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Estimation of the Return on Capital in Municipal Water Systems

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Abstract: The shadow return on capital in 75 small municipal water systems is estimated using a gamma frontier variable cost function. The estimated Social Cost of capital exceeds the shadow return by an average ratio of 4.37:1, with a median capital stock inefficiency of $70, 500 per year per system owing to overinvestment in public water supply capital. In addition, actual production costs exceed minimum costs by 36 percent, with a median inefficiency of $24,300 per system. Combining both types of inefficiency and extrapolating to the seven thousand comparable systems nationwide suggests economic waste of more than $663 million per year.

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Introduction

A furious debate began when David Aschauer (1989) published estimates of significant positive external effects resulting from public infrastructure investments, such as roads and water systems. Using a Cobb-Douglas production function fitted to state-level data, he estimated that a 10 percent increase in public infrastructure capital would increase private sector productivity by 4 percent. His reasoning has come to be known as the public capital hypothesis, according to which private sector firms enjoy considerable spillover benefits from public capital investments, which in turn foster rapid economic growth. He hypothesized that the slowdown in productivity growth then receiving so much attention was partly attributable to under investment in public capital. Subsequent research has challenged these claims as the product of a poorly specified econometric model that omits spatial and temporal fixed effects. Once these are controlled for, there appears to be little marginal effect of public capital on private sector productivity [Holtz-Eakin (1994)]. The more recent literature is reviewed by Batina (2001), and he concludes that the prevailing opinion today is that there is likely some positive external effect, but its magnitude is uncertain. Largely neglected in this large body of research is the equally important question of the direct return from the services provided by infrastructure investments.

It is relatively simple to evaluate the direct return on public water supply capital because drinking water has most of the characteristics of a private good, since it is both rival and excludable. Moreover, few external costs or benefits likely attach to the production and consumption of potable water *at the margin* in a developed country such as the United States. In poor countries where unhealthy water is a major cause of disease, this would not be the case. For all intents, the American drinking water market is characterized by public provision of a private good, usually by a monopoly not-for-profit municipal water undertaking. Although the incidence of un metered "free" water is increasingly rare, it is unlikely that prices reflect social marginal cost. Nevertheless, it is possible to estimate the shadow rate of return on public water capital by means of a variable cost function. The envelope theorem allows one to infer the implicit return on water works capital, which can then be compared with an estimate of the social opportunity cost of capital in order to determine if society is over or under investing–which is the question originally raised by Aschauer. Because the envelope theorem *assumes* the enterprise is minimizing costs, a stochastic frontier variable cost function is fitted in order to determine the actual degree of cost efficiency. Actual costs are then adjusted to their theoretical minimum, from which the shadow return may be inferred.

The Model

Apart from the need to adjust for any departures from cost minimization in order to estimate shadow returns, the public ought to be concerned with the efficiency with which the water department converts inputs into the output of potable drinking water because waste translates into higher than necessary costs, which must be borne as higher water prices, higher taxes, reduced municipal spending on other activities, or a need for greater grants and subsidies from the state or federal government. The water department's *Expenditures* E can be represented as follows:

$$E = \sum w\,x = VC(\,Y, Q,\ w \mid K\,; \beta)e^{\,v+u} \qquad (1)$$

or,

$$\ln E = \ln VC\,(\,\bullet\,) + v + u \qquad (2).$$

Where the following definitions are used (e is the base of the natural logarithm):

E = annual expenditure on hiring variable inputs.
w = a vector of input prices (e.g., wage rates and electricity price).
x = a vector of input quantities (e.g., workers and kilowatt hours of electricity).
$VC(\bullet)$ = variable cost function.
Y = water system production (e.g., average daily production in gallons).
Q = water quality.
K = the quasi-fixed input, such as water system capital stock.
β = a vector of cost function parameters to be estimated.
$v + u = \epsilon$, the composed error consisting of random noise v plus cost inefficiency u.

Thus the model contains a traditional variable cost function representing the *least cost* of producing varying levels of output (Y, Q), given input prices, the quantity of the fixed input (K) and available technology (summarized in the betas). Owing to the durability of water system capital, and natural monopoly as well as legal entry barriers, it seems appropriate to test if water systems are operating on their least cost curves, either short or long-run, rather than *assume* cost minimization. Observed expenditure thus consists of a least cost kernel and an appended two part error term to capture ordinary statistical noise plus departures from least cost behavior.

Least cost requires that water system decision makers operate on the relevant production isoquant and at the appropriate point thereon, so as to equate marginal rates of substitution with the relative prices of the variable inputs. In addition, the fixed amount of capital should also be the optimum amount so as to make the rental price of capital divided by the marginal product of capital equal to the common price-marginal product ratios of the variables inputs. Cost inefficiency may arise from overuse of inputs, i.e., operating above the relevant isoquant, or by using the wrong mix of variable inputs or, by using too large or small a stock of fixed inputs. Departures from the ideal of least cost may occur from uncontrollable random events such as weather (drought, floods, earthquakes, frost, etc.) plus systematic departures due to management failures, perhaps due to inadequate incentives that cause water system operators to choose a quiet life rather than single-mindedly optimize, civil service or union work rules, and politics. Random events are captured by the *v* error term, which is assumed to be a normally distributed variable with a zero mean. Fitting a traditional econometric cost function *assumes* each firm or decision making unit is a least cost operation, and any departures are randomly distributed. The stochastic frontier model employed here appends a non-random error term to the model that is intended to capture the systematic departures from cost efficiency. Apart from the usual data issues, estimation of equation (2) involves the choice of a functional form for VC(•) and for the distribution of the *u* inefficiency parameter.

Economic efficiency may be modeled using a cost frontier or a production frontier, the latter requiring estimation of a production function. A cost frontier is more attractive for the task at hand because policy makers are more interested in the dollars being wasted, if any. Moreover, a cost function readily admits multiple outputs, including quality measures, and it assumes the output and input prices are exogenous to the decision making unit. As a public utility providing water on demand to its customers, this is a plausible assumption. In a production function, the quantities of the inputs are assumed exogenous–which seems less plausible. And each water system is small enough to be a price taker in factor markets. Even if wages are union-negotiated, they are invariant with respect to the number of workers hired.

The translog and Cobb-Douglas functional forms have dominated the received literature. The translog is a flexible form, within which the Cobb-Douglas is nested as a special case. Its virtues include allowing returns to scale to vary with output and a non-homothetic expansion path, which implies a varying elasticity of substitution among inputs. Its flexibility comes at the price of requiring many more parameters to be estimated, most of which are collinear [Kumbhakar and Lovell, 144]. Efficient estimation of the translog really requires a system of equations, but that requires data about the quantities of inputs for use in the auxiliary equations, which is not available. Moreover, the appropriate assumption about how the error terms in the auxiliary input demand equations are related to the two-part error in the central cost equation is yet to be resolved satisfactorily (this is sometimes referred to as the “Greene Problem” (1980), after William H. Greene who first pointed it out). For these reasons the following analysis of water system cost efficiency employs the well-known Cobb-Douglas functional form. However, use of a single equation model means it is not possible to break apart the cost inefficiency into that attributed to being off the isoquant versus that due to being at the wrong place on the isoquant. But a variable cost function does have the considerable virtue of allowing one to infer if the quasi-fixed input is being over or underutilized. Misallocation of quasi-fixed inputs is characterized as another type of cost inefficiency in the literature [Kumbhakar and Lovell, 146]. Basically, the frontier inefficiency parameter u measures the extent to which a firm or agency is operating above the textbook (least cost) curve, while the shadow price method measures the inefficiency from being on the short-run instead of the long-run cost curve to produce observed output.[1]

As noted, stochastic frontier estimation requires some assumption about the probability density function of the inefficiency parameter u. Economic theory has little to say about what the frequency distribution of inefficient firms should look like. In a competitive market, inefficient firms should ultimately be driven out. On the other hand, inefficient operation may persist indefinitely with a publicly operated monopoly protected by high entry barriers resulting from legal impediments and significant economies of scale.

In their pioneering paper, Aigner, Lovell and Schmidt (1977) employed a half-normal distribution of *u*. In other words, the distribution of inefficient firms is characterized by the positive half of the normal curve. This was done because the half-normal is tractable and its properties well known. Since then several other distributional assumptions have entered the received literature, including the half normal truncated at a non-zero point, the exponential and the gamma distribution. The truncated half-normal is rarely used because its likelihood function frequently fails to converge. It can be readily tested against the half-normal, within which it is nested. Likewise the exponential is nested within the more flexible gamma distribution function. Greene (1990) proposed the gamma largely because it permits the shape of the distribution to itself be estimated rather than be predetermined. Estimation of all the received versions of frontier models is fully automated nowadays [Limdep8 (2002)].

Following Kumbhakar and Lovell (*op.cit*., 145), the shadow price of the quasi-fixed input K is given by : $\rho_K = -\partial VC/\partial K = -\beta_K (VC^* / K)$ (3).

Where ρ_K is the estimated shadow price of capital (the Appendix derives this result), β_K is the estimated coefficient on the capital stock variable from the frontier cost equation (2), and VC* is the estimated *least cost* of producing the observed level of output by utilizing the current fixed capital stock K. One obtains VC* by adjusting observed cost VC for the estimated inefficiency: $VC^* = (VC)e^{-u}$, with *u* being the estimated inefficiency parameter. Since *u* is estimated for each water system, it becomes possible to determine the extent to which each system has too large or too small a capital stock, relative to the long-run least cost level.

Data

The Office of the New York State Comptroller kindly collated 1999 cost information about water supply operations for 75 small villages from unpublished reports filed by municipal governments. All the systems serve a compact geographic area averaging only 1.4 square miles, and each is a separate water department. Thus, we are not comparing low density rural or suburban systems with high density urban systems.

Variable costs include employee compensation and contractual purchases, e.g., supplies and energy. Excluded are debt service and purchases of equipment and capital. Water system characteristics, e.g., output, production capacity, treatments used, violations, etc. are from the New York State Department of Health. The Census of Government contains information about water system employees and their wages. Electricity prices are from the state Public Service Commission.

The econometric cost frontier employs total variable costs as the dependent variable. Output is measured by the annualized average daily production of water leaving the facility. Monthly wages of water department employees and the kilowatt hour price of electricity are the two input prices. The presence or absence of a filtration plant is denoted by a dummy variable. Another dummy variable records if the water system was cited for a serious health violation during the period 1993-1999. The latter two variables function partly as water quality proxies, but also capture aspects of the production process. For example, five of the nine water systems with health violations do not disinfect the water (e.g., chlorinate). Because water systems vary in the number of supply sources used (e.g., the number of wells supplying a ground water system), a variable to capture this likely source variation in cost is included. Finally, the quasi-fixed capital stock is proxied by the maximum water production capacity of the system. This is analogous to comparing vary sizes of plants by their rated capacity. Table 1 presents descriptive statistics of the data used to estimate the variable cost frontier.

Although the data set is not a random sample, it does seem reasonably representative of the approximately seven thousand government owned community water systems in the U.S. serving populations between 501 and 3300 people, and relying primarily on groundwater sources [EPA (2002, vol. ii). Of the 75 systems in our data base, 64 fall into this population range, and 69 of them use only groundwater. Nationwide, the mean population of systems of this size and type is 1440 people, versus 1894 in this data set. Mean *total* expenses are $167,000 versus $156,725 for the group of 75. The national mean water production is 158,000 gallons (average daily amount) versus 273,371 in the sample set, but the latter is skewed by a few larger systems since the median production rate is 161,000 [EPA (2002) *ibid*.].

Table 1
Descriptive Statistics
(N = 75)

	Mean	Range
Variable Costs	$120,197†	$4561 - $685,424
Daily Average Water Production	273, 371 gallons	43,000 - 1,850,000 gallons
Maximum Production Capacity	661,123 gallons	80,000 - 5,000,000 gallons
Monthly Wages	$2,267	$1,042 - $3826
Electricity Prices (per KWH)	$0.09	$0.025 - $0.114
Filtration Plant (1 = yes, 0 = no)	11	-
Health Violations (1 = yes, 0 = no)	9	-
Number of Water Sources	2.5	1-7

† standard deviation = 135,829.

Costs and input prices are normalized by the price of electricity in order to impose linear homogeneity, a requisite for a well-behaved cost function. Prior to estimation, the software automatically tests if the composed error ϵ is properly skewed in conformity with the theory. If ϵ is normally distributed, $u = 0$ and only statistical noise v is present. In this case, the firms or operating units are fully efficient and no frontier is estimated.

The frontier cost function presented in Table 2 employs the gamma distribution of the inefficiency parameter *u* because that gives the lowest estimated inefficiency, with a mean of .31 (median .23).[2] In the case of the half-normal, mean *u* is .54, and for the exponential the mean is .37. The cost function coefficient estimates are virtually the same across the three alternate models (available from the author).[3] The more flexible gamma distribution apparently permits the mass of the inefficiency distribution to move closer to the zero point than do the alternate specifications, although the exponential gives quite similar results. The correlation coefficient across the three estimates of *u* is .995. Adopting the most favorable efficiency estimate as the benchmark result is part of the larger research strategy to adopt the most favorable assumption about water system performance whenever a choice presents itself. This tends to reinforce the credibility of any adverse conclusion.

Table 2
Gamma Distributed Variable Cost Frontier

Variable	*Coefficient*	*t-ratio*
Constant	-4.94	-4.86
ln(Average Daily Production)	1.11	7.08
ln(Wage/Electricity Price)	0.86	4.55
ln(Capacity)	-0.373	-2.90
Filtration	0.458	2.25
Health Violations	-0.671	-3.75
Number of Water Sources	0.098	1.78
Theta (θ)	2.383	2.71
P	0.745	1.28
Sigma v	0.379	4.59

Mean u = P/θ = .312, mean inefficiency = exp(u)-1 = .366. Log likelihood function: -55.29. Variances: Sigma-squared (v) = .144, Sigma-squared (u) = .131. N = 75.

Before turning to the estimated return on capital, let us consider the results in Table 2. Returns to Scale in a variable cost setting is given by [Braeutigam and Daughety (1983)]:

$$RTS = [1 - \partial(\ln VC)/\partial(\ln K)] \div [(\partial(\ln VC)/\partial(\ln Y)] = [1 - (-.373)] \div [1.115] = 1.23,$$

Thus there are significant economies of scale in water production, as expected. A filtration plant significantly increases variable costs, while the small number of systems with detected health violations spend significantly less on water production.[4]

Estimated Inefficiency

Producer specific inefficiency estimates in the case where u_i follows the gamma distribution is given by:

$$E[u_i \mid \epsilon_i] = h(P, \epsilon_i) / h(P\text{-}1, \epsilon_i) \qquad (5)$$

where ϵ_i is the composed error $v_i + u_i$ for the ith producer, and P is the estimated value of the shape parameter of the gamma distribution [Limdep8 (2002), E24-21]. Table 3 displays the distribution of the u_i inefficiencies. The normal - gamma model is consistently estimated by the method of simulated maximum likelihood [Greene (2003)]. The h( ) function has no operational closed form as it is the ratio of two extremely complex integrals and is estimated as the mean of a sample of random draws.[5]

Table 3

Estimated Water System Inefficiencies

u_i Range	Number of Water Systems
.087 - .317	50
.317 - .546	19
.546 - .776	3
> .776	3
	Total 75

Minimum .087, Maximum 2.38, Mean .31, Median .23

The results reported in Table 3 seem plausible, with only a small number of outliers.[6, 7] At the mean of u, least cost is $e^{-.31} = .73$ of actual cost, so these water systems could theoretically reduce their variable costs by about 27 percent. Alternately, actual variable costs are $e^{.31} - 1 = .36$ above minimum efficient costs.

In a study of the cost efficiency of 49 California municipalities, Grosskopf and Yaisawarng (1990) found that the minimum cost of police and fire services was 70 percent of observed costs, using a non-parametric mathematical programming technique (Data Envelopment Analysis).

Grosskopf and Hayes (1993) used a distance function methodology to test for cost minimization in the production of police and fire services among Illinois municipalities. They found average *technical* inefficiency (i.e., proportional overuse of labor and capital) of 10 percent. They also found that input proportions were not cost minimizing, but did not quantify the resulting percentage allocative inefficiency. And the estimated mean u = .31 reported in this paper for water systems is notably higher than the u = .16 reported for local highway departments in New York by Helfrich and Vitaliano (1998), who also employed a stochastic frontier cost function.

Adjusting the observed variable cost of each system using the estimated u_i yields a mean efficient cost of $84,700 (median $57,900), versus a mean of actual cost of more than $120,000 (median $82,200). A conservative estimate is that inefficient operation results in an average of $82,200 - $57,900 = $24,300 of economic waste per year, per system. If the national inventory of seven thousand public water systems of the size and type examined in this paper are comparably inefficient, the national economic waste is roughly $170 million per year.

The price of drinking water may be set above marginal costs to capture monopoly rent as a source of municipal revenue in lieu of property taxation (e.g., an indirect tax on universities or religious groups) or, prices may be set below marginal cost on distributional grounds since household spending on water is regressive with respect to income. Distorted drinking water prices deprive managers of a valuable guide for investment decision-making because revenues do not reflect consumer benefits. And input proportions, especially capital intensity, may be distorted by interest rate subsidies (e.g., tax exempt borrowing) or capital rationing because small municipalities are unable to have their bonds rated because of the small volume of potential debt to be issued. For example, the village of Victory was recently awarded a grant of $1.3 million and a $2 million zero-interest loan for 30 years to build a new water filtration plant, as part of $151 million in water works subsidies [*Times Union* (2003)]. The mayor of Victory is quoted as saying that the two thousand water customers in the village would have had to pay annual bills of $400 if the water board had been forced to borrow the money on the open market.[8]

Shadow Rate of Return on Capital

Proceeding from equation (3) above, the elasticity of variable cost with respect the fixed capital stock (capacity variable) is, from Table 2, $\beta_K = -.373$. Implementation of equation (3) also requires an estimate of the current value of the invested capital K, which is not available. However, the use of K merely converts the shadow return to a rate per dollar of invested capital.[9] Instead one can estimate the dollar amount of the return on water system capital and compare that with a dollar estimate of the social cost of capital invested in each water system. In other words, the shadow return is $-.373VC_i^*$ (i = 1 to75).

The Opportunity Cost of Capital

In order to estimate the Social Opportunity Cost of Capital, data about 32 proposed new water systems in comparable communities in Upstate New York is used to estimate the current replacement cost of each of the 75 water systems under investigation.[10] The mean new construction cost is $3.6 million and ranges from $2 million to $14.4 million.

Surveys of the real Social Discount Rate place it in the 2 to 8 percent range [Boardman, Greenberg, Vining and Weimer (2001), Ch. 10]. The U.S. Environmental Protection Agency uses a 7 percent real rate and a 20 year economic life when evaluating drinking water projects, with alternate rates of 3 and 10 percent [EPA (1999)]. Thus a lenient estimate of the Social Opportunity Cost of Capital is a 2 percent real rate and a 50 year economic life, which implies an annual annuity factor of .03. The resulting annualized Social Cost of Capital for each system has a mean value of $108,000 (median $90,700) and ranges from $60,800 to $433,500. Against this the mean shadow return is $31,500 (median $21,500), ranging from $1560 to $168,500. The median ratio of Social Cost to shadow return is 4.37:1 and varies from 1.45 to 51. The mean *difference* is $76,500 (median $70,500), and ranges from $33,700 to $270,000.

Although only a small number of water systems have been evaluated in this paper, the implications are more general. The Environmental Protection Agency (EPA) has estimated that New York State alone will have to spend $10 billion dollars over the next 20 years to ensure continued delivery of safe drinking water, with pre-applications by water works already amounting to $4.6 billion [New York State, EFC/DOH, (2000), 8]. As a matter of federal and state policy, resources to be used to comply with the Safe Drinking Water Act and to protect public health are made available at very favorable terms: interest rates may be subsidized down to zero, and grants are also made, based primarily on the median household income in the community. Typically, the loan/grant package is designed so that the annual charge for water doesn't exceed 1.75 percent of median household income (1 percent for low income communities). In other words, the relevant cost of capital is different for each potential borrower. Seven of the water systems analyzed in this paper received financing packages from the State Environmental Facilities Corporation in recent years for a variety of drinking water investments averaging $1.78 million each, all at *nominal* interest rates of 0 or 1 percent, and all but one for a period of 30 years. In addition, outright grants varying from 10 percent to 75 percent of project cost were awarded to five of these systems. These are typical of the financial cost of capital facing the water departments herein analyzed.[11] Capital is, however, rationed. Eligible projects are scored and queued, based largely on health considerations. In 2002 the total funding line for small New York water systems was drawn at $380 million, versus approved applications amounting to $548 million, for example. Unfortunately, the refinements of economic theory appear to exert little practical effect on the volume or mix of water supply projects that are funded. Boardman, *et. al.*, cite an unpublished survey that found that 43 percent of large U.S. municipalities don't even use discounting when evaluating projects (*op. cit.*, 249). This appears to be the case with New York's Drinking Water State Revolving Fund. Projects are scored and ranked based on health risk and compliance with the federal Safe Drinking Water Act. Broadly, it appears that municipal water operators decide to undertake water projects when they violate some regulatory standard or in response to system failures such as low pressure, discoloration, or frequent interruptions due to breaks. Financial feasibility is the main economic criteria: will user charges cover interest and amortization? Thus neither the price charged for public water nor the cost of capital confronting water system operators is likely to induce social efficiency.

Summary

If the results of the 75 systems herein analyzed are at all representative, it seems unlikely that there is serious under investment in public water supply infrastructure. Combining the median cost inefficiency with the median capital stock inefficiency implies annual deadweight costs of $94,800 per system. Extrapolated to the universe of the seven thousand comparable water systems nationwide implies annual inefficiency costs of $663,600,000. And the estimated returns on water system capital fall far short of a low estimate of social opportunity cost. The critique of Aschauer's work at the macro level is thus reinforced at the micro-level.

Appendix

Derivation of the Shadow Return on Capital

Let C = variable cost + fixed cost = V(Y, w, K) + $\sigma_K K$, where V is variable cost, Y is output, w is the price of the variable input(s), K is capital and σ_K the rental price of capital. Cost minimization with respect to K requires $\partial C/\partial K = \partial V/\partial K + \sigma_K = 0$, or $\sigma_K = -\partial V/\partial K$. Thus the derivative of the variable cost function with respect to the fixed capital input is the shadow price of capital (equation (3) in the text of the paper). For any level of output Y to be produced at minimum cost the scale of plant, measured by K, must be of optimal size. This requires that long-run marginal cost (LMC) be equal to short-run marginal cost (SMC) at that output: LMC = $dC/dY = \partial V/\partial Y + \partial V/\partial K(dK/dY) + \sigma_K(dK/dY) = \partial V/\partial Y + dK/dY(\sigma_K + \partial V/\partial K) = \partial V/\partial Y + dK/dY(0) = \partial V/\partial Y$ = SMC. Thus $\sigma_K + \partial V/\partial K = 0$ to make LMC = SMC [Layard and Walters (1978), 217]. If the size of the plant is too small, then $\sigma_K < - \partial V/\partial K$, and vice versa. The empirical evidence presented in the paper indicates that $\sigma_K > - \partial V/\partial K$, which implies that SMC < LMC at Y, since $dK/dY > 0$. In other words, more than the least cost size of the capital stock is being used to produce the observed level of output.

Since most capital, including public water supply capital, is not literally ‘rented’ on the open market it is necessary to proxy the rental price with the user cost of capital, i.e., the annualized interest and depreciation on invested capital. The Bureau of Economic Analysis estimates aggregate capital stock measures. Public water systems are estimated by BEA to have a service life of 60 years with declining balance depreciation, which implies annual depreciation of .0152 [BEA (2003)]. When combined with the real discount rate of .02 used in this paper, the annual user cost interest rate is .0352. This is a more stringent benchmark cost of capital than the

.03 actually employed and would reinforce the conclusion of over-investment, i.e., water plants are too large.

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Endnotes

1. Estimation of a cost function, frontier or not, neglects yet another potential source of economic inefficiency, namely over or under production of water because the price is not set at the social marginal cost of water.

2. The gamma density function is $f(u) = (\theta^P / \Gamma(P))e^{-\theta u} u^{P-1}$, $u, P, \theta > 0$. When P =1, the gamma reduces to the exponential distribution. Values of P greater than 1 allow the mass of the distribution to move away from zero, and values less than 1 resemble the exponential.P is the shape parameter, θ is the scale parameter and the mean = P/θ, and the variance = P/θ^2.

3. Ordinary least squares estimation yields the following equation (t-ratios in parentheses):
ln(VC/PE) = -4.65 + 0.962lnOutput + 0.942ln(Wage/PE) - 0.267ln(Capacity) +0.42Filtration -
(-4.14) (6.13) (4.85) (-2.07) (2.16)
0.635(Health Violations) + 0.129(Number of Sources); Adjusted R^2 = .73
(3.07) (2.19)
The simple correlation coefficient between the ols residuals and the (gamma) frontier inefficiency estimate (u_i) is .86. None of these coefficient estimates are statistically significantly different from the frontier model.

4. Five of the nine systems with one or more reported health violation since 1993 do not even disinfect (e,g., chlorinate) the drinking water, the most rudimentary form of treatment. Seven of the nine are recorded as exceeding maximum contaminant levels of coliform bacteria, a byproduct of human and animal excrement. Coliform is not itself a serious health risk, but is used as a marker for other possibly more dangerous bacteria. Nationally, the proportion of comparably sized systems with coliform violations is higher, fifteen percent versus nine percent of the sample in this paper [EPA (2002), vol.i].

5. $h(P, \epsilon_i) = E[z^P | z \geq 0]$ where $z = N[\mu_i, \sigma_v^2]$ and $\mu_i = \epsilon_i + \theta\sigma_v^2$. z is a random variable with mean and variance μ_i and σ_v^2. θ and P are the estimated parameters of the gamma distribution.

6. The estimated inefficiency does not include the effect of leaks in the distribution system, which are widely believed to be common. EPA [(2002), vol ii] reports that 10% of water produced by systems similar to those examined here is “unaccounted for.” In addition, a second-step truncated regression using the u_i as the dependent variable was estimated with the following explanatory variables: population, percent metered, income, number of poor people, and the ratio of capacity to average daily production. The latter is meant to test if excess peak load capacity might be a source of inefficiency. However, none of the coefficients is significant at even the 10% confidence level. The good news is that this result suggests that the frontier model has not omitted a variable correlated with the error term, a potential source of mis-specification bias.

7. A closer examination of the six outlier observations ($u_i > .70$) offers some clues as to the cause of their high level of inefficiency. The Village of Holley ($u = 2.38$) has production capacity of 1.4 million gallons but average daily production is only 175,000 gallons. The Village of Dexter ($u =$ 1.23) shows variable cost and total cost as both being $170,840. It could be that Dexter paid no debt service or acquired no new capital in 1999, but might well be a data error. The other four outlier villages display no obvious anomalies.

8. The national average residential water bill for systems of this size is $272 [EPA (2002), vol. ii].

9. As shown in the Appendix, fixed costs may be characterized as a rental price per unit of capital multiplied by the number of capital units (e.g., number of machines). This follows the standard textbook treatment that makes all input prices essentially hire prices. Alternately, one may specify fixed costs as the product of a percentage *rate* and the dollar value of capital. The latter corresponds to σ_K as a shadow rate of return whereas the former treatment equates to a rental price.

10. The New York State Environmental Facilities Corporation administers the State's Drinking Water Revolving Fund jointly with the Health Department. Municipal water systems must prepare detailed applications for financial assistance. From the Sept 20, 2000 Intended Use Plan, the following regression is estimated for the cost of constructing a new small water system:

Cost = $1.73 mil. + $990(Population); $R^2 = .25$. (Mean population 5812) N = 32.
(t = 6.44) (t = 3.15)

11. The New York State Environmental Facilities Corporation provided the author copies of eighty confirmation letters, granting approval of funding requests for drinking water investments from the Drinking Water State Revolving Fund. The mean project size is $1.8m, with sixty-five projects carrying a zero loan rate, five are charged a 1 percent interest rate, two are charged 3.65 percent, and eight projects were awarded grants equal to 100 percent of project cost. The State Environmental Facilities Corporation (EFC) issues tax exempt bonds and re-lends to local water systems at below the rate it pays, and also makes capital grants. The grants and subsidies are financed by federal Clean Water Act grants, State borrowing and interest earned on a portion of the federal grants set aside, plus interest arbitrage on the state issued debt. The EFC acting as an intermediary corrects a type of market failure because bond rating agencies will usually not devote the resources needed to assess the credit of very small government entities, effectively restricting their access to the municipal bond market.