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Abstract

Environmental policy analyses increasingly require the evaluation of benefits from large changes in spatially differentiated public goods. Such changes are likely to induce general equilibrium effects through changes in household expenditures and local migration, yet current practice "transfers" constant marginal values for even the largest changes. Moreover, it ignores important distributional effects of policy.

This paper demonstrates that recently developed locational equilibrium models can provide transferable general equilibrium benefit measures. Our results suggest that taking account of the potential for adjustment and household heterogeneity is important. Applying benefits estimated from this method to the effect of the Clean Air Act amendments in Los Angeles, we find that the estimated annual general equilibrium benefits in 2000 and 2010 are dramatically different by income group and location. The gains range from $33 to about $2,400 per household. These differences arise from variations in air quality conditions, income, and the effects of general equilibrium price adjustment.

Key Words: air quality, clean air act, non-market valuation, Tiebout model.

JEL Classification Numbers: H41, Q25, R13

Contents

General Equilibrium Benefit Transfers for Spatial Externalities: Revisiting EPA's Prospective Analysis

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1. Introduction

Environmental policy analyses increasingly require evaluation of the economic gains from large changes in spatially differentiated externalities. As these changes occur, we can assume the spatial variation in amenities provides scope for individual adjustment and often an observable, general equilibrium response through property markets. Several conceptual efforts have addressed this problem and proposed bounds using partial equilibrium benefit measures to approximate the gains from these types of changes.[1] By contrast, EPA's current assessment practices ignore these issues.[2] Benefit measurement is undertaken as if the policy intervention allows households to avoid a set of implicit expenditures. These avoided costs correspond to the monetized health effects that are estimated, in the case of air pollution, with concentration response (CR) functions. CR models link each pollutant to an observable, health-related, event

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[1] The earliest recognition of these issues can be found in Lind (1973) and Starrett (1981). Examples of the proposed bounds include Bartik (1988) and Kanemoto (1988).

[2] EPA's two largest benefit-cost assessments have been the Retrospective [1997] and Prospective [1999] Analyses. The first examines the period 1970-1990 with and without the Clean Air Act. The second examines the net gains from the 1990 amendments comparing annual net benefits in two years with and without the more stringent emission restrictions.

such as chronic asthmacases or cardiovascular admissions. In the most serious situations (and the ones accounting for the majority of the benefits) the event is a statistical life "saved."

People are treated as passive receptors.[3] While the unit values attributed to the health effects usually correspond to an average willingness to pay (WTP) per case (and not health costs), they are assumed to be constant, and independent of the level of the effects and of household income.[4] There is nothing in current practice that reconciles the independently assessed, implicit expenditures with income or considers whether the environmental changes are large enough that households would change their behavior. As a rule, these types of issues are dismissed because it is argued that practical versions of economically consistent models, capable of being used in a policy context with the level of detail required, do not exist.

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[3] Under ideal conditions, CR functions might be interpreted as reduced form relationships describing the health effect after households had adjusted to their current air quality conditions (see Evans and Smith, 2002). Using them for new situations implies no adjustment of the behavioral pattern from the conditions when they were estimated.

[4] The assumption of no variation in the WTP measure with income is particularly ironic because one of the most widely cited criticisms of benefit analysis is that WTP measures used incorporate an implicit acceptance of the distribution of income. A recent example can be found in Heinzerling and Ackerman (2002), who note in their executive summary that:

> "Cost-benefit analysis is a deeply flawed method that repeatedly leads to biased and misleading results. Far from providing a panacea, cost-benefit analysis offers no clear advantages in making regulatory policy decisions and often produces inferior results, in terms of both environmental protection and overall social welfare…" (p. 1).

These authors highlight the equity issues as one of the four "fatal" flaws, noting that:

> "…cost-benefit analysis ignores the question of who suffers as a result of environmental problems and, therefore threatens to reinforce existing patterns of economic and social inequality…Poor countries, communities, and individuals are likely to express less 'willingness to pay' to avoid environmental harms simply because they have fewer resources. Therefore, cost-benefit analysis would justify imposing greater environmental burdens on them than their wealthier counterparts" (p. 2).

As our description suggests, the practice of benefit-cost analyses rarely meet the theoretical idea that Heinzerling and Ackerman criticize. Most policy studies use constant unit values. Our approach does allow the income distribution to be taken into account.

In previous research we used the locational equilibrium framework to estimate household preferences for housing, air quality, and public education for the Los Angeles area (see Sieg et. al, [2002a]). To illustrate how the model performed in measuring the general equilibrium benefits, we evaluated the gains due to the actual improvements in ozone between 1990 and 1995. This paper considers a different issue, namely, whether the framework can provide transferrable general equilibrium benefit measures consistent with the demands of large-scale environmental policy analysis. We show that the locational equilibrium model meets policy needs—providing a consistent baseline and accommodating the level of detail inherent in a large-scale assessment for a spatial externality. To develop our findings, we use the EPA projected spatial variation in ozone concentrations developed for the recent Clean Air Act prospective analysis to compute general equilibrium household adjustment and prices.

Our results suggest that taking account of the potential for adjustment and household heterogeneity is important. In 2000 and 2010, the estimated annual general equilibrium benefits associated with the ozone improvements due to the Clean Air Act amendments (CAAA) are dramatically different by income group and location within the South Coast Air Quality Management District.[5] The gains range from \$33 to about \$2,400 per household (in 1990 dollars).[6] These differences arise from variations in air quality conditions, income, and the effects of general equilibrium price adjustment. To date, existing methods have been unable to consistently measure all of these effects together.

As we noted at the outset, EPA's recent benefit-cost analyses for the CAAA regulations relies on a damage-cost framework, where benefit estimates are derived by multiplying unit values of health-effect changes attributed to changes in pollution (from the estimated CR functions) with partial equilibrium unit benefit estimates. However, analyses of the benefits from air quality regulations have not always adopted this logic. For example, 20 years ago the Math-Tech (1983) net benefit analysis of alternative primary national ambient air quality standards for particulate matter, developed for EPA's Regulatory Impact Analysis at that time, used both health effects and other economic models.[7] The latter included property value and

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[5] This area includes most of Los Angeles, Orange, Riverside, San Bernadino, and Ventura counties.

[6] This range is based on the average welfare gain computed by income quantile in each of the 102 school districts in our extended model. It is the range from the smallest mean to the largest across the school district/quantile groups.

[7] See Palmquist and Smith (2002) for further discussion of this evaluation.

wage hedonics as well as consumer expenditure models.[8] These other approaches offered a cross-check and, in the case of the expenditure work, served to evaluate the plausibility of the benefit measures. That is, the estimated expenditure adjustment attributed to air pollution changes could be compared to what the adjustments would be for price changes. At a minimum, the locational equilibrium model illustrated here provides another opportunity to develop these types of cross-checks by considering the effects of income, and partial versus general equilibrium adjustment.

The Epple and Sieg (1999) framework has the potential to meet these needs and to provide an assessment of the importance of one type of general equilibrium adjustment. The new approach allows unobserved individual heterogeneity in preferences for public goods as well as differences in household income to be: (a) incorporated in the estimation of household preferences for spatially delineated nonmarket goods; (b) used in the computation of the new locational equilibrium from large and locally differentiated changes in public goods; and (c) consistently represented in Hicksian welfare measures that account for the change in quality, the resulting equilibrium relocation, and the price changes that are required for the new equilibrium.

The policy background and context for our analysis of EPA's projections for the ozone improvements to take place in the Los Angeles area are described in the next section. Section 3 describes our extension of Sieg et al. [2002a]'s model to allow us to revisit EPA's Prospective Analysis. Section 4 outlines the specific details of our policy evaluation and the results, and the last section discusses their more general implications.

2. Background

2.1 Context

Section 812 of the 1990 CAAA requires periodic assessments of the benefits and costs of the regulations intended to reduce the concentrations of criteria air pollutants. The first

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[8] The Math-Tech (1983) analysis was in many respects much more sophisticated in its economic methods than current practice. Hedonic property and wage models considered the need to approximate the areas under marginal damage functions (see Vol. III, pp. 5-32 – 5-35 and pp. 6-34 – 6-37). The marginal values as well as the nature of the approximation varied with the level of the air pollutant in each location. Their analysis of household expenditure models remains one of the most sophisticated evaluations for policy. The data used allowed the model to take account of differences in environmental conditions, family size, income, prices, and air pollution at the county level (see pp. 7-83 – 7-85).

Prospective Assessment (U.S. EPA, 1999) estimated the annual human health benefits (in 1990 dollars) of Titles I through V of the CAAA to be $68 billion in 2000 and $108 billion in 2010.[9] These results imply an approximate per-household annual benefit of $648 and $1,028 in 2000 and 2010, respectively.[10]

The methodology used for these estimates begins with a detailed inventory of the baseline emissions for major point and mobile sources under different regulatory regimes. The projected emissions are introduced into comprehensive air diffusion models to produce estimates of the ambient concentration for each criteria pollutant included in the assessment.[11] For ozone, the air pollutant of primary interest here, the spatial resolution varies depending on the region where these models are applied. For urban areas in the western United States, the Urban Airshed Model's (UAM) spatial resolution is either 4- or 5-km grids. These analyses produce hourly ozone concentrations for specific time periods. As a result, EPA selected a set of design days (with weather conditions and other factors that might influence ambient pollution concentrations specified) to represent a time span most likely to be relevant to the regulatory standard. In the case of ozone, this set corresponds to conditions yielding the highest concentrations. In EPA's analysis, this ambient concentration was predicted, along with the relevant health effect concentration-response functions, using a spatial resolution matching the air quality monitoring. The predicted difference in the count of each type of health effect from each of the CR functions, comparing with- and without-CAAA ozone condition, is then "monetized" with constant unit values to develop benefit measures for pollution reductions.

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[9] The Titles for the CAAA are given as follows:

Title I – volatile organic compounds (VOC) and nitrogen oxide (NO_X) satisfy reasonably available control technology and reasonable further progress requirements for ozone non-attainment areas.

Title II – motor vehicle and non-road engine provisions are met.

Title III – the 2- and 4-year maximum achievable control technology standards are met.

Title IV – the SO_2 and NO_X emission programs for utilities are met as relevant for each region of the United States.

Title V – the permitting system for primary sources of air pollution is implemented as mandated.

[10] In 2000, the U.S. population was approximately 275 million. In 1993, the average household size was 2.62. This would imply approximately 105 million households, the basis for our per household estimate.

[11] The Prospective Analysis considered ground level concentrations of ozone, particulate matter 10 microns in size, particulate matter 2.5 microns in size, sulfur dioxide, nitrogen oxide, and carbon dioxide. It also considers acid deposition and visibility impacts as well as stratospheric ozone concentrations.

2.2 EPA's Prospective Assessment for Southern California

The EPA Prospective Assessment of how the CAAA would affect future air quality conditions identified five major source categories for air pollutants: industrial point sources, utilities, non-road engines/vehicles, motor vehicles, and area sources.[12] For each source, both a base-year level of emissions and projected growth of the specific pollution-generating activities were developed for 2000 and 2010 in the absence of CAAA requirements. These projections were modified to reflect control assumptions in each of two "future years." For volatile organic compounds (VOC) and nitrogen oxide (NO_X), both contributors to ambient ozone, the national assessment estimates a 27% reduction due to CAAA's effects on VOCs in 2000 and a 35% reduction in 2010. For NO_X, the reduction is comparable for 2000 (i.e., 26%) and a little larger in 2010 (39%).

Sectoral emission projections were disaggregated to the state/industry level or below.[13] Spatially differentiated emission estimates, at least at the county level, were then introduced into one of several air quality models. Ozone concentrations were estimated using the Urban Airshed Model (UAM) and the variable grid UAM (UAM-V). For the Los Angeles area, the EPA analysis supplemented the regional scale modeling results with higher resolution analysis. As a result, ambient concentrations on an hourly basis are available at a 4-km resolution for this area.

The UAM framework is a three-dimensional, photochemical grid model that simulates the physical and chemical processes underlying the ambient concentration of pollutants. Four steps are involved in the computation of these estimates for ambient concentrations: emissions are introduced; horizontal diffusion is computed; vertical diffusion and deposition are calculated; and, finally, chemical transformations are evaluated for reactive pollutants. In the relevant ozone models for this analysis, these computations are undertaken every few minutes.

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[12] EPA's definition for the source categories are given as follows:

Industrial Point Sources – boilers, cement kilns, process heaters, turbines

Utilities – electricity producing utilities

Non-road Engines/Vehicles – air craft, construction equipment, lawn and garden equipment, locomotives, and marine engines

Motor Vehicles – buses, cars, trucks

Area Sources – agricultural tiling, dry cleaners, open burning, wildfires

[13] Appendix A of EPA [1999] provides a detailed description of the models used in the sectoral emission analysis.

The EPA air quality modeling for the Los Angeles area also relied on input data from the South Coast Air Quality Management District (SCAQMD) data.[14] The EPA projections with and without CAAA considered the conditions for two three-day periods (June 23-25, 1987 and August 26-28, 1987) with baseline conditions augmented by the emission profiles associated with each scenario. Table 1 summarizes the VOC and NO_X emission totals for each scenario in Los Angeles. With the assistance of EPA staff and its contractors, the three analyses undertaken for California were interpolated to a consistent 5-km by 5-km grid cell pattern for the study area included in our analysis.[15] For each scenario, the latitude and longitude of the centroid for all cells in California were computed. Hourly ozone values were summarized for each grid cell with 10 hourly ozone values (measured in parts per billion) as the 5th percentile through the 95th percentile of all the modeled hours in the six-day simulation period.

3. The Locational Equilibrium Model

Our policy evaluation uses estimates of the preference parameters for housing and public goods based on applying the Epple-Sieg (1999) locational equilibrium estimator to 92 school districts in the Los Angeles metropolitan area. A detailed overview of the model, as well as the specific features of the data and estimates, are described in Sieg et al. (2002a). In this section, we provide a brief summary of the structure of the model, focusing on how it can be used in policy analysis to offer the first benefit transfer.

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[14] The SCAQMD also was the source for the monitoring data used to estimate our economic model, discussed below in section three.

[15] James DeMocker of EPA's Office of Policy Analysis and Review arranged for these data to be developed. Kenneth Davidson of Abt Associates, Inc., developed the specific decile distributions geo-coded by latitude and longitude form the three California analyses. These were:

San Francisco – August 3-6, 1990: a 4km x 4km urban scale analysis covering San Francisco Bay area, Monterey Bay, Sacramento, and a portion of the San Josquin Valley.

Los Angeles – June 23-25, 1987 and August 26-28, 1997: a 5km x 5km grid covering the South Coast Air Basin from Los Angeles to beyond Riverside, including part of the Mojave Desert.

Rest of California – July 1-10, 1990: a 56km x 56km (regional scale) covering the 11 western-most states.

See private correspondence Kenneth Davidson, December 17, 2001.

Most of our analysis is focused on the South Coast Air Basin, but this additional analysis was required to assure the outermost areas of the counties included in our locational equilibrium analysis could be included in the evaluation of these policies.

3.1 Context

Three aspects of the locational equilibrium model are especially noteworthy for applications to non-market valuation. First, its use of the single crossing property relaxes the Willig (1978) condition that many revealed-preference methods have required, along with weak complementarity, to recover Hicksian willingness to pay for improvements in non-market environmental amenities (see Bockstael and McConnell, 1993). Second, it readily accommodates the spatially differentiated environmental conditions usually associated with externalities. The model estimates preference parameters are based on observed measures of public goods and housing prices for each community and the observed sorting of individuals among communities. Third, the same necessary conditions for a locational equilibrium used to estimate the preference parameters allow the framework to be used "in reverse." That is, with estimates for household preferences, including the parameters describing the heterogeneity, it is possible to use the model to compute the prices required for any new locational equilibrium due to an exogenous change in the vector of spatially delineated public goods. It is this last feature that allows the model to be used for a general equilibrium benefit transfer. It is possible to estimate the baseline market conditions hypothesized to characterize a without-CAAA scenario and to consider a change from that baseline to evaluate the "with CAAA" case.

3.2 Structure of the Model

The locational equilibrium model is a mixed discrete/continuous framework that maintains households' decisions can be described as involving two stages—the selection of a best community and then, conditional on that choice, an optimal demand for the community-specific good. In our case, this community-specific good is housing. Equation (3.1) describes the first stage of this hypothesized choice process with m_i the *i*th household's income; p_j, the *j*th community's price for housing; $g(q_j,a_j)$, a separable sub-function providing an index, θ_j, of the local public goods, q_j and air quality, a_j; and α_i an unobserved taste parameter.[16] By treating α_i as a random variable, jointly distributed with income, the model allows for unobserved heterogeneity in household preferences for public goods.

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[16] In the estimation of the model, $g(\cdot)$ is assumed to be linear with the coefficient of q_j, normalized to unity. Thus, $\theta_j = q_j + \gamma a_j$.

$$j_i^* = \underset{j \in A}{arg\,max} \left\{ V\left(\alpha_i, m_i, g(q_j, a_j), p_j\right)\right\} \quad \forall_i \tag{3.1}$$

where A = set of communities. The optimal housing demands for the ith household, h_i, conditional on j_i^* (the optimal community for household i), is given in equation (3.2).

$$h_i = -\frac{V_{p_{j*}}}{V_{m_i}} \quad \forall_i \tag{3.2}$$

The locational equilibrium requires a set of prices such that market demand for housing equals supply in each community. We omit the subscript i in what follows because households are identified through the joint distribution for α and m, $f(\alpha, m)$. Using this continuous formulation, the market equilibrium is defined in equation (3.3).

$$S_j(p_j) = \int_{c_j} h\ (\cdot) f(\alpha, m) d\alpha\, dm \quad \forall_j \tag{3.3}$$

where c_j = set of all households such that: $j_i^* = j$ and $S_j(p_j)$ = supply of housing in community j.

A locational equilibrium, with no two communities having the same housing prices, and preferences satisfying the single crossing condition, implies a sorting of households as well as an ordering of the public goods index ($\theta = g(.)$) and the housing prices across communities that satisfy three conditions: boundary indifference, stratification, and ascending bundles.[17]

Following Epple and Sieg (1999), we use a preference specification consistent with a constant income and price elasticities of demand for housing, and with separability between housing and community specific public goods. Equation (3.4) provides the specific form that is consistent with the single crossing condition (dropping the ith subscript for simplicity).

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[17] These conditions are described in detail in Epple and Sieg (1999). Boundary indifference refers to the existence of households that are indifferent between two communities that have been ranked to adjacent positions based on their prices and public goods. Stratification implies that households are distributed across communities with an ordering by income and the taste parameters for public goods. Ascending bundles implies the level of the index of public goods, housing prices (for a homogenous housing unit), and the highest income for each type of taste for public goods all increase in the same order.

$$V(\alpha, m, \theta_j, p_j) = \left[\alpha \cdot \theta_j^{\rho} + (exp((m^{1-v} - 1)/(1-v)) \cdot exp((1 - Bp_j^{\eta+1})/(1+\eta)))^{\rho}\right]^{1/\rho} \quad (3.4)$$

3.3 Welfare Measurement

The locational equilibrium model implicitly bundles two goods—the community, which conveys the index of public goods, and homogeneous housing. The latter is assumed to be equivalent across communities. This feature is central to the development of the price indexes for the model (see Sieg et al., 2002b, for further discussion). The public good index is a weak complement to the community in that the individual does not gain from improvements in public goods in other communities. There is, however, no added "consumption of the community" with increases in the public goods. That is, housing demand conditional on community selection is not affected by the index of public goods. Thus, the Willig condition is not needed to limit the role of income effects (see Smith and Banzhaf, 2002).[18]

The potential significance of relaxing the Willig condition can be gauged using the effect of a change in income on the incremental willingness to pay for a specified change in the public good. Equation (3.5) defines the general equilibrium willingness to pay (WTP_{GE}). We can gauge the importance of relaxing these restrictions by considering the responsiveness of the WTP_{GE} to income, labeled as WTP_m in Table 2 below. Equation (3.6) demonstrates that WTP_m is simply a measure of how the policy affects the marginal utility of income.

$$V(\alpha, m - WTP_{GE}, \theta_k^*, p_k^*) = V(\alpha, m, \theta_j, p_j) \quad (3.5)$$

$$WTP_m = \frac{\partial WTP_{GE}}{\partial m} = \frac{V_m(\alpha, m - WTP_{GE}, \theta_k^*, p_k^*) - V_m(\alpha, m, \theta_j, p_j)}{V_m(\alpha, m - WTP_{GE}, \theta_k^*, p_k^*)} \quad (3.6)$$

In order to implement the prospective policy analysis, we need to estimate the model and solve for the locational equilibrium under the various scenarios developed for EPA's Prospective Analysis. In Sieg et al. (2002a) we discuss in detail how to estimate the parameters of this model using available data on housing transactions, public goods, and community-specific amenities.

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[18] The model resolves the issues associated with recovering Hicksian welfare measures for quality changes because the necessary conditions for equilibrium provide sufficient information to estimate a price index for quality. A utility-consistent price index that defines when communities are equivalent, or, in other words adjusts for quality differences between them, is defined from the boundary indifference condition.

This paper uses the Sieg et al. (2002a) parameter estimates along with air quality projections from the Prospective Analysis that are described in Section 2. These air quality measures are adapted to match the spatial detail of an expanded version of Sieg et al. (2002a) model to develop a consistent general equilibrium (GE) baseline set of conditions and GE responses to the policies considered in the Prospective Analysis.

3.4 Using the Los Angeles Metropolitan Area Model to Calibrate Market Conditions for a General Equilibrium Benefit Transfer

The logic of using estimates of the parameters describing the joint distribution of income and the taste parameter (α) for public goods to compute locational equilibria was developed in Sieg et al. (2002a). We confine our discussion here to three new issues. First, we consider how using the model as a description of market equilibrium with a predefined, inelastic supply of housing in each of the original 92 school districts performs when reproducing in-sample and out-of-sample prices for standardized units of housing in each community.

Second, we outline the process used to replace one school district (L.A. Unified) with 11 new school districts. These new school districts were introduced recently by the L.A. Board of Education to decentralize decision-making for local education. This reduction in the size of the largest "community" in our model provides an opportunity to expand the choice set used to compute the locational equilibria for the Prospective Analysis. Finally, we apply our model to calibrate the household allocations to match projected 2000 and 2010 conditions without the more stringent emission restrictions in EPA's policy evaluation. This process defines the baseline housing prices and expenditures used to evaluate the added increment in regulations due to the CAAA that was considered in the Prospective Analysis. Without this recalibration it is not possible to establish a consistent GE baseline for the distributions of households and prices in 2000 and 2010. These "without CAAA" baseline conditions are not observed; they are hypothesized reference points that serve as the starting points for computing the adjustments associated with the increased restrictions for 2000 and 2010.

To evaluate how the Epple-Sieg framework would perform in benefit transfer we re-estimated the hedonic model for the 92 school districts for 1995 using the same specification for the price equation and constructed the normalized prices for each school district using the fixed effects. The proportionate change in price indexes between 1990 and 1995 could then be compared with two general equilibrium computations—one for 1990 and a second with modified ozone conditions so they corresponding to the 1995 concentrations.

As one would hope, the proportionate changes from the hedonics and the GE computations are positively correlated with a statistically significant correlation coefficient of 0.236 (p-value = 0.024).

The 1995 GE computations suggested price indexes that were somewhat larger than the hedonic estimates. Nonetheless, this comparison indicates a reasonably high level of consistency between the model's predictions and actual price changes. With the hedonic model, air quality is not the only change during this period and the only variables reflecting these effects are temporal fixed effects. Under these circumstances, a positive, significant correlation between proportionate price changes is supportive of the model. Larger price effects from the locational equilibrium model would suggest its price effects may best be considered an outer bound to the general equilibrium price effects. The findings also are consistent with the general economic intuition that the computational model does not include the transactions costs of moving.

The second modification to the model involved replacing the L.A. Unified School District with the 11 sub-districts established in April 2000 by the L.A. Board of Education. Introducing these new districts provided greater spatial resolution for the largest school district used with the original model. For example, ozone concentrations in 1995 varied from 70 to 121 parts per billion across the 11 new disaggregated districts. Thus, this spatial delineation allows the model to account for the substantial variation in the ozone concentrations in this area.

To compute the baseline prices for each district, we calculated the specific measure for ozone concentrations for 1990 and the education measure (i.e., the average math scores for the 1992-93 California Learning Assessment System Grade Level Performance Assessment test) for each new district, and used the model to compute the new normalized prices for 1990 as a result of these refined estimates for ozone and school quality. Initial housing supply was implicitly defined based on housing expenditures in 1990 reallocated to each of the new districts, assuming a constant 1990 baseline price for all of them in the pre-decentralization equilibrium.

The last step to adapting the model for analysis of EPA's prospective report was computing the 2000 and 2010 locational equilibria for the ozone levels corresponding to no new controls as a change from the 1990 levels. The prices and associated distributions of housing expenditures for each school district define the baseline conditions for the policy solutions with increased controls. Figures 1 and 2 plot the 1990 price and index of public goods, θ (labeled as "theta" in Figures 1 and 2) along with 2000 and 2010 baseline and control solutions for comparison. As the figures suggest and we discuss in detail below, the improved ozone conditions have a marked effect on the new equilibrium prices, improving, in all cases, the public good/price combinations available.

4. Reconsidering the Prospective Analysis

Our analysis had access to the results of three sets of simulations from the air diffusion models: (1) baseline runs for ambient ozone concentrations in 1990; (2) ambient concentrations in 2000 and 2010, when air pollution regulations are "frozen" at federal, state, and local controls corresponding to their 1990 levels of stringency and effectiveness; and (3) concentrations in 2000 and 2010, when federal, state, and local rules promulgated under the 1990 CAAA are implemented (the reduced emission levels presented in Table 1). Our labeling convention in Figures 1 and 2 for the price and public good index (labeled "theta") as well as in figures describing ambient ozone conditions for specific school districts identifies scenario and year (90 for 1990, 00 for 2000, and 10 for 2010). The "without CAAA" case is labeled "ba" for business-as-usual and the second "ct" for control in our graphs. All of these scenarios were developed by EPA staff to mimic the effects on the Los Angeles area.

Access to these data offers an unusual opportunity. It is possible to use the framework of the Los Angeles-area model (LAM) for a policy analysis with the ambient ozone concentration data developed for that assessment. This section describes three sets of welfare computations: (a) average general and partial equilibrium measures of Hicksian willingness to pay, homeowner rents, a relocation index (count), and the general equilibrium price and public good changes for the area as a whole; (b) average welfare measures for selected school districts; and (c) measures of the distribution of gains by income group identified through the LAM framework's ability to track income heterogeneity.

4.1 The Spatial Implications of the Air Pollution Policies

The EPA simulations provide distributions for ozone for each latitude and longitude in 1990, 2000, and 2010. Figure 3 indicates the geographic scale of these simulations in relation to the school districts, which provide the lowest level of spatial resolution for our model. The "dots" falling within the boundaries of each school district identify the number of projected ozone distributions from EPA's air diffusion analysis. Each is the centroid of a 5-km grid. Our estimated model relied on the average of the 30 highest hourly ozone readings for a year, for which we do not have an exact counterpart in the available data. As a result, we used the average across grids in a school district for the 0.95 deciles. To ensure consistency between the monitored and simulated readings, we constructed school district-specific scale adjustments based on the ratio of average monitored ozone readings in 1990 to the average of EPA projected concentrations for the 0.95 decile in 1990.[19]

Figure 3 labels three of the school districts in our revised choice set with identifying numbers. Figures 4 through 6 present three examples of the average simulated concentrations (before adjustment, in parts per billion of ozone). Each graph provides five empirical distribution functions, distinguished by year (1990, 2000, 2010) and air pollution control scenario (ba, ct). These cases illustrate the diversity of conditions across school districts. In some cases, these empirical distributions do not shift with either the year or control conditions. In others, there are pronounced differences between ba and ct, but they vary by year. For example, considering the 0.95 decile (the right-most point on each graph) in Figure 4 for the Santa Monica/Malibu Unified school district (ID = 102) along the coast (left side of Figure 3) of these highest readings, all are below the current 0.12 primary standard (parts per million). Moreover, the anticipated changes over time are modest, and closely aligned with the 1990 levels. Another coastal school district, Long Beach Unified (ID = 89) in Figure 5 displays an even more constant pattern of ambient ozone. By contrast, if we consider a district away from the coast, such as Claremont Unified (ID = 75) in Figure 6, the picture is quite different, with large changes in the ozone concentrations across scenarios. Examination of comparable graphs based on the Prospective Analysis scenarios for the other 99 school districts reinforces a general conclusion that is available from these three graphs. There is wide heterogeneity in the current and projected ozone concentrations as well as in the impact of the ba and ct implications for this set of school districts.

This point is implicitly made in a somewhat different format with Figures 3 and 4. These graphs plot the equilibrium prices and public good indexes for the without- and with-CAAA cases in 2000 and 2010, along with the initial baseline 1990 conditions for comparison. While the curves appear close, there are large differences in both of these normalized indexes across school districts and between the three sets of solutions. Movement from left to right of the schedule connecting the price/public-good index pairs in each graph indicates that, at each level of housing prices, a greater level of the composite public good is available. The increases in the public good index arise because the ozone concentrations are being reduced between the ba and

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[19] This process parallels EPA's practice in using their simulations (see EPA [1999], Appendix C, p. C-25).

ct cases for some, but not necessarily all, school districts. These changes are certainly not uniform.

When households are allowed to move in response to these exogenous changes in pollution, locational equilibria corresponding to the comparison of the ba and ct scenarios in each year will involve a different ordering of price and public good pairs. From the perspective of renters, the horizontal shift of the price/quality locus in the region of their initial location choice captures their welfare gain. It is the shift of the price/quality locus that determines their potential to realize a welfare gain from the policy, not necessarily the new price and air quality conditions of their initial location choice.

In the case of large improvement at one or a small number of locations, the impact on the location of the entire locus would be minimal. The improved locations would be reordered along the locus, but the locus itself would shift very little—leading to very little welfare gain for the renter households.[20] This is not the case for owner households, who would capture the impact of the quality improvement as it is capitalized into housing prices at their initial location.

In the large policy changes that we have analyzed, the close clustering of the observations between the with- and without-CAAA cases over the mid-range of values for the public good index, θ, indicates the relatively modest changes in ozone and limited impact on normalized prices for these communities. Nonetheless, even over this range the relative positions of the school districts' price and public good indexes change between the ba and ct solutions. Figure 7 reproduces Figure 2, dropping the 1990 solution and identifying the positions of a few of the school districts with and without the controls for the 2010 emission profiles. For example, the price/public good combination of school district 89 indicates an increased level of public good index but a lower normalized housing price, while that of 102 indicates an increase in both.

4.2 General and Partial Equilibrium Benefit Measures

Table 2 illustrates some of the benefit calculations that are possible with the model. We report estimates for each scenario for the area as a whole and a few selected school districts. Our policy comparison provides a baseline by estimating the locational equilibrium model under the

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[20] This problem resembles the issues posed by Palmquist (19920. To our knowledge, a tractable method for evaluating what is large versus small has not been offered. The ability to address it within the Epple-Sieg framework is another advantage of the model.

2000 and 2010 business-as-usual air pollution levels (ba00 and ba10). Next, for comparison, the 2000 and 2010 (ct00 and ct10) equilibria are estimated under the assumption that the 1990 CAAA are implemented. When moving to the new equilibria from the baseline, households can move and prices adjust to define the new locational equilibrium. Thus, our definition for WTP must take account of both potential adjustments.

As equation (3.6) suggested, a general equilibrium welfare measure recognizes that each household can experience both new prices and new levels of the composite public good as a result of both the CAAA regulations and their relocation to a new school district. In developing these solutions, we assume there are no transaction costs associated with these adjustments. We return to this point below. Equation (3.6) identifies the prospect for GE adjustments by allowing both p and θ to change (with θ^*, p^* designating after ozone change values). Relocation and the policy change imply that households with an initial location can realize change due to either or both influences. As a result, there is the potential for different community subscripts for θ^* and p^* versus θ and p on the left and right sides of our definition equation (3.6).

The computation of our locational equilibrium draws 1 million pairs of α and m from the estimated joint distribution. Our measures of housing expenditures in each school district serve to define the available housing supply. Each benefit measure reported in Table 2 varies with household. They are averages computed for the households (i.e., pairs of α and m) initially assigned to each school district. More formally, the average WTP_{GE}^{j} for school district j is defined in equation (4.1).

$$WTP_{GE}^{j} = \int_{c_j^0} WTP_{GE}(m,\alpha) f(\alpha,m)\, d\alpha dm / p(c_j^0) \qquad (4.1)$$

where c_j^0 is the baseline community designation and $p(c_j^0)$ is a measure of number of households in c_j^0. We also compute the average partial WTP defined by allowing only the public good index to vary in (4.2). Households' adjustments are overlooked.

$$V(\alpha, m - WTP_{PE}, \theta_j^*, p_j) = V(\alpha, m, \theta_j, p_j) \qquad (4.2)$$

Table 2 describes both the implications of the policy for ozone concentrations, how these are evaluated by the model, and a variety of welfare measures. The first column reports the average proportionate reduction in ozone implied by the with- and without-CAAA scenarios for the initially assigned school district *(Δ ozone)*. $\Delta\theta$, in the third column, is the proportionate change in the public good experienced by the household (after adjustment) identified by their initial school district. *Δp*, in the second column, is the average proportionate change in prices. The variable labeled "count" measures the fraction of the initially assigned households who move from their initial school district. The remaining statistics correspond to the general equilibrium willingness to pay relative to income, the levels of the average values for general and partial equilibrium willingness to pay (as defined in equations (4.1) and (4.2), respectively) for different subsets of the population, *Δ Rent*, and the change in willingness to pay with income (WTP_m defined in equation (3.7)). *Δ Rent* (i.e., $(p^* - p_j) \cdot H_j$) offers an approximate gauge of the importance of the owner/renter distinction.

There are wide discrepancies between the overall average general equilibrium WTP (computed over all school districts) and the WTP_{GE} estimates averaged by the initial school district. Several of the school districts presented in Table 2 were selected to match the graphs for the changes in the ozone concentrations. The full range of average values for 2000, considering the WTP_{GE} averaged over all households in each school district, is from $53 (in the Anaheim Union High School district) to $1,694 for the Palos Verdes Peninsula Unified. The differences between partial and general equilibrium measures vary with the size and magnitude of the price changes, as we might expect. Average rent changes are less than 20% of the general equilibrium willingness to pay for the 2000 comparison, but they are larger for the projected effects of the CAAA controls in 2010. Finally, the implied responsiveness of *WTP* to income (WTP_m) is consistently greater than zero. In contrast, the Willig condition often used in non-market valuation requires that $WTP_m = 0$.

Our estimates for the per-household benefits in 2010 are generally smaller than for the ozone concentrations implied by regulatory effects in 2000 if we exclude changes in rents. Comparing the WTP_{GE} plus the *Δ Rent* for the two years implies that 2010 would have larger gains for homeowners than 2000. Neither our study nor the EPA analysis allows for real income growth at the household level to affect the benefit measures.[21] Thus, our findings for 2010 arise solely from the implied public good and general equilibrium price changes induced by the spatially delineated pattern of ozone concentrations for the business-as-usual and control scenarios.

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[21] Thus, there is a subtle contrast between the assumptions used to compute emissions, which assume growth, and to compute benefits, which do not. This difference implies growth in the level of economic activity does not affect real income.

4.3 Distributional Effects of Policy

Table 3 summarizes the *WTP* estimates comparing without- and with-CAAA in 2000 and 2010, averaged (across school districts) by income quantile. As with our other summaries, results for 2010 generate lower benefits to all groups and higher rents than those in 2000. On average, the highest income group realizes twice as much in willingness to pay as the lowest quantile. This comparison holds regardless of whether we consider the general or the partial equilibrium measures. In some respects, it understates the heterogeneity in gains by income group and location. Figures 8 graphs the average general equilibrium measures of *WTP* by school district (ranked by income) for the without- and with-CAAA comparison in 2000. The distribution measure labeled A reports the average value for households falling in the 0-to-25% interval of the WTP distributions by initial school district. B, C, and D correspond to 25 to 50, 50 to 75, and top intervals for the general equilibrium willingness to pay.

The pattern displayed in Table 3 also is apparent in Figure 8, in that higher benefits are experienced with each increase in income, as theory would suggest. However, the range of gains is much wider than what is implied by the overall means, with the lowest community and lowest quantile gaining about $33 in 2000 and the second highest school district and quantile at $2,317. Thus, heterogeneity in income, taste for public goods, and spatial differences in the implications of environmental policy induce a diverse pattern of gains that would be overlooked with area-wide averages, even when they are disaggregated by income quantile.

5. Implications

The locational equilibrium framework provides a consistent basis for computing benefit measures for large and diverse changes in environmental conditions. The model identifies how diverse households (in terms of income and tastes for public goods) adjust to "the environmental cards they are dealt" by the combination of policy and nature. We have demonstrated it is possible to develop models that allow general equilibrium benefit transfer; the model matches the spatial detail used to describe variations in environmental conditions that characterize most large-scale policy analyses and offers a consistent basis for describing how households would adjust to such diverse conditions. As we noted at the outset, current practice uses both the damage function and unit benefit estimates in ways that assume households do nothing different as a result of the changes to environmental conditions. This assumption is not only unrealistic, but also inconsistent with the revealed preference methods for benefit measurement. Large policy interventions will likely generate different types of adjustments from those "embedded" in the reduced-form ecologic or prospective cohort models used to describe health effects. Economic values change as well. The challenge is in consistently representing the market consequences of these adaptations.

The locational equilibrium model relies on heterogeneity in tastes, the levels of public goods, and income to estimate preferences. When used to consider air pollution, we must assume that people recognize local air quality conditions, appreciate the consequences of that pollution, and use this information in their decisions about where to live. Our application may well offer the best test case for this set of maintained assumptions. The *Los Angeles Times* publishes spatially differentiated information about air pollution, including ozone, and its real-estate guide routinely identifies air quality conditions in different housing areas. This information also is a part of local TV news reports. Thus, while ozone itself may not be visible, a spatially differentiated information base is available for the area as a whole.

Does this imply that the preference function estimated using a locational equilibrium fully captures all the health effects conventionally enumerated in damage functions along with the aesthetic effects? There is no way to directly answer this question until we are able to introduce health effects as a set of structural restrictions to preference functions (e.g., as household production functions for health). Such an approach would imply that the damage functions affect the consumption choices of private goods (including leisure) in a way that could be identified. At this stage, our estimated preference function allows air pollution to affect choices. The specification does not “explain” why.

By contrast, the damage function approach measures a physical effect hypothesized to be related to pollution concentrations in areas where these health effects are observed. This strategy is limited in a different way. Epidemiological research isolates the health effects, but it cannot “prove” that households recognize them. Equally important, these physical effects are converted to a WTP estimate with unit benefit from avoiding each type of health effect. This does not mean people would pay this amount. Households at all income levels are assumed capable of paying the same unit benefits. Payments to avoid one type of damage are assumed not to affect what could be paid for another type (see Smith et al., 2002, for a discussion of this issue).

One further qualification to our formulation of the locational equilibrium merits discussion. We assume adjustment is costless. This is certainly not the case. It seems reasonable to expect that this simplification would overstate the importance of general equilibrium effects. With household adjustment costs, responses would be smaller. There are other feedback effects that we omitted. With higher costs of production, prices of other goods should change and pollution-related consumption choices for households may change as well. For now, we have taken a step toward providing bounds on the importance of adjustment. This is the problem originally posed by Lind (1973) in the literature on spatial externalities (both negative and positive) and ours is the first attempt, to our knowledge, to address it in a practical policy context.

At a minimum, the current generation of locational equilibrium models has the ability to provide more detailed cross-checks for the conventional benefit estimates. They also allow consistent measurement of partial and general equilibrium benefit measures, and therefore offer a simple gauge of the likely impact of adjustment for benefit transfers. Finally, because the model relies on heterogeneity, it also allows consistent measures of the distribution in gains (or losses) from policy that can be related to the observable sources of that heterogeneity.

Our application illustrated that these differences can be substantial. The estimated annual benefits for different groups, as a result of the environmental conditions experienced, their income, and their tastes for public goods, can vary by a multiple of more than 50—from about $30 to more than $2,000 (in 1990 dollars) for the ozone changes—due to continued implementation of the CAAA mandates.

An important challenge that remains is to determine how these estimates of heterogeneous gains, derived by tracking distributional effects, can be used as part of an informative assessment of the equity effects of environmental policies.

Table 1. EPA Projected VOC and NO_X Emissions for Los Angeles (tons per day)

Source/ Pollutant	Base 1990	Without CAAA 2000	 2010	With CAAA 2000	 2010
VOC					
Area	758	770	871	607	700
On-Road Mobile	1,179	999	1,168	410	213
Point					
Low Level	197	196	196	196	158
Elevated	1	3	2	3	2
Total	2,135	1,968	2,237	1,216	1,073
NO_X					
Area	450	467	529	453	463
On-Road Mobile	993	1,280	1,573	879	626
Point					
Low Level	216	186	186	139	139
Elevated	19	19	12	18	8
Total	1,678	1,953	2,300	1,489	1,236

Source: U.S. Environmental Protection Agency (1999). These estimates were taken from Table C-4.

Table 2. Alternative Benefit Estimates With and Without CAAA for 2000 and 2010

	Δozone	Δp	Δθ	count	WTP_{GE}/m	WTP_{GE}	WTP_{PE}	ΔRent	WTP_m
2000									
area-wide	-0.128	-0.002	0.014	0.401	0.023	910	817	94	0.102
Upland Unified (52)	-0.204	0.015	0.021	0.164	0.016	617	541	111	0.076
Claremont Unified (75)	-0.233	-0.005	0.013	1.000	0.018	676	569	109	0.078
La Canada Unified (88)	-0.184	0.003	0.015	0.916	0.035	1434	488	-217	0.154
Long Beach Unified (89)	-0.019	-0.016	0.009	1.000	0.028	1084	676	126	0.116
Santa Monica/Malibu Unified (102)	-0.106	-0.012	0.009	0.254	0.040	1629	1025	606	0.165
"Area B" new LA Unified Sub-district (201)	-0.208	0.015	0.017	0.142	0.027	1076	1006	137	0.123
2010									
area-wide	-0.107	-0.001	0.013	0.274	0.022	846	679	180	0.095
Upland Unified (52)	-0.180	-0.027	0.003	0.020	0.017	651	711	-122	0.071
Claremont Unified (75)	-0.204	-0.015	0.008	1.000	0.009	651	332	323	0.073
La Canada Unified (88)	-0.205	0.022	0.019	1.000	0.028	1135	2005	-901	0.132
Long Beach Unified (89)	-0.009	-0.023	0.006	0.000	0.027	1070	387	710	0.113
Santa Monica/Malibu Unified (102)	-0.187	0.008	0.016	1.000	0.033	1383	1839	-472	0.150
"Area B" new LA Unified Sub-district (201)	-0.181	-0.003	0.011	0.000	0.010	1075	380	784	0.118

Table 3. Willingness to Pay by Income Quantile

Policy Scenario/Quantile	m	WTP_{GE}	WTP_{PE}	ΔRent
2000				
First Quantile	22,478	604	543	62
Second Quantile	33,400	793	712	82
Third Quantile	41,675	961	863	100
Fourth Quantile	58,216	1,282	1,151	133
2010				
First Quantile	24,443	560	451	118
Second Quantile	33,376	736	593	154
Third Quantile	41,611	891	717	187
Fourth Quantile	58,101	1,189	956	249

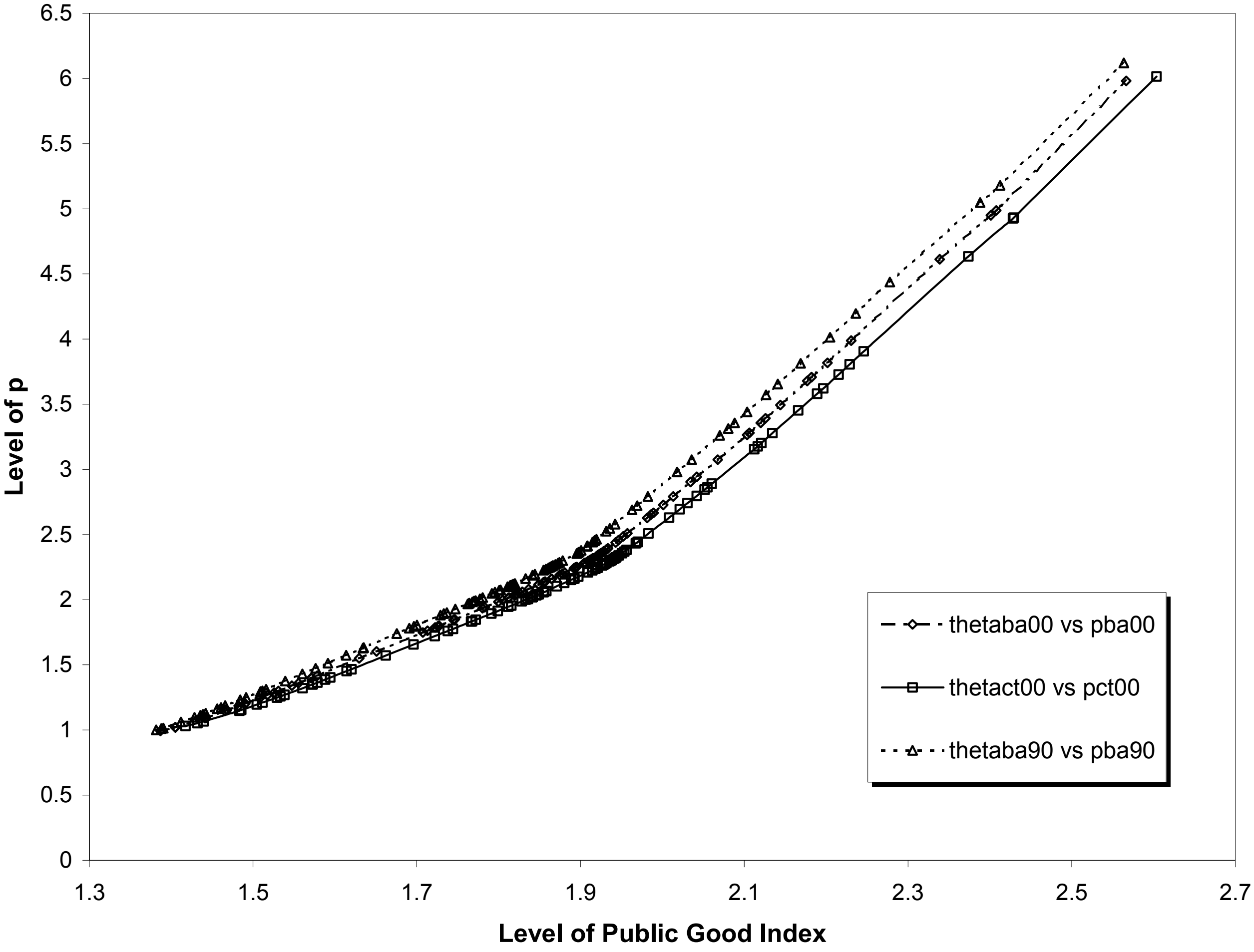

Figure 1. Computed General Equilibrium Prices for With and Without CAAA 2000 Compared to Baseline 1990[*]

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[*] The legend follows the format described in the text. "Theta" refers to the value for the index of public goods (θ). p is the price of normalized housing, "ba" refers to the "without CAAA", and "ct" the "with CAAA." 90 refers to 1990, 00 to 2000, and 10 to 2010.

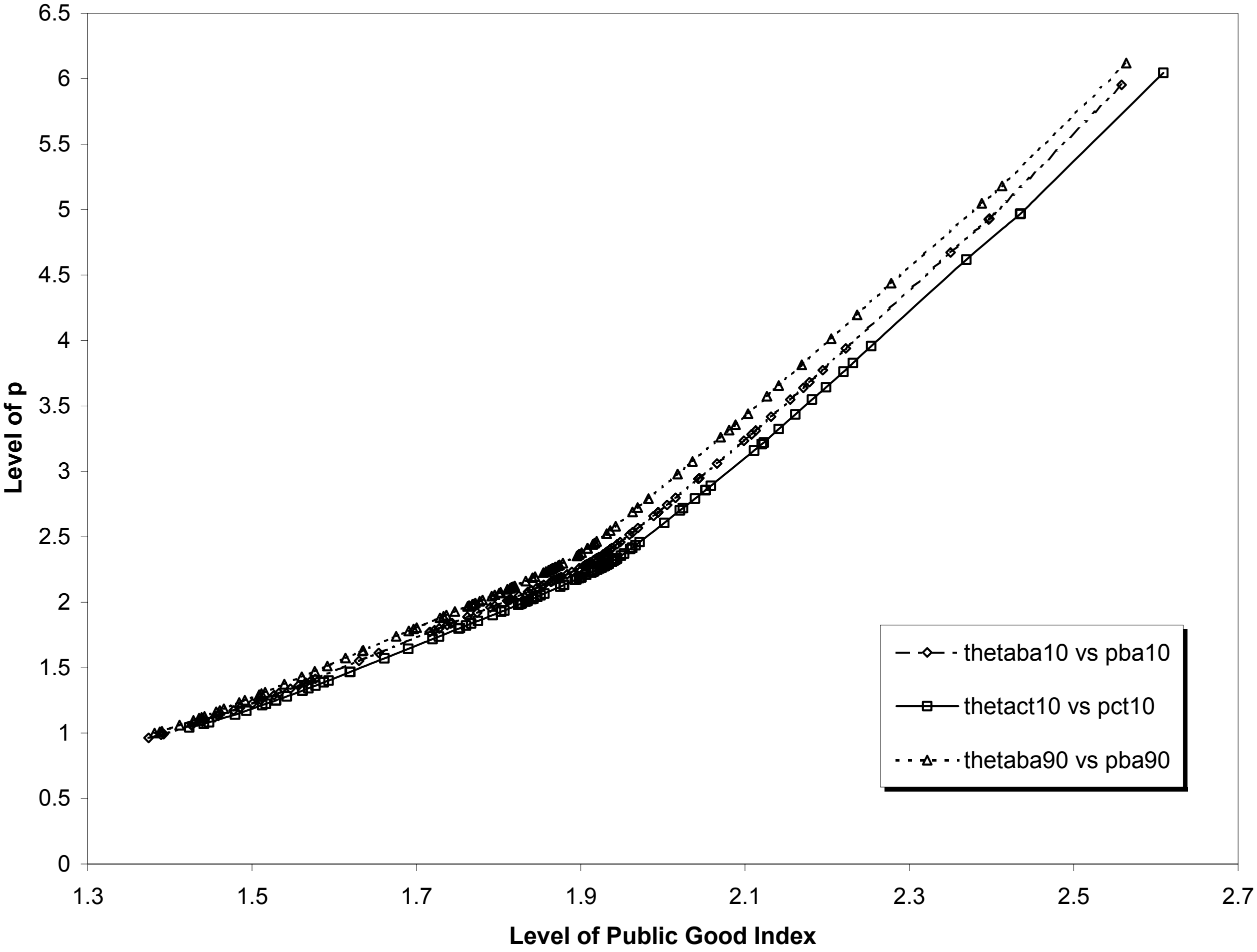

Figure 2. Computed General Equilibrium Prices for With and Without CAAA 2010 Compared to Baseline 1990[*]

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[*] See Figure 3 for explanation of the legend.

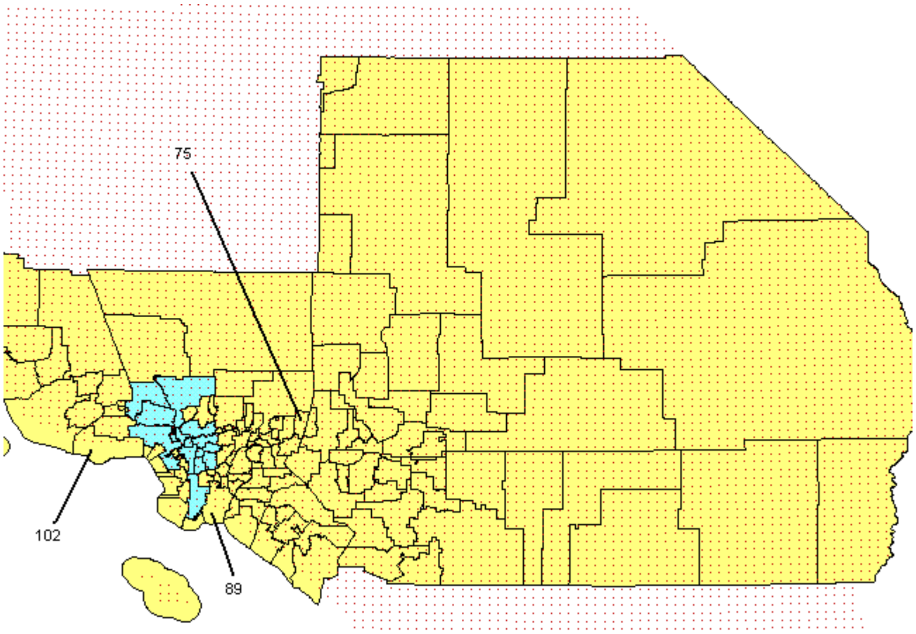

Figure 3: LAM School Districts and EPA UAM Locations for Estimated Ozone Readings

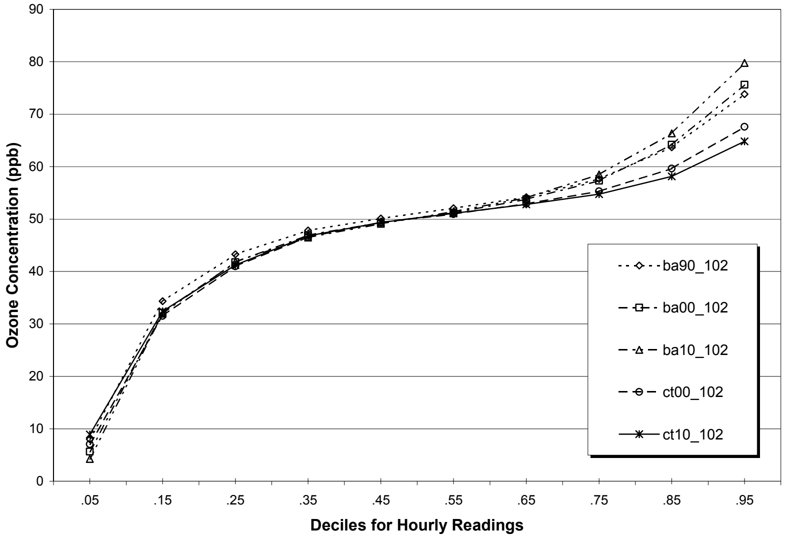

Figure 4: A Comparison of Projected Ozone Concentrations for Santa Monica-Malibu Unified School District (102)

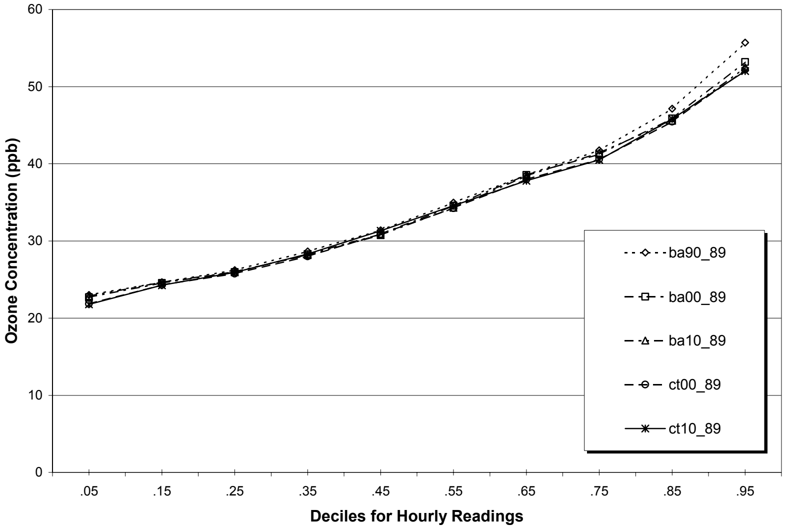

Figure 5. A Comparison of Projected Ozone Concentrations With and Without CAAA for Long Beach Unified School District (89)

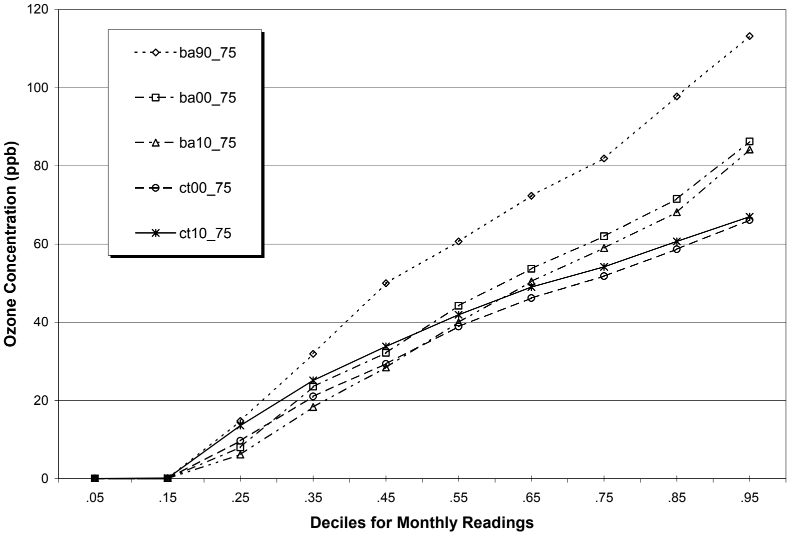

Figure 6: A Comparison of Projected Ozone Concentrations With and Without CAAA for Claremont Unified School District (75)

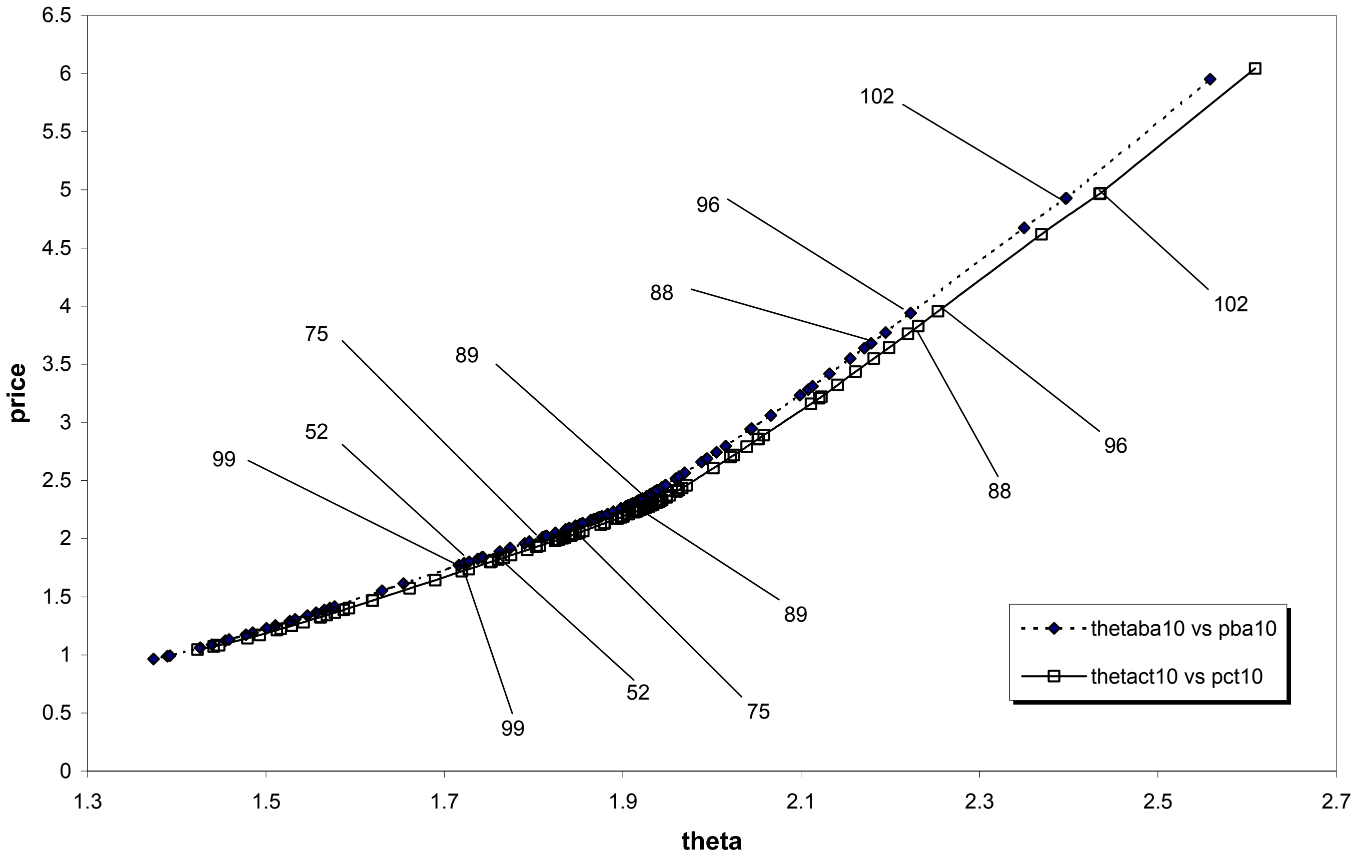

Figure 7: Change in Ordering of Price/Public Good Pairs for 2010

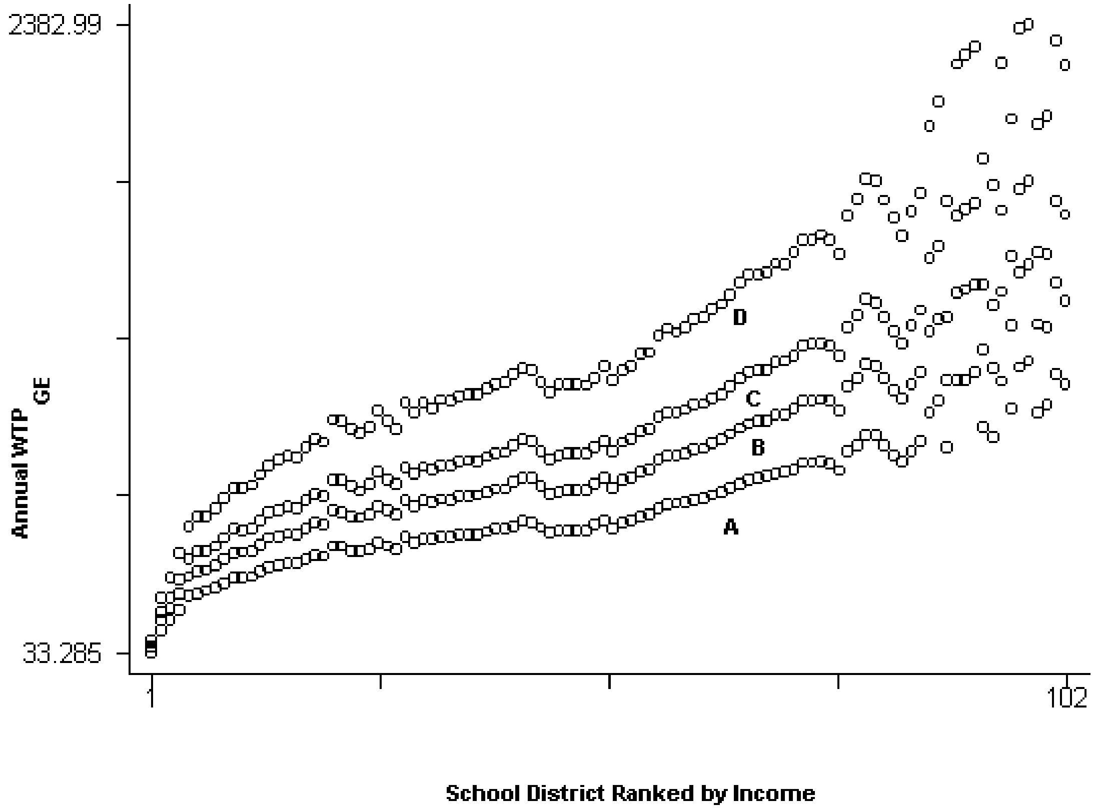

Figure 8: Average Willingness to Pay by Income Quantile in Each School District

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