Abstract - Seventy-five small, municipal water systems are analyzed to determine if they employ the least-cost long-run capital stock, and if they minimize cost given the observed level of capital. The social cost of capital exceeds the estimated return on investment by more than four times, and actual production costs are 36 percent above minimum costs. Median capital stock inefficiency, due to overinvestment, is $70,500 per system, and median cost inefficiency is $24,300. Extrapolated to the seven thousand similar systems nationwide, the combined cost of these two types of inefficiency is $663.6 million per year.

INTRODUCTION

The optimal amount of public sector investment and the efficiency with which it is used to provide public services are enduring topics of economic research. This paper addresses both of these issues in the specific instance of municipal water supply systems. A socially efficient level of public capital requires that the marginal return on water capital approximate the social cost of capital. If the estimated return exceeds the social cost, additional investment is called for, and vice versa. In addition, the invested capital and other associated inputs such as labor and energy should produce the desired output at minimum cost.

Much of the existing literature about the optimal amount of public capital has focused on the external effect of public capital on private sector growth. For example, Aschauer (1989) employed a Cobb–Douglas production function fitted to aggregate U.S. data to estimate that a ten percent increase in public infrastructure capital would increase private sector productivity by about four percent. This paper instead focuses on directly estimating the firm–level return on capital and the overall cost efficiency of a sample of typical small municipal water systems. The results show that the return on

---

1 Subsequent research has challenged this and related estimates 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.
capital falls far short of a quite modest annual cost of capital of three percent. Thus, at least in the case of public capital used in municipal water systems, the claim by Aschauer of significant under-investment is not supported. With respect to cost efficiency, the sampled systems incur costs that are 36 percent above least cost. The overall impression is of significant mislocation of resources in the area of public water supply.

It is relatively straightforward to evaluate the 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 unsafe 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 cost, 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 with greater precision.

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:

\[ E = \Sigma w \times = VC(Y, Q, w|K; \beta)e^{nu}, \]

or

\[ \ln E = \ln VC(\bullet) + \nu + u, \]

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 prices),
- \( x \) = 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,
- \( \beta \) = a vector of cost function parameters to be estimated, and
- \( \nu + u = \epsilon \), the composed error consisting of random noise, \( \nu \), plus cost inefficiency, \( u \).

Thus, the model contains a traditional variable cost function representing the least cost of producing different levels
of output \((Y, Q)\), given input prices, the quantity of the quasi–fixed input \((K)\) and available technology (summarized in the betas). Owing to the durability of water–system capital, natural monopoly and 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 to 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 optimal size of the capital stock is the amount so as to make the rental price of capital equal to the marginal return on capital. Cost inefficiency may arise from overuse of inputs, i.e., operating above the relevant isoquant, using the wrong mix of variable inputs, or using too large or small a stock of quasi–fixed inputs. Departures from the ideal of least cost may occur from uncontrollable random events such as weather (e.g., drought, floods, earthquakes, frost, etc.) or systematic departures due to management failures, perhaps resulting from 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 \(\nu\) 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 all departures are randomly distributed. The stochastic frontier model employed here appends a non–random error term to the model, which 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 \(VC(\bullet)\) 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 generally invariant with respect to the number of workers hired.

The translog and log–linear 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, 2000, 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,” after William H. Greene who first pointed it out (Greene,
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 under-utilized. Misallocation of quasi-fixed inputs is characterized as another type of cost inefficiency in the literature (Kumbhakar and Lovell, 2000, 146). Basically, the frontier inefficiency parameter 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 the observed level of output.2

As noted, stochastic frontier estimation requires some assumption about the probability density function of the inefficiency parameter. 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 . 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.

Following Kumbhakar and Lovell (op. cit., 145), the shadow price of the quasi-fixed input is given by:

\[ \rho_K = -\frac{\partial VC}{\partial K} = -\beta_K (VC^*/K), \]

where \( \rho_K \) is the estimated shadow price of capital (the Appendix derives this result), \( \beta_K \) 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 capital stock \( K \). One obtains \( VC^* \) by adjusting actual variable cost to purge 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, by comparing the shadow return \( \rho_K \) with an estimate of the social opportunity cost of capital.

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 (State of New York, 2001). All the systems serve a compact geographic area averaging only

---

2 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.
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 per–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 may use several supply sources (e.g., the number of wells supplying a ground water system), a variable to capture this likely source of 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 various 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 statistical 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 3,300 people, and relying primarily on groundwater sources (EPA, 2002b). Of the 75 systems in our data base, 64 fall into this population range, and 69 use only groundwater. Nationwide, the mean population of systems of this size and type is 1,440 people, versus 1,894 in this data set. Mean total expenses are $167,000, versus $156,725 for the sample 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, 2002b).

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 Limdep 8 software automatically tests if the composed error, \( \varepsilon \), is properly skewed in conformity with the frontier theory. If \( \varepsilon \) 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.

<table>
<thead>
<tr>
<th>TABLE 1</th>
<th>DESCRIPTIVE STATISTICS</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>( N = 75 )</td>
</tr>
<tr>
<td>Variable Costs</td>
<td>$120,197†</td>
</tr>
<tr>
<td>Daily Average Water Production</td>
<td>273,371 gallons</td>
</tr>
<tr>
<td>Maximum Production Capacity</td>
<td>661,123 gallons</td>
</tr>
<tr>
<td>Monthly Wages</td>
<td>$2,267</td>
</tr>
<tr>
<td>Electricity Prices (per KWH)</td>
<td>$0.09</td>
</tr>
<tr>
<td>Filtration Plant (1 = yes, 0 = no)</td>
<td>11</td>
</tr>
<tr>
<td>Health Violations (1 = yes, 0 = no)</td>
<td>9</td>
</tr>
<tr>
<td>Number of Water Sources</td>
<td>2.5</td>
</tr>
</tbody>
</table>

† standard deviation = 135,829.
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 0.31 (median 0.23). In the case of the half–normal, mean $u$ is 0.54, and for the exponential, the mean is 0.37. The cost function coefficient estimates are virtually the same across the three alternate frontier models (available from the author) and a least squares estimation—which is presented in the Appendix. 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 0.995.

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.

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):

$$[4] \quad RTS = \frac{[1 - \frac{\partial (\ln VC)}{\partial (\ln K)}]}{[\frac{\partial (\ln VC)}{\partial (\ln Y)}]} = \frac{1 - (-0.373)}{1.115} = 1.23.$$  

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.

ESTIMATED INEFFICIENCY

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

$$[5] \quad \mathbb{E}[u_i | \varepsilon_i] = \frac{h(P, \varepsilon_i)}{h(P - 1, \varepsilon_i)}.$$  

### TABLE 2

<table>
<thead>
<tr>
<th>Variable</th>
<th>Coefficient</th>
<th>t-ratio</th>
</tr>
</thead>
<tbody>
<tr>
<td>Constant</td>
<td>-4.94</td>
<td>-4.86</td>
</tr>
<tr>
<td>ln(Average Daily Production)</td>
<td>1.115</td>
<td>7.08</td>
</tr>
<tr>
<td>ln(Wage/Electricity Price)</td>
<td>0.860</td>
<td>4.55</td>
</tr>
<tr>
<td>ln(Capacity)</td>
<td>-0.373</td>
<td>-2.90</td>
</tr>
<tr>
<td>Filtration</td>
<td>0.458</td>
<td>2.25</td>
</tr>
<tr>
<td>Health Violations</td>
<td>-0.671</td>
<td>-3.75</td>
</tr>
<tr>
<td>Number of Water Sources</td>
<td>0.098</td>
<td>1.78</td>
</tr>
<tr>
<td>Theta ($\theta$)</td>
<td>2.383</td>
<td>2.71</td>
</tr>
<tr>
<td>$P$</td>
<td>0.745</td>
<td>1.28</td>
</tr>
<tr>
<td>Sigma $v$</td>
<td>0.379</td>
<td>4.59</td>
</tr>
</tbody>
</table>

Notes: Mean $u = P/\theta = 0.312$; mean inefficiency $= \exp(u) - 1 = 0.366$. Log likelihood function: -55.29. Variances: Sigma–squared ($v$) = 0.144; sigma–squared ($u$) = 0.131. N = 75.

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

4 Five of the nine systems with one or more reported health violations during the period 1993–1999 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—15 percent versus nine percent of the sample in this paper (EPA, 2002a).
where the composed error $\varepsilon_i = \nu_i + \mu_i$ for the $i^{th}$ producer, and $P$ is the estimated value of the shape parameter of the gamma distribution (Greene, 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.\(^5\)

The results reported in Table 3 seem plausible, with only a small number of outliers.\(^6\,\^7\) At the mean of $u_i$, least cost is $e^{-0.031} = 0.73$ of actual cost, so these water systems could theoretically reduce their variable costs by about 27 percent. Alternatively, actual variable costs are $e^{0.31} - 1 = 0.36$ 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 program-

<table>
<thead>
<tr>
<th>Range of $u_i$</th>
<th>Estimated Inefficiency</th>
<th>Number of Water Systems</th>
</tr>
</thead>
<tbody>
<tr>
<td>0.087–0.317</td>
<td>0.317–0.546</td>
<td>0.546–0.776</td>
</tr>
<tr>
<td>&gt;0.776</td>
<td>Total 75</td>
<td></td>
</tr>
</tbody>
</table>

Notes: Minimum—0.087; maximum—2.38; mean—0.31; median—0.23.

\(^5\) $h(P, \varepsilon_i) = E[z^P | z \geq 0]$, where $z = N[\mu, \sigma^2]$ and $\mu = \varepsilon_i + \theta \sigma^2$. $z$ is a random variable with mean and variance $\mu$ and $\sigma^2$. $\theta$ 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 (2002b) reports that ten percent 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 ten percent 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 ($u_i > 0.70$) 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 it might well be a data error. The other four outlier villages display no obvious anomalies.
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, leading to non-optimal capacity choices. 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 State 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 to the fixed capital stock (capacity variable) is, from Table 2, $\beta_K = -0.373$. 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 $0.373VC_i$ ($i = 1$ to 75).

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 two to eight percent range (Boardman, Greenberg, Vining and Weimer, 2001, Ch. 10). The U.S. Environmental Protection Agency (EPA) uses a seven percent real rate and a 20 year economic life when evaluating drinking water projects, with alternate rates of three and ten percent (EPA, 1999). Thus, a quite lenient estimate of the social opportunity cost of capital is a two percent real rate and a 50 year economic life, which implies an annual annuity factor of 0.03. 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 $433,500.11 Against

---

8 The national average residential water bill for systems of this size is $272 (EPA, 2002b).
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 capital costs as the product of a percentage rate and the dollar value of capital. The latter corresponds to $\sigma_K$ 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:

\[
\text{Cost} = 1.73 \text{ million} + 990(\text{Population}); R^2 = 0.25. (\text{Mean population—}5,812.) N = 32.
\]

\[t = 6.44\] \[t = 3.15\]

11 Multiplying the annual social cost of 0.03 by the mean replacement cost of $3.6 million gives $0.108 million per year.
this, the mean shadow return is $31,600 (median $21,500), ranging from $1,560 to $168,500.\textsuperscript{12} 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 EPA has estimated that New York State alone will have to spend ten 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 (State of New York, 2000, p. 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 does not exceed 1.75 percent of median household income (one 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 zero or one percent, and all but one for a period of 30 years. In addition, outright grants varying from ten 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.\textsuperscript{13} 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 do not even use discounting when evaluating projects (2001, p. 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.

\textsuperscript{12} For example, multiplying the cost function capital coefficient of 0.373 by the adjusted mean variable cost ($VC^*$) of $84,700 = mean shadow return of $31,600.

\textsuperscript{13} The New York State Environmental Facilities Corporation (EFC) 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 $1.8 million, with 65 projects carrying a zero loan rate, five of them charged a one percent interest rate, two charged 3.65 percent, and eight projects awarded grants equal to 100 percent of project cost. The 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.
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 $663.6 million. The published critiques of Aschauer’s work at the macro level are, thus, reinforced at the micro level. It is, however, conceivable that unmeasured marginal external benefits might justify capital investment in water supply beyond that justified by the direct return on investment, but the existing evidence in that regard is not very convincing.

Having identified significant misallocation of resources in small municipal water systems, further research is needed to see if this pattern holds for other system types. It is also important to understand better the reasons for the detected inefficiency so that policies might be developed to correct the situation. Apart from the obvious suggestion to privatize water supply, introducing more rigorous economic criteria for the ranking of projects and a more uniform subsidy policy to equate marginal borrowing costs across water systems may be indicated. Consolidation or regional water authorities are also a possible solution because significant economies of scale are present.

REFERENCES

Aigner, Dennis, C.A. Knox Lovell, and Peter Schmidt.

Aschauer, David.

Batina, Raymond G.

Boardman, Anthony E., David H. Greenberg, Aidan R. Vining, and David L. Weimer.

Braeutigam, Ronald R, and Andrew F. Daughety.

Bureau of Economic Analysis(BEA).

Greene, William H.

Greene, William H.

Greene, William H.

Greene, William H.

Grosskopf, Shawna, and S. Yaisawarng.

Grosskopf, Shawna, and Kathy Hayes.

Helfrich, Kathleen, and Donald F. Vitaliano.
“Size and Cost Efficiency in the Production of Local Road Services.” *Public Works


Derivation of the Shadow Return on Capital

Let \( C = \text{variable cost} + \text{fixed cost} = V(Y, w, K) + \sigma K \), where \( V \) is variable cost, \( Y \) is output, \( w \) is the price of the variable input(s), \( K \) is capital and \( \sigma \) the rental price of capital. Cost minimization with respect to \( K \) requires \( \frac{\partial C}{\partial K} = \frac{\partial V}{\partial K} + \sigma = 0 \), or \( \sigma = -\frac{\partial V}{\partial K} \). 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 \( (\text{LMC}) \) be equal to short-run marginal cost \( (\text{SMC}) \) at that output: \( \frac{\partial C}{\partial Y} = \frac{\partial V}{\partial Y} + \frac{\partial V}{\partial K} \frac{dK}{dY} + \sigma \frac{dK}{dY} = \frac{\partial V}{\partial Y} + \frac{dK}{dY} (\frac{\partial V}{\partial K} + \sigma) = \frac{\partial V}{\partial Y} + \frac{dK}{dY} (0) = \frac{\partial V}{\partial Y} = \text{SMC} \). Thus, \( \sigma + \frac{\partial V}{\partial K} = 0 \) to make \( \text{LMC} = \text{SMC} \) (Layard and Walters, 1978, 217). If the size of the plant is too small, then \( \sigma < -\frac{\partial V}{\partial K} \), and vice versa. The empirical evidence presented in the paper indicates that \( \sigma > -\frac{\partial V}{\partial K} \), which implies that \( \text{SMC} < \text{LMC} \) at \( Y \), since \( \frac{dK}{dY} > 0 \). 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 0.0152 (BEA, 2003). When combined with the real discount rate of 0.02 used in this paper, the annual user cost interest rate is 0.0352. This is a more stringent benchmark cost of capital than the 0.03 actually employed and would reinforce the conclusion of overinvestment, i.e., water plants are too large.

Least Squares Cost Function

Ordinary least squares (OLS) estimation yields the following equation (t–ratios in parentheses):

\[
\ln(\frac{VC}{PE}) = -4.65 + 0.962 \ln(\text{Output}) (6.13) \\
+ 0.942 \ln(\frac{\text{Wage}}{PE}) - 0.267 \ln(\text{Capacity}) (4.85) (-2.07) \\
+ 0.42 \text{Filtration} - 0.635 \text{Health Violations} (2.16) (3.07) \\
+ 0.129 \text{Number of Sources}; (2.19)
\]

Adjusted \( R^2 = 0.73. \)

The simple correlation coefficient between the OLS residuals and the (gamma) frontier inefficiency estimate \( (\mu_i) \) is 0.86. None of these coefficient estimates is statistically significantly different from the frontier model.