmission lines to the United States to obtain a presidential permit, which in turn is conditional on a U.S. Department of Energy (DOE) environmental impact assessment (EIA). Based on a 2001 EIA (which was refined in 2004), both plants were granted presidential permits (DOE 2004). However, as noted above, the EIA lacked a rigorous assessment of the health impacts of the power plant emissions, and partly as a result, it was contested in federal court by local environmental advocacy groups (Barron 2005; GAO 2005).

Figure 1. Location of Integen and Sempra Power Plants in Mexicali, Baja California and Air Quality Modeling Domain (36-km, 12-km, and 4-km grids)



The Intergen plant is known as the La Rosita Power Complex (LRPC). It houses two units. The first is owned by Energía Azteca X, S. de R.L. de C.V. (EAX), an Intergen subsidiary. It comprises three Seimens-Westinghouse Model W501F 160-MW combustion turbines and one Alstrom 270-MW steam turbine. Collectively, the unit has 750 MW of capacity. The second unit is owned by Energía de Baja California S. de R.L. de C.V. (EBX), a Mexican company. It comprises one Seimens-Westinghouse Model W501F 160-MW combustion turbine and one Alstrom 150-MW steam turbine. Collectively, this unit has a capacity of 310 MW.

Two-thirds of the power generated by EAX—all of the power from two of its three combustion turbines and two-thirds of the power from its steam turbine (2X160 MW+180MW = 500 MW)—is sold to Mexico, and the rest (160 MW + 90 MW = 250 MW) is exported to the United States. All of the EBC 310-MW capacity is exported. Hence, overall, the Intergen plant has a capacity of 1060 MW, of which 560 MW is devoted to exports and 500 MW to domestic production.

The Intergen plant does not use oxidizing catalysts or any other abatement method to reduce CO emissions. When the plant began operating in 2003, only one of its four combustion turbines—that in the EBC unit—employed selective catalytic reduction (SCR) to reduce emissions of NOx.[3] However, the three combustion turbines in the EAX unit were subsequently retrofitted with SCR (turbine 1 in March 2005, turbine 2 in April 2005, and turbine 3 in March 2004). Hence, since April 2005, all four combustion turbines in the Intergen plant have employed SCR.

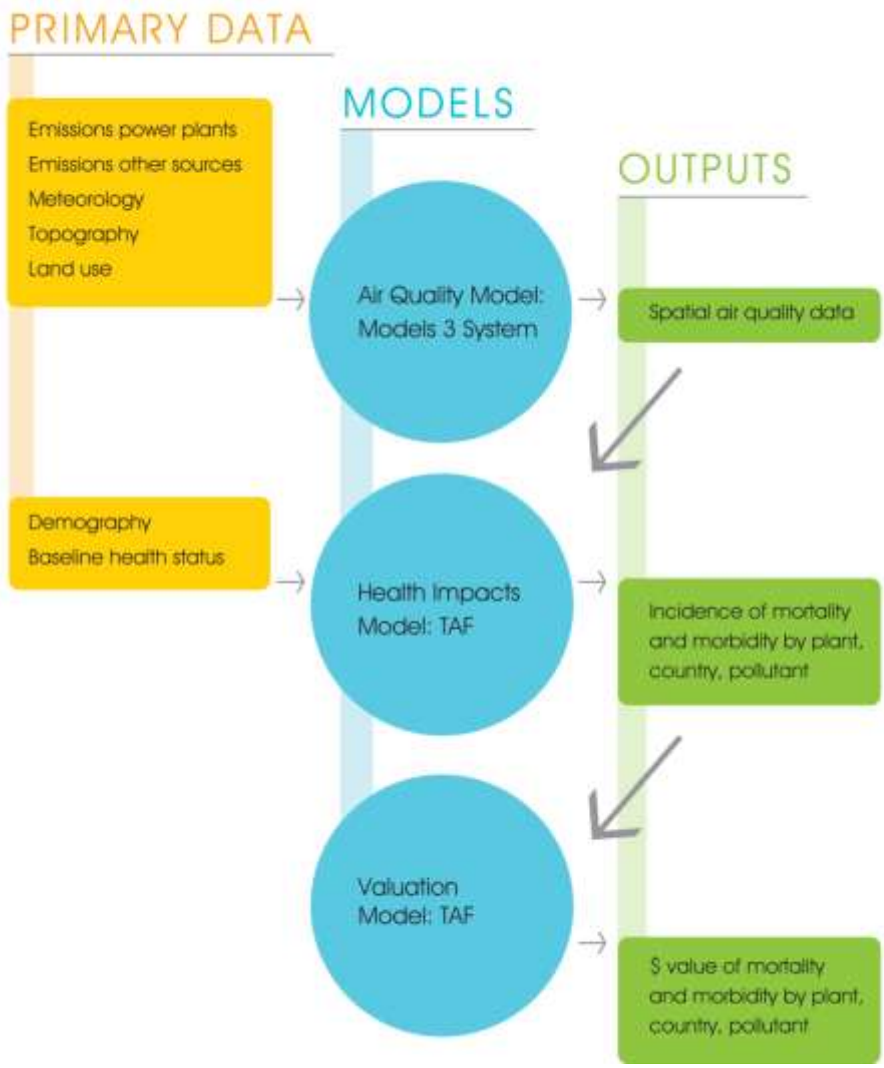
The Sempra plant is owned by Termoeléctrica de Mexicali (TDM), a Sempra subsidiary. It consists of two General Electric Model 7FA 170-MW combustion turbines and one Alstrom 310-MW steam turbine. The plant thus has 650 MW of generating capacity. One hundred percent of the plant's power output is exported to the United States. Both combustion turbines in Sempra plant were built with SCR and oxidizing catalyst units to reduce NOx and CO emissions.

4. Modeling Overview

Our analysis of the effects of the Intergen and Sempra plants' pollution on human health has three broad components, which are described in detail in Sections 6, 7, and 8 (Figure 2). The first is air quality modeling, for which we use the Models-3 system, a three-dimensional chemical-transport air quality modeling system. Inputs for this component include data on polluting emissions from the Intergen and Sempra plants, polluting emissions from other sources in our modeling domain, and meteorology, topography, and land use in this domain. The output from our air quality model is hourly spatial data on air quality for two multiday modeling episodes.

[3] SCR units reduce NOx by exposing combustion emissions to a spray of ammonia in the presence of a catalyst, typically platinum. The NOx reacts with the ammonia to produce nitrogen and water vapor.

Figure 2. Overview of Data and Models



The second component of our analysis is health impacts modeling, for which we use the Tracking and Analysis Framework (TAF), an integrated tool for benefit-cost analysis. TAF estimates health impacts for two pollutants: O3 and particulate matter smaller than 2.5 microns (PM2.5). In addition to the output from our air quality model (spatial data on air quality), inputs for the TAF health impacts model include spatial data on demography in our modeling domain and baseline data on health status. The outputs from this model are estimates of the number of cases of mortality and morbidity attributable to Intergen and Sempra pollution, broken down by plant (Intergen and Sempra), country (Mexico and the United States), and pollutant (O3 and PM).

The final component of our analysis is valuation modeling, for which we again use TAF. This component assigns monetary values to our estimates of the incidence of various health effects.

5. Emissions

Our estimates of most polluting emissions from the Intergen and Sempra plants—specifically, NO2, CO, particulates matter smaller than 10 microns (PM10), ammonia (NH3), and VOCs—are based on 2003 and 2004 third-party tests reported in GAO (2005), which characterizes these estimates as the most accurate emissions data available. For some pollutants (PM10 and VOCs), GAO (2005) reports hourly emissions, and for others (NOx, CO, and NH3), it reports stack gas concentrations. We convert the hourly emissions to annual emissions assuming the plants operate at capacity 64 percent of all available hours, the average capacity factor for recently built U.S. combined-cycle natural gas plants (Paul et al. 2009). We convert the concentration data to annual emissions using measurements of actual fuel consumption from DOE (2004, Table G1) and assuming again the plants operate at capacity 64 percent of all available hours (see Appendix 1). Finally, we estimate emissions of particulate matter smaller than 2.5 microns (PM2.5) from PM10 emissions (following U.S. EPA 1997) and SO2 emissions from DOE (2004, Table G1). Table 1 reports our results.

Table 1. Intergen and Sempra Plant Mass Emissions Rates (short tons/year*)

Pollutant	Intergen[a]	Sempra
NOx[b]	327.08	155.95
CO[b]	63.55	0.00
PM10[c]	84.21	89.01
NH3[b]	63.53	11.24
VOCs[c]	9.54	0.00
PM2.5[d]	84.21	89.01
SO2[e]	12.47	6.99

NOx = nitrogen oxides; CO = carbon monoxide; PM10 = particulate matter smaller than 10 microns; NH3 = ammonia; VOCs = volatile organic compounds; PM2.5 = particulate matter smaller than 2.5 microns; SO2 = sulfur dioxide.
*1 short ton = 0.907 metric tons.
[a]Assumes all Intergen plant turbines have selective catalytic reduction (SCR) units. SCR was installed on the Energía de Baja California (EBC) unit when it was built but was added to the Energía Azteca X (EAX) after construction. It was added to EAX Turbine 3 prior to the third-party tests reported in GAO (2005), and to EAX turbine 1 and 2 after these tests. SCR affects emissions of NOx and NH3. Therefore, we assume that NOx and NH3 emissions from EAX turbines 1 and 2 are the same as emissions from EAX turbine 3.
[b]Estimated from concentration in ppm assuming 65% capacity factor; see Appendix 1.
[c]Estimated from emissions in pounds per hour, which are converted to short tons per year assuming a 65% capacity utilization factor.
[d]Estimated from PM10 following (U.S. EPA 1997).
[e]Estimated from DOE (2004, Table G1).

Sources: GAO (2005); DOE (2004); own calculations.

6. Air Quality Modeling

To gauge the effect of emissions from the Intergen and Sempra plants on ambient air pollution concentrations in and around our study area, we applied an extended version of the Models-3 system, a three-dimensional chemical-transport air quality modeling system, to the Mexicali–Imperial Valley (Byun et al. 2005; U.S. EPA 1999). This application is described in detail in Mendoza-Dominguez et al. (2007), and its use in generating the results presented in this paper is described in Chandru (2008). Drawn from Chandru (2008), this section offers a brief overview of the principal steps involved in applying the Models-3 system and summarizes the main results.

6.1. Modeling Domain

We applied the Models-3 system using nested grids covering the border region (Figure 1). Vertically, we used 15 layers—that is, 15 slices of atmosphere of variable thickness. The lowest layer, the air people typically breathe, is 18 meters thick. The top of the modeling domain is 15 km above ground. Horizontally, at the most coarse resolution, we used 36-km grids. We used 12-km and 4-km grids for the Mexicali–Imperial Valley area. The coarse grid system allows relatively rapid simulation to set appropriate boundary conditions for the finer grid.

6.2. Application of Models-3 System

6.2.1. Episode Selection

The Models-3 system is quite computationally intensive and therefore simulates changes in air quality for multiday episodes within a defined year or years, rather than for an entire year. Following Boylan et al. (2005), Kuebler et al. (2002), and others, we used classification and regression tree (CART) analysis to select the days for these episodes. The objective was to select multiday periods that represent a variety of meteorological conditions that generate high air pollution levels. Also known as binary recursive partitioning, CART is a nonparametric statistical technique used for data classification and predictive modeling (Brieman et al. 1984). It generates a decision tree that defines the relationship between a categorical dependent variable and a set of independent variables. In our case, days of the years are observations, the categorical dependent variable is a series of ranges of average daily pollutant concentrations, and the independent variables are average daily meteorological conditions, such as temperature and wind speed. (As discussed in Section 7.1.1, a decision tree generated by CART also is used in our health impacts modeling to determine how representative each episode-day is in terms of meteorological conditions associated with high concentrations.) The meteorological data used for the CART analysis were collected from three air quality monitoring stations close to the border region in Calexico for 2001 and 2002. They include maximum daily temperature, mean wind direction, mean wind velocity, mean solar radiation, and maximum and minimum humidity. Based on the CART analysis, two episodes were selected: one, August 18–27, 2001, was intended to capture high O3 in summer, and a second, January 6–15, 2002, high CO and PM in winter.

6.2.2. Mesoscale Meteorological Model

We used a fifth-generation Mesoscale Meteorological Model (MM5) to simulate atmospheric circulation (Seaman 2000). It consists of several auxiliary programs: TERRAIN horizontally interpolates and generates terrain, land-use, and map-scale data; REGRID uses gridded meteorological data to forecast pressure levels and interpolates these to the horizontal grid and map projection defined by TERRAIN; LITTLE_R develops gridded pressure-level meteorological data (wind, temperature, relative humidity, sea level pressure) used as a first guess; INTERPF transforms data from the above programs to a mesoscale model and performs vertical interpolation; MM5 uses meteorological data generated by other auxiliary programs to predict weather over time; and the Meteorology-Chemistry Interface Processor links MM5 output to other parts of Models-3 framework.

6.2.3. Sparse Matrix Operator Kernel Emissions Module

We used the Sparse Matrix Operator Kernel Emissions Module (SMOKE) to generate spatially gridded emissions files (Houyoux and Vukovich 1999). The input data for SMOKE are raw emissions inventories comprising total annual emissions for area and point sources and total monthly emissions for mobile sources. The output data are hourly emissions for each grid cell and pollutant. SMOKE includes both criteria and toxic pollutants and accommodates both biogenic and nonbiogenic (area, point, and mobile) emissions sources. The data input into SMOKE were drawn from the 2001 U.S. National Emissions Inventory and the 1999 Big Bend Regional Aerosol and Visibility Observational Study and 1999 Mexican National Emissions Inventory of the six Mexican border states.

6.2.4. Community Multiscale Air Quality Model

The Community Multiscale Air Quality (CMAQ) model with the decoupled direct method represents the state-of-the-science for modeling air quality (Byun et al. 2005; Napelenok et al. 2006). It consists of several processors: the initial conditions processor provides concentration data for pollutants for the first hour of the simulation; a boundary conditions processor creates concentration data for the ends of the domain grids; a photolysis rate processor calculates photo-dissociation reaction rates; and a chemical-transport model processor simulates relevant atmospheric chemistry and transport processes.

Using the CMAQ model to estimate the effect of the Intergen and Sempra plants on ambient concentrations of O3 and PM2.5—the two pollutants in our TAF health models—is challenging, for several reasons. First, O3 and some PM2.5 are generated by chemical reactions involving precursor pollutants, such as NOx and VOCs, in processes that are complex and nonlinear. Second, both are regional pollutants affected by processes spanning large areas. Finally, natural gas–fired power plants are much cleaner than coal-fired plants and emit relatively low levels of pollution.

Two general strategies are available for estimating the effect of the Intergen and Sempra plants on ambient concentrations. The first is the brute force method, which entails simulating air quality first without the plants (i.e., determining a baseline) and then with the plants, and comparing the two simulations over space. However, this approach tends to be reasonably accurate only for large-scale emission changes. The second strategy is to use the decoupled direct method, which calculates the derivative of pollutant concentrations with respect to changes in emissions directly from the governing equations of the air quality model and then linearly extrapolates the derivative using the specific emission change associated with the power plants (Cohan et al. 2005). We use this second strategy because the emissions from the Intergen and Sempra power plants are relatively small and locally concentrated and would not be accurately captured using the brute force approach (Cohan et al. 2005).

The CMAQ model does not explicitly account for uncertainty in the relationships between model inputs (emissions, meteorology, topography, land use) and spatial air quality data by, for example, predicting distributions of air quality results instead of deterministic values. That said, the CMAQ model performance in our study area was evaluated by comparing predicted values of ambient O3 with actual O3 measurements for baseline scenarios. These evaluations indicated that the model preformed quite well (Mendoza-Domínguez et al. 2011).

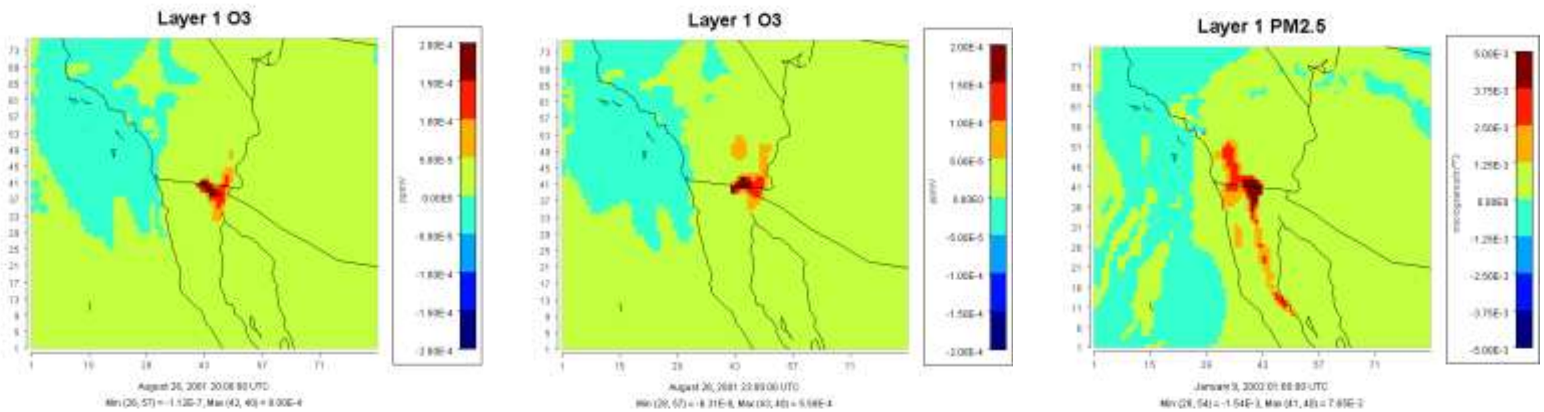
6.3. Model Results

This section describes the Models-3 system's estimates of the effect of the Intergen and Sempra plants on concentrations of O3 and PM2.5 during the two 10-day study episodes: August 18–27, 2001, a summer peak O3 period, and January 6–15, 2002, a winter peak PM 2.5 period.

6.3.1. Intergen Plant

Ozone. During the January episode, the Intergen plant's effect on ambient O3 concentrations is negligible. During the August episode, however, the Intergen plant's peak O3 effect is 8×10^{-1} ppbv. During this episode, prevailing local winds blow northwest while synoptic winds tend to have a northeasterly direction (Vanoye and Mendoza-Domínguez 2009). As a result, O3 plumes from the Intergen plant affect Calexico, just north of Mexicali, and the border region between Mexico and Arizona (Figures 3a and 3b). O3 plumes are transported into Arizona, and plumes of up to 3×10^{-2} ppbv affect Grand Canyon National Park, east of Las Vegas. Plumes also move southeast into Sonora, Mexico.

Figures 3a, 3b, 3c. Simulated Plumes from Intergen Plant: Ozone (O3) on August 26, 2001 and Particulate Matter Smaller than 2.5 microns (PM2.5) on January 9, 2002



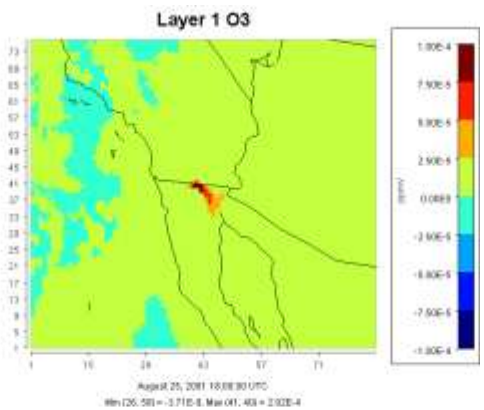
(a) (b) (c)

PM2.5. The Intergen plant's peak effect on PM2.5 occurs during the January episode, when plume concentrations reach 2.0×10^{-1} $\mu g/m^3$ near the power plant (Figure 3c). A PM2.5 plume from the Intergen plant moves both southeast into Baja California and northwest into southern California.

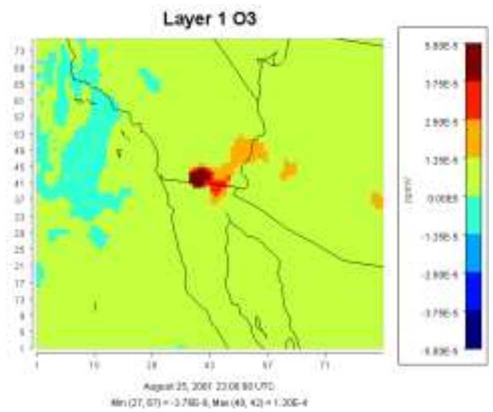
6.3.2. Sempra Plant

Ozone. During the January episode, the Sempra plant's effect on ambient O_3 concentrations is negligible. During the August episode, however, its peak O_3 effect is 2.9×10^{-1} ppbv over the Mexicali-Calexico border region (Figure 4a). Some plumes are also transported to California and Arizona (Figure 4b).

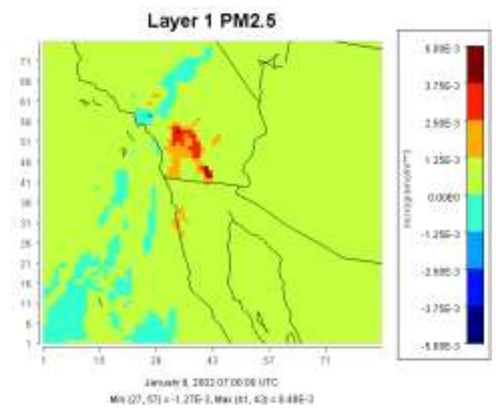
Figures 4a, 4b, 4c. Simulated Plumes from Sempra plant: Ozone (O3) on August 25, 2001 and Particulate Matter Smaller than 2.5 microns (PM2.5) on January 9, 2002



(a)



(b)



(c)

*PM2.5.*The Sempra plant's peak effect on PM2.5 occurs during the January episode, when concentrations reach 2.1×10^{-1} μg/m^{3}south of Mexicali (Figure 4c). Plumes from the plant also are transported to California.

7. Health Impacts: Incidence

7.1. Model

Our health impacts incidence analysis estimates the number of cases of human morbidity and mortality caused by exposure to O3 and PM2.5 attributable to the Intergen and Sempra plants. This analysis entailed three steps, each of which is described below in separate subsection.

7.1.1 CART Analysis: Average Annual Pollutant Concentrations

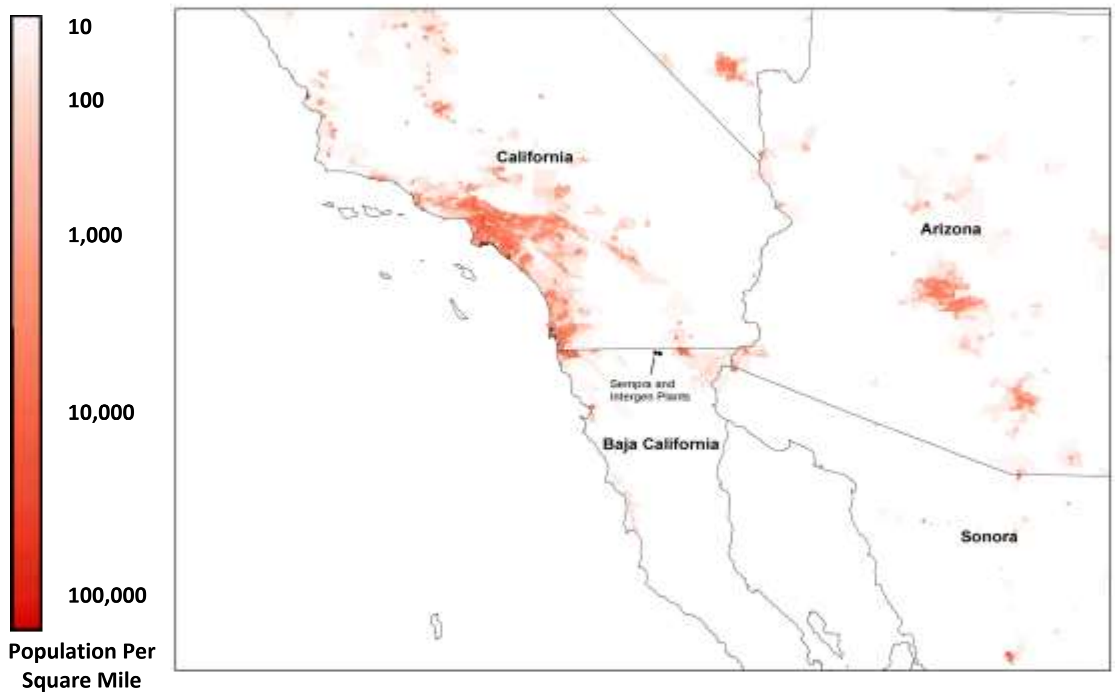
The first step in generating estimates of health impacts from Intergen and Sempra air pollution was to convert the output of the air quality models into the format required by our health model (Figure 2). As discussed above, our air quality models generate estimates of hourly concentrations of O3 and PM2.5 during two 10-day episodes (August 18–27, 2001, and January 6–15, 2002). However, to estimate health impacts, we require data on annual changes in human exposure to pollutants, which in turn requires data on average annual concentrations of O3 and PM2.5. Following Palmer et al. (2007), Boylan et al. (2005), and others, we converted changes in daily concentrations to changes in annual concentrations using a weighted average. A CART analysis was used to construct the weights. As discussed in Section 6.2.1, CART analysis is a nonparametric statistical technique that essentially determines the extent to which each episode-day is representative of meteorological conditions associated with high O3 and PM2.5 concentrations. We used 14 of our 20 episode-days to calculate weighted averages, omitting the first three days of each period because they are a stabilizing period.

7.1.2. Demographic Model: Average Annual Human Exposure

The second step in generating estimates of health impacts from Intergen and Sempra air pollution was to calculate the number of people in different age groups exposed to this pollution. To do this, we obtained year 2000 census data at the level of *areas geoestadíticas básicas* for two Mexican border states (Baja California and Sonora) and census tracts for three U.S. border states (Arizona, California, and Nevada). Figure 5 presents the results of this exercise. Next, we used geographic information system software to overlay these demographic data onto gridded data on changes in average annual concentrations of O3 and PM2.5 from our air quality and

CART models, and to calculate the number of people in various age groups exposed to this pollution.

Figure 5. Population Density in Study Area



7.1.3. Tracking and Analysis Framework: Incidence of Human Mortality and Morbidity

The final step in generating estimates of health impacts from Intergen and Sempra air pollution was to use our estimates of human exposure in combination with concentration response (CR) coefficients drawn from the epidemiological literature on air pollution to estimate incidence of human morbidity and mortality. To do this, we used the 2006 version of the TAF, an integrated tool for benefit-cost analysis created to evaluate proposals for abating sulfur dioxide emissions (which cause acid precipitation) and later updated and used in numerous other studies (Bloyd et al. 1996; Lankton 2006; Palmer et al. 2007). We use TAF to model incidence rates for 15 health endpoints: mortality from O3, mortality from PM2.5, 2 types of morbidity from O3, and 11 types of morbidity from PM2.5 (Table 2).

Table 2. Health Effects of Ozone (O3) and Particulate Matter Smaller than 2.5 Microns (PM2.5) Exposure Modeled by TAF, by Age Group and Method Used to Value Effects

Linked to O3 exposure	**Age group**	**Valuation method**
Respiratory hospital admissions	< 2	COI
	65 +	COI
Asthma emergency room visits	All ages	COI
Short-term mortality	All ages	WTP
Linked to PM2.5 exposure		
Mortality	< 1	WTP
	30 +	
Chronic bronchitis	27 +	WTP
Nonfatal heart attacks	18 +	COI
Respiratory hospital admissions	All ages	COI
Cardiovascular hospital admissions	18–64	COI
	65 +	COI
Asthma emergency room visits	All ages	COI
Acute bronchitis in children	8–12	WTP
Upper respiratory symptoms in children	9–17	WTP
Lower respiratory symptoms in children	7–14	WTP
Asthma exacerbation	5–17	WTP
Work loss days	18–64	COI
Minor restricted activity days	18–64	WTP

COI = cost of illness; WTP = willingness to pay.

Of the various types of morbidity included in the TAF model, most are self-explanatory, but a few require clarification. A minor restricted activity day, as defined by the U.S. Department of Health and Human Services in its Health Interview Survey, refers to restrictions on daily activities that are less severe than spending the day in bed or missing work or school, but more serious than sneezing or coughing that does not restrict activity. Short-term mortality refers to premature mortality predicted by econometric studies relating daily mortality rates in a given city to daily measures of air pollution and other variables, such as temperature. Asthma exacerbation refers to an episode of coughing, wheezing, or shortness of breath caused by asthma.

To estimate incidence rates for the 15 health endpoints in Table 2, TAF relies on a set of CR coefficient distributions, drawn from the epidemiological literature, that indicate the marginal probability of a range of cases of mortality and morbidity due to a change in exposure to O3 and PM2.5 given a baseline rate of mortality or morbidity. For many of these health endpoints, TAF uses a weighted average of two or more CR distributions reported in the literature.[4] Most baseline incidence rates were obtained from the BenMap model used by U.S. EPA for regulatory analyses (U.S. EPA 2011). Appendix 2 provides details on the TAF CR functions.

TAF uses a Monte Carlo numerical simulation procedure to generate 95 percent confidence intervals around mean predictions of the incidence of each health endpoint. For each heath endpoint, TAF randomly chooses values from the probability distribution for each CR coefficient and then calculates an incidence rate based on the chosen values. It repeats this process hundreds of times to generate a distribution of estimated incidence rates, which is used to calculate 95 percent confidence intervals. Note these confidence intervals do not account for uncertainty in our air quality modeling and therefore are likely understate the true variability in our results.

A complication arises because our study domain spans two countries in which epidemiological responses to air pollution may differ because of socioeconomic and environmental conditions, baseline health status, health care quality, and other factors. Ideally, we would use CR distributions drawn from epidemiological studies in the United States to estimate U.S. health impacts, and CR distributions from Mexico to estimate Mexican health impacts. However, we have used the TAF module, based mostly on U.S. epidemiological studies, for both countries, for several reasons. First, far fewer epidemiological studies are available for Mexico than for the United States, so even if we attempted to use separate CR distributions for Mexico, we still would be forced to use many TAF CR distributions to fill gaps. Second, using different CR distributions for the two countries would make it difficult to determine whether disparities in health impacts arose from differences in exposure or differences in CR distributions. Third, as discussed below, exposure to Intergen and Sempra power plant pollution is far greater in the United States, so it is arguably less important to account for differences in epidemiological responses in Mexico. And finally, given the resources available for this study, constructing a different health impacts model for each country was simply not feasible.

[4] The weights (which are the default weights in TAF) mimic the methods that the U.S. Environmental Protection Agency (EPA) used in its analysis of the proposed Interstate Air Quality Rule and Nonroad Diesel Rule.

7.2. Results

Table 3 reports TAF model estimates of the annual incidence of human mortality and morbidity due to O3 and PM2.5 pollution attributable to Intergen and Sempra plants. A few broad trends are apparent. First, overall, health impacts from both plants are quite limited. Of the 15 health endpoints included in the TAF model, mean predicted annual incidence is zero for 5, and less than one for 13. Second, O3 in particular has negligible effects. Even at the low end of the 95% confidence interval, predicted incidence is zero for all ozone health endpoints. Third, the incidence of less serious health endpoints, such as work loss days and minor restricted activity days, due to PM2.5 is not insignificant. For example, the mean estimate for all minor restricted activity days is 366 cases per year, and the mean estimate for all work loss days is 65 per year. Finally, most health effects from the two plants occur in the United States. The average ratio of (nonzero) health effects in the United States versus Mexico is 2.3.

What explains these results? The modest overall health effects stem from the fact that, as GAO (2005) emphasizes, the Intergen and Sempra plants' emissions are low relative to coal-fired plants and older gas-fired plants. In addition, most of the areas covered by the Intergen and Sempra plants' emissions plumes are sparsely populated (Figures 3, 4, and 5). Three factors explain our finding that effects from NOx are completely negligible: (i) as just noted, the plants are relatively clean and the downwind areas are sparsely populated; (ii) in contrast to the PM2.5 health endpoints, all the NOx health endpoints are relatively serious, involving either hospital visits or mortality; and (iii) titration (a phenomenon in which NOx "scavenges," or reduces, O3 during certain times of the day) offsets the contribution of NOx to the formation of O3 (Mendoza-Domínguez et al. 2011; Seinfeld and Pandis 1997; DOE 2004). Finally, our finding that PM2.5 from Intergen and Sempra has nontrivial health effects, most of which occur in the United States, stems from the fact that the plants' PM2.5 plumes are transported northwest into inhabited parts of southern California.

Table 3. Annual Cases of Human Mortality and Morbidity from Pollution Generated by Intergen and Sempra Plants, by Pollutant and Country: Mean Estimates and 95% Confidence Intervals

Pollutant	Health endpoint	Intergen						Sempra					
		United States			Mexico			United States			Mexico		
		Low	Mean	High	Low	Mean	High	Low	Mean	High	Low	Mean	High
O3	Respiratory hospital admissions, ages 65+	0.00	**0.00**	0.00	0.00	**0.00**	0.00	0.00	**0.00**	0.00	0.00	**0.00**	0.00
	Respiratory hospital admissions, ages < 2	0.00	**0.00**	0.00	0.00	**0.00**	0.00	0.00	**0.00**	0.00	0.00	**0.00**	0.00
	Asthma emergency room visits	0.00	**0.00**	0.00	0.00	**0.00**	0.00	0.00	**0.00**	0.00	0.00	**0.00**	0.00
	Short-term mortality	0.00	**0.00**	0.00	0.00	**0.00**	0.00	0.00	**0.00**	0.00	0.00	**0.00**	0.00
PM2.5	Mortality, ages < 1	0.00	**0.00**	0.00	0.00	**0.00**	0.00	0.00	**0.00**	0.00	0.00	**0.00**	0.00
	Mortality, ages 30+	0.01	**0.04**	0.08	-0.01	**0.02**	0.04	0.01	**0.03**	0.05	-0.01	**0.02**	0.03
	Chronic bronchitis, ages 18+	0.03	**0.17**	0.34	-0.01	**0.03**	0.06	0.02	**0.11**	0.23	0.00	**0.02**	0.04
	Nonfatal heart attacks, ages 18+	0.01	**0.09**	0.16	0.01	**0.07**	0.12	0.01	**0.06**	0.11	0.01	**0.05**	0.08
	Respiratory hospital admissions	-0.01	**0.03**	0.05	0.01	**0.03**	0.06	-0.01	**0.02**	0.03	0.00	**0.02**	0.04
	Cardiovascular hospital admissions, ages 18–64	0.00	**0.01**	0.01	0.00	**0.01**	0.01	0.00	**0.00**	0.01	0.00	**0.00**	0.01
	Cardiovascular hospital admissions, ages 65+	0.01	**0.01**	0.02	0.00	**0.00**	0.01	0.00	**0.01**	0.02	0.00	**0.00**	0.00
	Asthma emergency room visits, ages < 18	0.01	**0.02**	0.03	0.02	**0.04**	0.06	0.01	**0.02**	0.02	0.01	**0.02**	0.04
	Acute bronchitis in children, ages 8–12	-0.04	**0.11**	0.26	-0.01	**0.16**	0.35	-0.03	**0.08**	0.18	0.00	**0.11**	0.24
	Upper resp. symptoms in children, ages 9–17	1.28	**12.02**	28.07	0.45	**6.43**	13.38	0.87	**8.19**	19.12	0.31	**4.44**	9.24
	Lower resp. symptoms in children, ages 7–14	2.25	**4.93**	9.65	0.56	**1.70**	2.71	1.53	**3.36**	6.57	0.39	**1.18**	1.89
	Asthma exacerbations, ages 6–18	0.75	**4.88**	7.34	0.91	**1.80**	2.76	0.51	**3.32**	5.00	0.63	**1.25**	1.92
	Work loss days, ages 18–64	27.15	**30.54**	34.88	6.82	**7.89**	9.68	18.49	**20.79**	23.75	4.70	**5.43**	6.66
	Minor restricted activity days, ages 18–64	142.19	**173.28**	211.58	37.21	**44.68**	54.01	96.81	**117.98**	144.06	25.61	**30.76**	37.17

8. Health Impacts: Valuation

8.1. Model

We use the valuation module of TAF to assign monetary values to the estimates of the incidence of human mortality and morbidity discussed above (Figure 3). This module consists of valuation functions drawn from the environmental and health economics literatures. For details on the TAF valuation functions, see Appendix 2. As in the health effects incidence module of TAF, several functions are sometimes used in conjunction with a single health impact. In such cases, estimates for each are a weighted average of estimates from each valuation function.[5] Also, the valuation module of TAF, like the health impacts module, uses Monte Carlo techniques to generate 95 percent confidence intervals around mean value estimates.

The TAF valuation module uses two types of valuation functions: cost of illness (COI) and willingness to pay (WTP) (Table 2). COI functions value health impacts by estimating the pecuniary and nonpecuniary expenses paid by individuals and insurance companies for illness, including the payments for actual health care, lost wages, and opportunity costs of time. WTP functions aim to capture the maximum amount individuals would be willing to pay to avoid illness or the risk of premature death. WTP is estimated by revealed- or stated-preference methods. Reveled-preference methods tease out individuals' WTP from market behavior affected by health concerns. For example, some revealed-preference studies estimate WTP by examining the correlation between wages and occupational safety hazards. Stated-preference methods involve developing and administering surveys designed to elicit individuals' true preferences for avoiding health risks (Freeman 1993).

Not surprisingly, monetary values assigned to mortality, known as the value of a statistical life (VSL), are generally an order of magnitude greater than those assigned to morbidity. We use the VSL estimate from Mrozek and Taylor (2002), which has a central value of $2.324 million. This estimate is quite conservative: it is at the low end of the values used in benefit-cost analysis. For example, 2009 U.S. EPA rules mandate that

[5] As in the case of the health impacts incidence analysis, the weights (which are the default weights in TAF) mimic the methods EPA used in its analysis of the proposed Interstate Air Quality Rule and Nonroad Diesel Rule.

benefit-cost analyses use a VSL of $7.9 million, and 2009 U.S. Department of Transportation rules mandate a VSL of $6.0 million (Copeland 2010).

Like the incidence of mortality and morbidity, the values of mortality and morbidity probably differ between the two countries, in this case because of differing perceptions of mortality risk and differing types and costs of medical care, among other factors. Ideally, we would use Mexican COI and WTP parameters to estimate the value of Mexican health damages. However, to estimate values of health impacts in Mexico would require collecting COI and WTP data that are comparable to those for the United States, in terms of both the type and the severity of illness. Unfortunately, to our knowledge, comparable data are not available from secondary sources for Mexico, and collecting them from primary sources is beyond the scope of our effort. Hence, we use the U.S. COI and WTP parameters in TAF to value Mexican mortality and morbidity. This is arguably acceptable in our study because Mexican health damages are minor compared with those in the United States. Also, using the same valuation functions avoids the difficult issue of valuing health impacts differently for two populations that are geographically, culturally, and economically close. That said, studies typically assign lower values to health impacts in developing countries (Alberini et al. 1997; Loehman et al. 1979). Therefore, our estimates of the value of Mexican morbidity and mortality are likely biased upward.

8.2. Results

The patterns of the valuation results mirror those described in the above discussion of health impacts incidence: although overall health effects are limited, particularly from O3, health effects from PM2.5 are nontrivial, and most health effects are in the United States, not Mexico (Table 4). The valuation models simply put numbers to these findings. Our mean estimates of the annual value of health damages attributable to Intergen emissions are $230,000 in the United States and $104,000 in Mexico. Mean estimates of annual damages attributable to Sempra emission are $160,000 in the United States and $72,000 in Mexico. The total value of annual health damages attributable to both plants is $566,000.[6]

[6] The $566,000 figure is the sum of the total mean values of the Intergen plant’s health damages in the United States ($230,000) and in Mexico ($104,000) and the Sempra plant’s health damages in the United States ($160,000) and in Mexico ($72,000). All these values are in the bottom row of Table 4.

Table 4. Annual Value of Human Mortality and Morbidity from Pollution Generated by Intergen and Sempra Plants, by Pollutant and Country: Mean Estimates and 95% Confidence Intervals (thousands of year-2000 US$)

Pollutant	Health endpoint	Intergen						Sempra					
		U.S.			Mexico			U.S.			Mexico		
		Low	Mean	High	Low	Mean	High	Low	Mean	High	Low	Mean	High
O3	Respiratory hospital admissions, ages 65+	1	**4**	7	0	**0**	0	0	**2**	3	0	**0**	0
	Respiratory hospital admissions, ages < 2	1	**3**	5	0	**1**	2	0	**1**	2	0	**0**	1
	Asthma emergency room visits	0	**0**	0	0	**0**	0	0	**0**	0	0	**0**	0
	Short-term mortality	6	**10**	16	1	**2**	3	2	**4**	6	1	**1**	1
	Subtotal	8	**18**	28	1	**3**	5	3	**6**	10	1	**2**	2
PM2.5	Mortality, ages < 1	117	**455**	891	93	**457**	972	83	**322**	630	64	**314**	666
	Mortality, ages 30+	19,531	**100,976**	175,452	-17,114	**56,759**	105,587	13,757	**71,125**	123,584	-11,850	**39,303**	73,114
	Chronic bronchitis, ages 18+	9,017	**57,260**	115,565	-2,198	**9,520**	19,400	6,137	**38,969**	78,649	-1,519	**6,577**	13,404
	Nonfatal heart attacks, ages 18+	3,282	**24,344**	43,855	3,850	**19,147**	33,559	2,328	**17,262**	31,097	2,651	**13,184**	23,108
	Respiratory hospital admissions	-923	**1,719**	3,078	428	**1,691**	3,911	-653	**1,216**	2,176	295	**1,166**	2,697
	Cardiovascular hosp. admissions, ages 18–64	233	**599**	1,097	364	**558**	864	165	**425**	779	251	**384**	595
	Cardiovascular hospital admissions, ages 65+	572	**1,265**	2,052	249	**358**	497	402	**890**	1,444	173	**249**	345
	Asthma emergency room visits, ages < 18	10	**26**	43	21	**43**	77	7	**18**	30	14	**29**	53
	Acute bronchitis in children, ages 8–12	-15	**42**	106	-4	**61**	169	-11	**29**	74	-3	**43**	118
	Upper resp. symptoms in children, ages 9–17	127	**2,406**	5,678	69	**1,318**	4,523	86	**1,639**	3,867	48	**911**	3,125
	Lower resp. symptoms in children, ages 7–14	98	**317**	642	25	**109**	265	66	**216**	437	17	**76**	184
	Asthma exacerbations, ages 6–18	201	**934**	1,778	128	**342**	703	137	**636**	1,211	89	**238**	488
	Work loss days, ages 18–64	4,041	**4,545**	5,192	3,842	**4,440**	5,450	2,868	**3,225**	3,684	2,644	**3,055**	3,750
	Minor restricted activity days, ages 18–64	17,193	**34,626**	60,330	3,933	**9,632**	14,619	11,706	**23,576**	41,078	2,707	**6,629**	10,062
	Subtotal	53,483	**229,514**	415,758	-6,314	**104,433**	190,595	37,079	**159,549**	288,740	-4,419	**72,157**	131,710
	TOTAL	53,491	**229,532**	415,786	-6,312	**104,437**	190,600	37,082	**159,555**	288,751	-4,418	**72,158**	131,712

9. Conclusion

We have used a suite of air quality, health impacts, and valuation models to analyze the effects on human morbidity and mortality of air pollution from the Intergen and Sempra power-exporting plants located just south of the California border. As discussed above, our analysis has limitations: our estimates of annual changes in air quality are extrapolated from model predictions for two 10-day episodes; confidence intervals for our health impacts and valuation results do not account for uncertainty in our air quality modeling; and we use U.S. CR and valuation functions for Mexico. That said, we find that Intergen and Sempra air emissions have limited but nontrivial health effects, mostly by exacerbating PM2.5 pollution in the United States. In total, the cost of the health damages is more than half a million dollars per year.

Although we find that overall health effects from the plants are limited, our modeling clearly indicates that U.S.-owned power-exporting plants can have cross-border health effects in the United States. Of course, the magnitude of these cross-border effects depends on the specific characteristics of the plants (e.g., size, emissions characteristics, and abatement technologies) and of airsheds in which they are sited (e.g., demography and meteorology). In principle, these effects could be quite significant, a risk that makes systematic evaluation using the type of modeling strategies described in this paper advisable.

Presumably, cross-border health effects could be largely avoided if the plants had mandates or incentives to reduce or offset their emissions. What policies could be used to achieve this end? Three broad approaches have been proposed (GAO 2005). One is to legally mandate that power-exporting plants meet U.S. emissions standards and offset requirements. This could be done either by enacting new federal or state legislation or by modifying DOE regulations for granting presidential permits to power-exporting plants for the use of cross-national transmission lines.

A second, far more ambitious option is to develop a cross-border cap-and-trade system (Erickson et al. 2004; Johnson and Alvarez 2003). An aggregate cap on emissions in a national airshed would be negotiated and all sources in the airshed would be assigned permits that entitle them to quantities of emissions commensurate with the aggregate cap. Sources would be allowed to buy and sell permits, creating incentives for a cost-effective allocation of abatement across facilities.[7] Implementing such a program faces daunting challenges, however, including developing accurate emissions inventories covering all sources in the binational airshed; harmonizing U.S. and Mexican air pollution permitting; negotiating international agreements on aggregate caps, individual permit levels, and program rules; establishing a binational institution to administer the program; and coordinating monitoring and enforcement. Given these requisites, it is perhaps not surprising that this approach has yet to be piloted.[8]

A third option is to establish a binational trust fund to identify, prioritize, and fund specific air quality management projects (Ryan et al. 2008; GNEB 2010). Funding could be derived from, for example, direct appropriations from U.S. and Mexican legislatures, fees charged to cross-border commuters, or taxes on large emissions sources, such as power plants. International coordination would be required to collect and administer the trust fund. Advocates have argued that the fund could be housed within the bilateral institutions created by the environmental side agreements to the North American Free Trade Agreements, specifically the North American Development Bank, which finances environmental infrastructure projects, and the Border Environmental Cooperation Commission, which assists local communities in developing environmental infrastructure. Questions have been raised about the efficiency and responsiveness of both institutions, however (Ryan et al. 2008).

Of those three options, the first—regulatory mandates for air pollution control for power-exporting plants—is almost certainly the most feasible. Unlike the other two options, it requires no international coordination and very little institution building. Indeed, as noted in the introduction, in September 2010, one border state, California, passed legislation (Senate bill 2037) that requires new electricity-generating units in Mexico that sell power to the state to comply with all the air pollution control regulations—including best available control technology standards and offset requirements—applicable in the California air basin adjacent to the facility. The bill does not apply retroactively to the Intergen and Sempra plants in Mexicali, only to new

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[7] Facilities with low abatement costs would have incentives to cut their emissions and sell their excess permits to plants with higher abatement costs. Therefore, the burden of abatement would be shifted from high abatement cost plants to low abatement cost plants (Sterner 2003).

[8] A less ambitious baseline-and-trade program has been piloted in Texas. The program allows Texas facilities on the U.S.–Mexico border to meet their pollution reduction mandates by purchasing offsets from sources in Mexico with which they share an airshed (Erickson et al. 2004).

facilities built after January 1, 2011, and to additional capacity installed at existing facilities after that date.

California's legislation has not resolved the problem of transborder pollution from Mexican power-exporting plants, however. It remains to be seen whether the three other U.S. border states will follow California's lead. Also, given that lower permitting, capital, and labor costs in Mexico—not lower environmental regulatory costs—appear to have driven Intergen's and Sempra's decisions to build there, it is reasonable to expect more power-exporting plants to be built if energy supply in the border region falls short of demand again, regardless of whether such legislation is passed.

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Appendix 1. Method Used to Convert Concentrations of NOx, NH3, and CO (in ppm) to Annual Mass Emissions Rate (in short tons/yr) for Intergen and Sempra Plants

For three pollutants—NOx, NH3, and CO—we estimated the annual mass emissions rate from data on stack gas concentrations from GAO (2005) along with information on fuel consumption from DOE (2004). We used this method because data on exhaust flow rates, which are often used to estimate mass emissions rates, are not available. We used the following formula

$$MER = GC \times ERTS/2000$$

$$ERTS = CON \times CF \times MW \times (1/SMV) \times 10^{(-6)} \times FD$$

where the parameters (along with units and sources) are defined in Table A1.

Table A1. Parameters Used to Convert NOx, NH3, and CO Concentrations to Annual Mass Emissions for Intergen and Sempra Plants

Parameter	Definition	Unit	Value for NOx	Value for NH3	Value for CO	Source
CF	Conversion factor: 15% O2 Concentration to 0% O2	n/a	(20.9/(20.9-15))	(20.9/(20.9-15))	(20.9/(20.9-15))	Walter Smith and Assoc. Inc.
CON (Sempra)	Pollutant concentration at 15% O2	ppmdv	2.21[a]	0.43[a]	0	GAO (2005) p. 13, Table 2
CON (Intergen)	Pollutant concentration at 15% O2	ppmdv	2.59[b]	1.36[b]	0.83[b]	GAO (2005) p. 13, Table 2
ERTS	Emissions rate of thermal system	lbs/MMBtu	—	—	—	calculated
FD	Dry oxygen F-factor for natural gas	dscf/MMBtu	8710 at 68°F	8710 at 68°F	8710 at 68°F	40 C.F. R. Part 60 Appendix A 2.1, or EPA Method 19, page 1144, Table 19-2
GC (Sempra)	Gas consumption	MMBtu/yr	3,8400,000 for turbines 1&2	3,8400,000 for turbines 1&2	3,8400,000 for turbines 1&2	DOE (2004) Appendix G, Table G-1
GC (Intergen)	Gas consumption	MMBtu/yr	68,500,000 for turbines 1-4	68,500,000 for turbines 1-4	68,500,000 for turbines 1-4	DOE (2004) Appendix G, Table G-1
MER	Mass emissions rate	Short tons/yr	—	—	—	Calculated
MW	Molecular weight	u	46	17	28	
SMV	Specific molar volume at 68°F	dscf/lb-mole	385.3	385.3	385.3	US EPA AP-42. "ref02_c01s02.pdf" available at: www.epa.gov/ttn/chief/old/ap42/ch11/s17/reference/ref_11c11s17.pdf

[a] Average for turbines 1 and 2.
[b] Average for turbines 1-4.

Appendix 2. Tracking Analysis Framework (TAF) Health Impacts and Valuation Models

As noted in Sections 7 and 8,, many of the concentration-response (CR) and valuation distributions in the TAF model are actually weighted averages of several distributions reported in the literature. Drawn from Lankton (2006), this appendix provides information on these distributions. For the health effects model, this information includes the originating study, the time period used to measure the pollutant, the CR functional form, the mean and standard deviation of the distribution, what other pollutants were included in the study, and the weight assigned to each study if multiple studies were used to generate a weighted average distribution for a single health endpoint. For the valuation module, this information includes the originating study and the valuation function. Note that the weights we use are the default weights in the TAF model. They approximate the weights that EPA used when pooling both CR and (separately) valuation studies in its analysis of the proposed Interstate Air Quality Rule and Nonroad Diesel Rule.

A2.1. Ozone

For each of the five ozone-related health effects, the concentration-response and valuation studies are summarized in Tables A2 and A3.

Respiratory Hospital Admissions

Respiratory hospital admissions (RHAs), in number of incidences per season, are calculated for two age groups: 65 and over, and under 2. The CR functions used are aggregated across three RHA subcategories: asthma, chronic lung disease, and pneumonia.

RHA valuation is calculated separately for each age group. However, the only difference between the two age-group studies is the parameter for mean length of hospital stay. CPI Med and ECI Wage are price indices from BenMap.

Asthma Emergency Room Visits

Asthma emergency room visits (AERVs), in number of incidences per season, are calculated for all ages (aggregate). The AERV CR functions in the TAF model are based on five-hour average ozone concentrations, for which eight-hour average ozone concentrations are substituted. This results in a lower-bound estimate of AERV incidences and valuation.

AERV valuation is calculated for all ages (aggregate). ECI Wage is a price index from BenMap. There are two valuation studies for AERV (these valuation studies are also used to value AERVs due to PM2.5 exposure):

School Absence Days

School absence days (SADs), in number of incidences per season, are estimated for the age group 5 to 17 (inclusive). The SAD CR functions used are Gilliland et al. (2001) for ages 9 to 10 and Chen et al. (2000) for ages 6 to 11. TAF applies a weighted average of these studies to the population age group 5 to 17. The SAD baseline incidence rates are from a 1996 National Center for Educational Statistics study.

There is one SAD valuation study in TAF. Each SAD is valued at $75 and adjusted by ECI Wage, a price index from BenMap.

Minor Restricted Activity Days

Minor restricted activity day (MRAD) incidences are calculated by season for the age group 18 to 64 (inclusive). Baseline incidence rates for MRADs are from Ostro and Rothschild (1989).

For the purposes of valuation, the model treats minor respiratory restricted activity days (MRRADs) and MRADs interchangeably because no studies have determined willingness to pay to avoid MRADs (see BenMap documentation, page H-16). The MRAD valuation study in TAF is Industrial Economics Incorporated (IEc 1993), a contingent valuation study for willingness to pay of an adult to avoid a three-symptom day of coughing, throat congestion, and sinusitis. This study is also used when estimating the value of MRAD incidences related to PM2.5 exposure.

Short-term Mortality

Ozone-related short-term mortality (STM) incidences are calculated by season for all ages (aggregate). Baseline incidence rates are from BenMap.

Valuation for short-term mortality related to ozone exposure is estimated using the value of a statistical life.

Table A2. Ozone Concentration-Response Distributions in TAF Model

Health endpoint Study (year)	Time period (Location)	CR functional form Mean (s.d.)	Other pollutants	Weight
Respiratory hospital admissions (RHAs) *Age 65+*				
Schwartz (1995)	24 hours (New Haven)	Log-linear 0.002652 (0. 001398)	PM10	0.50
Schwartz (1995)	24 hours (Takoma)	Log-linear 0.007147 (0. 002565)	PM10	0.50
Respiratory hospital admissions (RHAs) *Age < 2*				
Burnett et al. (2001)	1 hour (Toronto)	Log-linear 0.006309 (0. 001834)	PM2.5	1.00
Asthma emergency room visits (AREVs) *All ages*				
Weisel et al. (1995)	5 (8) hours (NJ)	Linear 0.0443 (0.00723	None	0.49
Cody et al. (1992)	5 (8) hours (NJ)	Linear 0.0203 (0.00717)	SO_2	0.51
Short-term mortality (STM) *All ages*				
Bell et al. (2004)	24 hours (95 urban communities)	Log-linear 0.000519 (0.0002776)	PM10	1.00

Table A3. Ozone Health Effects Valuation Methods in TAF Model

Health endpoint Study (year)	Valuation function	Weight
Respiratory hospital admissions (RHAs) *Age 65+*		
Abt Associates (2003); Agency for Healthcare Research and Quality (2000)	Incidences × CPI Med + B × I × ECI Wage *B = 6.882932 = Mean length of hospital stay in days* *I = Median daily income* *Year 2000$*	1.00
Respiratory hospital admissions (RHAs) *Age < 2*		
Abt Associates (2003); Agency for Healthcare Research and Quality (2000)	Incidences × CPI Med + B × I × ECI Wage *B = 2. 974239 = Mean length of hospital stay in days* *I = Median daily income* *Year 2000$*	1.00
Asthma emergency room visits (AREVs) *All ages*		
Smith et al. (1997)	Incidences × Distribution *Distribution is triangular, adjusted by ECI wage (min 230.7, mode 311.6, max 430.9, year 2000$)*	0.50

Stanford et al. (1999)	Incidences × Distribution *Distribution is normal, adjusted by ECI wage (mean 260.7, sd 5.225, year 2000$)*	0.50
Short-term mortality (STM) *All ages*		
Mrozek and Taylor (2002) (all ages)	• Incidences × (Value of a statistical life) • Value of a statistical life = $2.324 million × CPI *Year 2000$*	1.00

A2.2. PM2.5

For each of the 12 PM2.5-related health effects, the concentration-response and valuation studies are summarized in Tables A4 and A5.

Mortality

PM2.5 mortality incidences are estimated annually for two age groups: under 1, and 30 and over. Baseline incidence rates are from BenMap.

Valuation for mortality related to PM2.5 exposure is estimated using the value of a statistical life.

Chronic Bronchitis

Chronic bronchitis (CB) incidences are estimated annually for the age group 27 and over. Baseline incidence and prevalence rates are from BenMap.

There are three valuation studies for chronic bronchitis. All three are from the BenMap model, and no specific studies are cited. The two cost-of-illness studies, one with a 3 percent discount rate and one with a 7 percent discount rate, are weighted by age within the 27-and-over age group. The other study is based on willingness to pay to avoid a case of pollution-related chronic bronchitis; this valuation does not vary within the 27-and-over age group.

Nonfatal Heart Attacks

Nonfatal heart attack (NFHA) incidences are estimated seasonally for the age group 18 and over. Baseline incidence rates are from BenMap.

There are two NFHA valuation studies in TAF, both from BenMap with no specific study cited: one with a 3 percent discount rate, and one with a 7 percent discount rate. Both studies incorporate 10 years of medical costs and 5 years of wage costs.

Respiratory Hospital Admissions

Respiratory hospital admission (RHA) incidences related to PM2.5 exposure are estimated seasonally for all ages. Baseline incidence rates are from BenMap.

There is a single RHA valuation study in TAF for RHA incidences related to PM2.5 exposure.

Cardiovascular Hospital Admissions

Cardiovascular hospital admission (CHA) incidences are estimated seasonally for two age groups: 18 to 64 (inclusive), and 65 and over. There are four studies for the 18-to-64 age group. The Moolgavkar (2003) study aggregates all cardiovascular symptoms. The Ito (2003) studies are separated by symptom: ischemic, dysrhythmia, and heart failure. Moolgavkar (2003) uses an aggregate baseline incidence rate; the Ito studies use incidence rates specific to each symptom. All incidence rates are from BenMap. There is a single study for the 65-and-over age group, aggregate for all cardiovascular symptoms.

The CHA valuation parameters for mean hospital charge and mean length of stay differ between the 1-to-64 and 65-and-overage group.

Asthma Emergency Room Visits

Asthma emergency room visit (AERV) incidences related to PM2.5 exposure are estimated seasonally for the age group under 18. Baseline incidence rates

There are two valuation studies for AERVs. These valuation studies are shared with AERV incidences due to ozone exposure.

Acute Bronchitis in Children

Acute bronchitis in children (ABiC) incidences are estimated annually for the age group 8 to 12 (inclusive). Baseline incidence rates are from BenMap.

The valuation study for ABiC is a six-day illness study.

Upper Respiratory Symptoms in Children

Upper respiratory symptoms in children (URSiC) incidences are estimated seasonally for the age group 9 to 17 (inclusive). Baseline incidence and prevalence rates are from BenMap.

The URSiC valuation study is a two-symptom, one-day study from BenMap.

Lower Respiratory Symptoms in Children

Lower respiratory symptoms in children (LRSiC) incidences are estimated seasonally for the age group 7 to 14. Baseline incidence rates are from BenMap.

The LRSiC valuation study is a two-symptom, one-day study from BenMap.

Asthma Exacerbations

Asthma exacerbation (AE) incidences are estimated seasonally for the age group 5 to 17 (inclusive). These incidences are estimated by symptom and then aggregated for total AE incidences. Baseline incidence rates are from BenMap.

Three valuation studies for AE exist in TAF: one symptom day, one bad asthma day, and two bad asthma days. The default TAF valuation study is the one bad asthma day study.

Work Loss Days

Work loss day (WLD) incidences are estimated seasonally for the age group 18 to 64 (inclusive). Baseline incidence rates are from BenMap.

WLD valuation is estimated by adjusting median income by the wage price index from BenMap.

Minor Restricted Activity Days

Minor restricted activity day (MRAD) incidences are estimated seasonally for the age group 18 to 64 (inclusive). Baseline incidence rates are from BenMap.

For the purposes of valuation, the model treats minor respiratory restricted activity days (MRRADs) and MRADs interchangeably because no studies have determined willingness to pay to avoid MRADs (see BenMap documentation, H-16). The MRAD valuation study, Industrial Economics Incorporated (1993), is a contingent valuation study for an adult’s willingness to pay to avoid a three-symptom day of coughing, throat congestion, and sinusitis. These valuation studies are also used when estimating the value of MRAD incidences related to ozone exposure.

Table A4. PM2.5 Concentration-Response Distributions in TAF Model

Health endpoint Study (year)	Time period (Location)	CR functional form Mean (s.d.)	Other pollutants	Weight
Mortality *Age <1*				
Woodruff et al. (1997)	Annual average (86 cities)	Logistic 0.003922 (0.001221)	None	1.00
Mortality *Age 30+*				
Pope et al. (2002)	Annual average (61 cities)	Log-linear 0.004018 (0.001642)	None	1.00
Chronic bronchitis (CB) *Age 27 +*				
Abbey et al. (1995)	Annual average SF, SD, South Coast Basin	Logistic 0.006796 (0.006796)	None	1.00
Nonfatal heart attacks (NFHAs) *Age 18+*				
Peters et al. (2001)	24-hour daily Boston	Logistic 0.02412 (0.009285)	None	1.00
Respiratory hospital admissions (RHA) *All ages*				
Burnett (1997)	24-hour daily Toronto	Log-linear 0.002422 (0.001039)	O3	1.00
Cardiovascular hosp. admissions (CHA) *Age 18–64*				
Moolgavkar (2003)	24-hour daily Los Angeles	Log-linear 0.001568 (0.0003420)	None	0.979
Ito (2003) (Ischemic)	24-hour daily Detroit	Log-linear 0.001435 (0.0001156)	None	0.007
Ito (2003) (Dysrhythmia)	24-hour daily Detroit	Log-linear 0.001249 (0.0002033)	None	0.007
Ito (2003) (Heart Failure)	24-hour daily Detroit	Log-linear 0.003074 (0.0001292)	None	0.007
Cardiovascular hosp. admissions (CHA) *Age 65+*				
Moolgavkar (2000)	24-hour daily Los Angeles	Log-linear 0.001568 (0.0003420)	None	1.00
Asthma emergency room visits (AERV) *All ages*				
Norris et al. (1999)	24-hour daily Seattle	Log-linear 0.01471 (0.003492)	None	1.00
Acute bronchitis in children (ABiC) *Age 8–12*				
Dockery et al. (1996)	Annual average 24 cities	Logistic 0.02721 (0.01710)	None	1.00
Upper resp. symp. in children (URSiC) *Age 9–17*				
Pope et al. (1991)	24-hour daily Utah Valley	Logistic 0.003600 (0.001500)	None	1.00
Lower resp. symp. in children (LRSiC) *Age 7–14*				

Schwartz and Neas (2000)	24-hour daily 6 cities	Logistic 0.01901 (0.006005)	None	1.00
Asthma exacerbation (AE) *Age 5–17*				
Ostro et al. (2001) (cough)	24-hour daily Los Angeles	Logistic 0.000985 (0.000747)	None	0.37
Ostro et al. (2001) (wheeze)	24-hour daily Los Angeles	Logistic 0.001942 (0.000803)	None	0.24
Ostro et al. (2001) (short breath)	24-hour daily Los Angeles	Logistic 0.002565 (0.001335)	None	0.38
Work loss days (WLDs) *Age 18–64*				
Ostro (1987)	24-hour daily nationwide	Log-linear 0.004600 (0.0003600)	None	1.00
Minor restricted activity days (MRADs) *Age 18–64*				
Ostro and Rothschild (1989)	24-hour daily nationwide	Log-linear 0.007410 (0.000700)	None	1.00

Table A5. PM2.5 Health Effects Valuation Methods in TAF Model

Health endpoint Study (year)	Valuation function	Weight
Mortality *Age <1* All ages Mrozek and Taylor (2002)	• Incidences × (Value of a statistical life) • Value of a statistical life = \$2.324 million × CPI *Year 2000\$*	1.00
Mortality *Age 30+* All ages Mrozek and Taylor (2002)	• Incidences × (Value of a statistical life) • Value of a statistical life = \$2.324 million × CPI *Year 2000\$*	1.00
Chronic bronchitis (CB) *Age 27 +* WTP average severity Abt Associates (2003); Krupnick and Cropper (1992); Viscusi et al. (1991)	A × CPI *Adjusted by CPI* *(A = 340481.843750 = WTP to avoid a case of pollution related chronic bronchitis, year 2000\$)*	1.00
Nonfatal heart attacks (NFHAs) *Age 18+*		

COI 3% discount rate Cropper and Krupnick (1990)	A × CPI_Med + B × ECI_Wage *Adjusted by CPI and ECI wage* *A = PDV of medical costs, 3% discount* *B = PDV of opportunity costs, 3% discount* *Parameter values vary by age group:* *18–24: (A = 49650.941406, B = 0.00)* *25–44: (A = 49650.941406, B = 9032.534180)* *45–54: (A = 49650.941406, B = 13313.006836)* *55–64: (A = 49650.941406, B = 76950.710938)* *65 and over: (A = 49650.941406, B = 0.00)*	1.00
Respiratory hospital admissions (RHAs) *All ages*		
Abt Associates (2003); Agency for Healthcare Research and Quality (2000)	A × CPI_Med + B × I × ECI_Wage *Adjusted by CPI and ECI wage* *A = 14999.00 = Mean hospital charge* *B = 5.630323 = Mean length of hospital stay* *I = Median income*	1.00
Cardiovascular hosp. admissions (CHAs) *Age 18–64*		
Abt Associates (2003); Agency for Healthcare Research and Quality (2000)	A × CPI_Med + B × I × ECI_Wage *Adjusted by CPI and ECI wage* *A = 22300.00 = Mean hospital charge* *B = 4.150256 = Mean length of hospital stay (days)* *I = Median income*	1.00
Asthma emergency room visits (AERVs) *Age < 18*		
Smith et al. (1997)	Distribution × CPI Med *Distribution is triangular, adjusted by CPI med (mode 311.6, min 230.7, max 430.9, $ year 2000)*	0.50
Stanford et al. (1999)	Distribution × CPI Med *Distribution is mormal, adjusted by CPI med (mean 260.7, sd 5.225, year 2000$)*	0.50
Acute bronchitis in children (ABiC) *Ages 8–12*		
6-day illness Abt Associates (2003); IEc (1994)	Distribution × CPI *Distribution is uniform, adjusted by CPI (mean 355.849243, min 105.059128, max 606.639404, year 2000$)*	1.00
Upper resp. symp. in children (URSiC) *Ages 9–17*		
2 symptoms, 1 day Abt Associates (2003); IEc (1993)	Distribution × B × CPI *Distribution is uniform, adjusted by CPI* *B = 2.00 = Adjustment of WTP from adult to child (mean 24. 637644, min 9.222648, 43.109287, year 2000$)*	1.00
Lower resp. symp. in children (LRSiC) *Ages 7–14*		

2 symptoms, 1 day Abt Associates (2003); IEc (1993)	Distribution × B × CPI *Distribution is uniform, adjusted by CPI* *B = 2.00 = Adjustment of WTP from adult to child* *(mean 15.573099, min 6.943336, max 24.466366, year 2000$)*	1.00
Asthma exacerbation (AE) *Ages 5–17*		
2 bad asthma days Abt Associates (2003); Rowe and Chestnut (1986)	Distribution × CPI *Distribution is uniform, adjusted by CPI* *(mean 42.793083, min 15.559923, max 70.882629, $ year 2000)*	1.00
Work loss days (WLDs) *Ages 18–64*		
Abt Associates (2003)	I × ECI Wage *I = Median income* *Adjusted by ECI wage*	1.00
Minor restricted activity days (MRADs) *Ages 18–64*		
Abt Associates (2003); IEc (1993)	Distribution × CPI *Distribution is triangular, adjusted by CPI* *(mean 50.55, min 20.71, max 80.37, year 2000$)*	1.00

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