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The Economics of Efficient Phosphorus Abatement in a Watershed

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This study presents a method to determine efficient environmental targets at a watershed level. Efficient targets are devised by estimating abatement cost and cost of environmental damages and minimizing their sum. The method was applied to a case study of phosphorus pollution in a watershed in Oklahoma. Several cumulative scenarios with alternative abatement options were simulated and efficient targets were determined. As the number of abatement options at disposal to agricultural sources increased, their optimal abatement expanded relative to the abatement at the point source. Efficient targets were found to be dependent on the choice of policy that stimulates abatement.

Key words: efficiency, environmental targets, phosphorus pollution, watershed

Introduction

Water pollution from agricultural sources remains one of the most serious challenges for effective watershed management (U.S. Environmental Protection Agency, 2001). Economic analyses of surface water pollution typically involve a combination of economic and biophysical modeling (Khanna et al., 2003; Lintner and Weersink, 1999). This study also takes an integrated approach but differs from the previous literature in an important way. Most published work presents research that refers to exogenously set environmental targets. Such targets are typically determined through some form of political process that does not necessarily rely on any economic advice (Freeman, 2003). The role of economic modeling is reserved for devising a way to meet these targets at least possible cost.

The above approach, however, ignores the efficiency of the targets determined in this manner. Theoretically, there exists some socially optimal, efficient level of abatement that ensures efficiency in allocating resources to pollution abatement in a watershed. If it can be empirically determined, such an efficient level of abatement should be an objective of the environmental policy design. Despite the existence of economic models that can, in principle, represent the derivation of efficient environmental targets (Freeman,

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Haveman, and Kneese, 1973), empirical work in this direction has been lacking. Empirical determination of efficient abatement targets is complicated by the need to estimate the environmental damage costs associated with pollution. Some previous studies have reported estimates of environmental damage costs, but fell short of determining efficient environmental targets (Johansson, 2002).

Accordingly, the main objective of this study is to fill this knowledge gap by presenting a method for determination of efficient environmental targets for water quality, and to test the proposed method in a case study watershed. An additional objective is to identify an optimal allocation of abatement among nonpoint agricultural sources and a point source in the case study watershed. This goal is pursued by explicit modeling of abatement technologies at both types of sources.

Pollution in a watershed often originates from agricultural nonpoint sources and from point sources (Johansson, 2002; Randall and Taylor, 2000). Recent economic literature emphasizes agricultural abatement rather than point source abatement, and for good reason. High costs of additional abatement at the point sources are usually assumed because of the already high level of abatement necessary to comply with the existing point source discharge regulation. Although this is generally a reasonable assumption, there are instances where abatement at the point source would be more cost-effective than the corresponding abatement at agricultural nonpoint sources. This may be the case with smaller point source polluters, such as rural municipalities or concentrated animal feeding operations, which are not always tightly regulated. In these cases, there are cost-effective possibilities for abatement at the point sources.

Attaining environmental targets by optimal spatial allocation of abatement activities is one complexity related to nonpoint source pollution (Kaplan, Howitt, and Farzin, 2003, p. 107; Khanna et al., 2003; Shortle and Horan, 2001). Another complexity comes from an uncertainty about the magnitude of discharges from individual nonpoint sources. This analysis explicitly addresses the former, while pointing to possibilities to treat the latter within the proposed method. Additionally, and as an extension to models of watersheds presented in previous literature, a relatively large watershed is modeled, and at a high level of spatial detail.

Efficient phosphorus abatement targets were determined for the case study watershed of Eucha-Spavinaw in Oklahoma. This watershed has been a focus of the debate about imposing phosphorus total maximum daily loads (TMDLs) in Oklahoma (Haggard, 2003).[1] In this study, we examine the economic efficiency of alternative approaches to reduce phosphorus pollution in this watershed.

Conceptual Framework

The Socially Optimal Level of Pollution Abatement: Efficiency

Consider a pollution problem from the perspective of a hypothetical watershed manager. The manager's objective is to choose a level of abatement that maximizes total social welfare in the watershed, accounting for the interests of both the polluters and the

[1] Interestingly, only minimal funding from conservation programs ($87,000 under EQIP for the period 1996–2002, and much less under the CRP) has been devoted to ameliorating the problems with phosphorus discharges linked to animal production in this watershed (USDA/Natural Resources Conservation Service, 2002; USDA/Farm Service Agency, 2004).

parties affected by pollution. This problem can be represented by a model originally proposed by Freeman, Haveman, and Kneese (1973), a version of which was recently presented by Johansson (2002). In the current study this model is utilized to explicitly derive efficient environmental targets.

Suppose the abatement costs and the environmental damage costs are functions of emissions of a single pollutant, Z. Assume that social well-being can be succinctly summarized with two groups of goods and services, market (M) and environmental (E). The potential for producing market goods and services is reduced as resources are diverted to pollution abatement activities. On the other hand, the supply of environmental goods and services is reduced by pollution. The following optimization problem arises:

$$\text{(1)} \qquad \max_{Z} W(Z) = \overline{M} + \overline{E} - \big(A(Z) + D(Z)\big),$$

where W represents the total social well-being function, $\overline{M}$ denotes the maximum amount of market goods and services flowing from the watershed when no resources are devoted to pollution abatement, $\overline{E}$ represents the maximum potential supply of environmental services from the watershed in pristine condition, $A(Z)$ represents the costs associated with pollution abatement, and $D(Z)$ represents the cost of environmental damages caused by pollution.

The first-order condition for optimality is given by:

$$\text{(2)} \qquad \frac{dW}{dZ} = -\frac{dA}{dZ} - \frac{dD}{dZ} = 0,$$

where dA/dZ is the marginal abatement cost, and dD/dZ is the marginal environmental damage costs. It follows that at the optimum, the marginal abatement costs must be equal to the marginal environmental damage costs. In addition, the social well-being function is maximized when the sum $[A(Z) + D(Z)]$ is minimized [equation (1)]. Thus, the condition for efficiency of abatement in the watershed requires the sum of abatement and damage costs to be minimized, which occurs where the marginal abatement and marginal damage costs are equalized. To ensure a maximum of the welfare function, its second derivative must be nonpositive:

$$\frac{d^2W}{dZ^2} = -\frac{d^2A}{dZ^2} - \frac{d^2D}{dZ^2} \leq 0.$$

Specifically, this simply means that for an interior solution, marginal abatement costs are nondecreasing in abatement, and marginal damage costs are nondecreasing in pollution.

The presented model leads to an endogenously determined efficient environmental target. Let the solution to the optimization problem above [equation (1)] be denoted by $Z^*(A, D)$. If $Z^*(A, D)$ is a solution, then, by construction, it is the optimal level of pollution that maximizes social welfare, $W(Z^*)$. This level of emission represents an efficient environmental target. The efficient level of abatement can be found as the difference between the existing and optimum emissions, $Z - Z^*$. In empirical work, one has to estimate $A(Z)$ and $D(Z)$ to arrive at Z^*.

Estimating Abatement Cost:
Least-Cost Criterion–Cost-Effectiveness

To derive the abatement cost $A(Z)$ for agricultural pollution reduction, the conventional case of exogenous environmental targets can be analyzed. The model outlined below was used in the ensuing empirical work. Suppose an environmental target $\overline{Z}$ has been exogenously imposed on the system. Although this target is exogenous, and likely not efficient, it is still desirable to attain it at least possible cost (cost-effectiveness). Consider a watershed composed of n distinct heterogeneous land areas, each denoted by index i. Let the quantity of agricultural outputs from the ith land area be denoted by $\mathbf{Y}_i = f_i(\mathbf{X}_i)$, where $\mathbf{X}_i$ is a vector of input quantities used in area i, and $\mathbf{Y}_i$ is a vector of agricultural outputs produced from that land area. Each unique combination of inputs represents a separate production activity. Let Z_i be the amount of agricultural pollutant that leaves area i when the combination of inputs $\mathbf{X}_i$ is used. Pollutant discharges are a function $Z_i = g_i(\mathbf{X}_i)$ of the input combination $\mathbf{X}_i$ and of other factors, as described in Shortle and Horan (2001, p. 260). It is assumed the region of the watershed is sufficiently small so that vectors of output ($\mathbf{P}_y$) and input prices ($\mathbf{P}_x$) are fixed in the short run. The profit to the agricultural producer operating in the ith land area is expressed as:

(3) $$\Pi_i(\mathbf{X}_i) = \mathbf{P}_y^T f_i(\mathbf{X}_i) - \mathbf{P}_x^T \mathbf{X}_i,$$

where superscript T denotes transpose. Total net benefits from farming for the whole watershed are represented by the sum of profits over all n land areas,

$$\sum_{i=1}^{n} \Pi_i(\mathbf{X}_i).$$

Further suppose there is a point source of pollution discharging this same pollutant in the watershed. It is assumed the point source can employ an abatement technology to reduce pollution. The point source cost of abatement, *PSC*, is an increasing function of the abated pollution discharges from the point source, *ZPS*.

The problem for the entire watershed is then to maximize total social benefits[2] in the watershed net of point source abatement costs, subject to meeting the exogenous pollution target, $\overline{Z}$. The optimization problem can be expressed in a Lagrangian form as:

(4a) $$\max_{\mathbf{X}_i, ZPS, \lambda} L = \left(\sum_{i=1}^{n} \Pi_i(\mathbf{X}_i) - PSC(ZPS) \right) + \lambda \left(\overline{Z} - \sum_{i=1}^{n} Z_i - ZPS \right),$$

where the Lagrangian multiplier λ is the amount of change in the value of the objective function as the quantity of allowable pollutant emissions in the entire watershed is decreased (increased).

The first-order conditions for optimality are given by:

(4b) $$L_{\mathbf{X}_i} = \Pi_i'(\mathbf{X}_i) - \lambda\big(g_i'(\mathbf{X}_i)\big) = 0, \quad \forall i,$$

[2] This assumes agriculture is the predominant production activity in the watershed.

(4c) $$L_{ZPS} = PSC'(ZPS) - \lambda = 0,$$

(4d) $$L_{\lambda} = \overline{Z} - \sum_{i=1}^{n} Z_i - ZPS, \quad \forall i,$$

where $\Pi_i'(\mathbf{X}_i)$ and $g_i'(\mathbf{X}_i)$ are gradient vectors. From the first-order conditions [equation (4b)]:

(5) $$\lambda = \frac{\Pi_i'(\mathbf{X}_i)}{g_i'(\mathbf{X}_i)}$$

defines the marginal abatement costs for the *i*th agricultural nonpoint source. This implies that at the optimum, marginal abatement costs must be equalized across all n agricultural sources.

In light of equation (5), the condition given in equation (4c) states that at the optimum, the marginal abatement cost at the point source is equal to the marginal abatement cost at each of the nonpoint sources. The problem of comparing the abatement from the point and nonpoint sources is complicated by the uncertainty associated with the nonpoint source pollution emissions (Johansson, 2002, p. 222; Randall and Taylor, 2000). The question remains whether a nonpoint source could be treated as a point source with uncertain emissions and/or location (Kaplan, Howitt, and Farzin, 2003), with implications for the so-called "trading ratios" in a tradable pollution permit scheme (e.g., Tar-Pamlico as reported in Randall and Taylor). Although the present study does not explicitly address the problem of trading ratios of discharge reductions from point and nonpoint sources (Malik, Letson, and Crutchfield, 1993), the methodology applied allows for their incorporation into the analysis.

Data

Background on the Eucha-Spavinaw Watershed

The theoretical concept of efficient environmental targets described above was tested on the case of the Eucha-Spavinaw watershed in Oklahoma. A map of the watershed is presented in figure 1. Eutrophication of lakes Eucha and Spavinaw is caused by high phosphorus loading in the watershed attributed to excessive land application of litter produced by intensive poultry industry in the area. There are about 1,000 poultry houses in the watershed, housing approximately 120 million birds per year, and resulting in some 84,000 tons of broiler litter annually. Most of the litter is applied to agricultural land, predominantly for its value as nitrogen fertilizer. Unfortunately, the amount of litter that satisfies crop nitrogen requirements has a phosphorus content (usually 1.5% of the litter weight) far greater than the crop phosphorus requirements. A significant portion of the unused phosphorus applied with litter is accumulated in the soil, and gets deposited in the surface water as sediment bound, soluble, or organic phosphorus (Sharpley et al., 1999).

An additional source of phosphorus loading in the watershed is the discharge of municipal wastewater from the township of Decatur, Arkansas (Storm, White, and Smolen, 2002). The municipal sewage treatment plant in this town is combined with the treatment of effluent from a poultry processing plant located within the township.

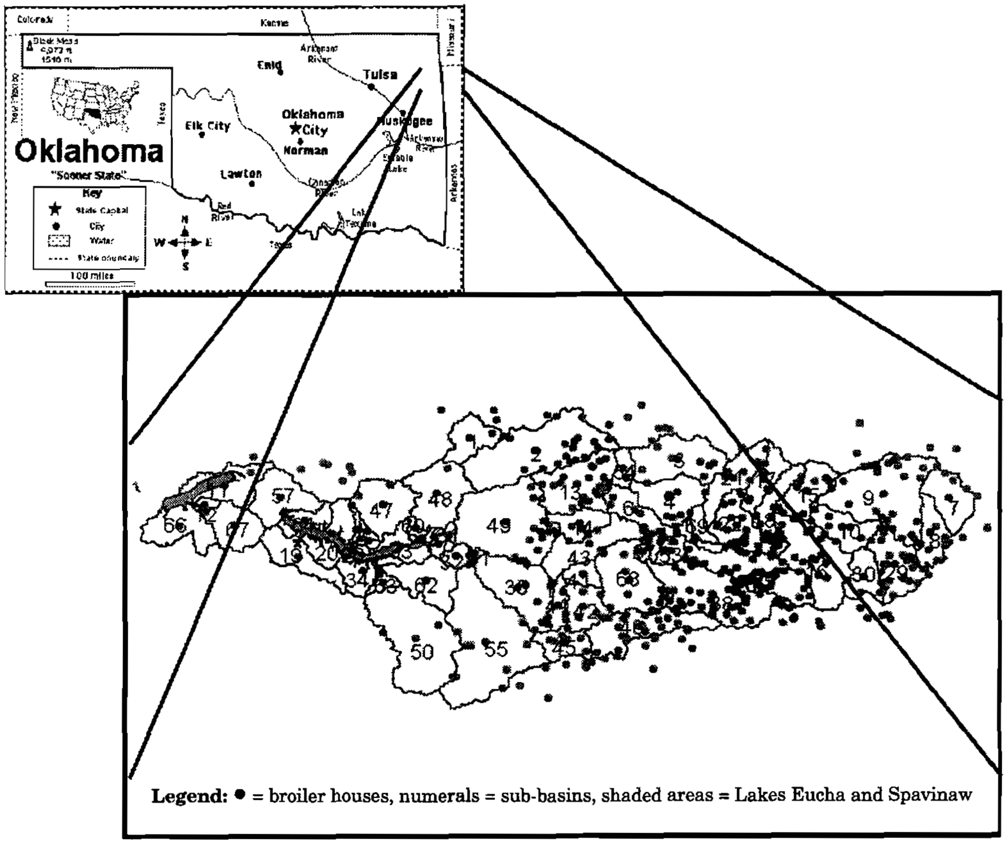

Figure 1. Map of the Eucha-Spavinaw watershed in Oklahoma

Storm, White, and Smolen estimate the phosphorus discharges from this plant at 12 tons per annum. Phosphorus loading from both agricultural and municipal sources results in increased concentration of phosphorus in streams and lakes in the watershed. Increased phosphorus concentration in the water promotes algal growth. Chemicals released by algae impart a bad taste and make the water undesirable for drinking (Oklahoma Water Resources Board, 2002). Recreational use of the lakes in the watershed has also declined as algae levels have risen.

The SWAT Model

The Soil and Water Assessment Tool (SWAT) was used as a biophysical model for the watershed (Arnold et al., 1998; Srinivasan et al., 1998). SWAT is a GIS-based, basin-scale, continuous-time hydrologic model that is capable of simulating hydrology, sediment, and nutrient dynamics in a large watershed or a basin. The following geographic information systems (GIS), weather, and management data were used in the SWAT model: Digital Elevation Model data (U.S. Geological Survey), soil data layer (STATSGO), agricultural management data (Oklahoma State University extension fact sheets), soil nutrient availability data (soil samples), stream flow data (U.S. Geological Survey), and precipitation data (Cooperative Observation Network gage data). To complement precipitation data, high detail daily rainfall estimates were derived from Next Generation Weather Radar data (National Oceanic and Atmospheric Administration) and incorporated into the model. Land cover data were developed from satellite imagery and ground truthing.

Within the SWAT model, the watershed was partitioned into 69 sub-basins (figure 1) and 1,052 hydrologic response units (HRUs), with an average of 16 HRUs per sub-basin. A sub-basin is a unique collection of streams that drain to a single outlet. An HRU is a homogeneous land unit with respect to soil type and land use. Out of 1,052 HRUs (on approximately 100,000 ha) in the watershed, 695 were agricultural (comprising 45,800 ha, or 46%) and the remainder were forest (50,900 ha, or 51%), urban (1,300 ha, or 1.3%), and water (1,700 ha, or 1.7%). The average agricultural HRU contained 66 hectares. However, 435 of the 695 agricultural HRUs were smaller than 50 ha, and 112 were smaller than 5 ha.

SWAT was calibrated to match recorded stream and base flow as well as organic, sediment bound, and soluble phosphorus. The calibrated SWAT model was used to run simulations for the period 1993–2002 and to estimate the current average annual phosphorus loading from nonpoint sources in the watershed (34 tons/year). Nonpoint sources comprised agricultural areas under the following land covers: hay, well-maintained pasture, overgrazed pasture, and row crops. Table 1 presents the areas of agricultural land covers and their contribution to the total land area and phosphorus loading in the watershed. As shown by table 1, row crops and overgrazed pastures contribute disproportionately to the phosphorus loading in the watershed relative to the land area they occupy.

Nonpoint Sources Abatement Cost Data

Determining the costs of phosphorus abatement for nonpoint agricultural sources was a complex task. The true cost structure is difficult to estimate because of the heterogeneity of agricultural enterprises, including the differences in management. The actual abatement costs could be revealed through some form of conservation auctions, where farmers bid for government compensations (Babcock et al., 1996). Alternatively, as was done here and in previous studies (e.g., Shortle and Horan, 2001, p. 260), abatement costs could be estimated as a change in quasi-rents between the current (non-abatement) management regime and agricultural management regimes conducive to reduction of phosphorus loading.

Several alternative management practices, summarized in table 2, were simulated for each of the four agricultural land uses. One practice was to vary the rate of litter application, combined with possible substitution of the required nitrogen from commercial sources. Nitrogen content of poultry litter is 5% (1% mineral and 4% organic), while phosphorus content is 1.5% (0.4% mineral and 1.1% organic). One considered option ("with commercial N" column in table 2) was for each simulated litter application rate lower than the existing rate, to apply commercial nitrogen to fill the difference created by lowering litter application. For example, for "0.75 of existing" for hay, 50 kg of active nitrogen per hectare was added [$50 = (4{,}000 - 0.75 \times 4{,}000) \times 0.05$, which amounts to109 kg/ha Urea]. Another considered option ("without commercial N" in table 2) was to not compensate for nitrogen reductions caused by lower litter application. Both of these

Table 1. Agricultural Land Covers in the Eucha-Spavinaw Watershed

Land Use	Land Area: hectares (acres)	Land Area (% of total)	Average Annual P Load: kg/hectare (lbs./acre)	Average Annual P Load (% of total)	Existing Litter Application Rate: tons/hectare (lbs./acre)
Grassland used for pasture (well-maintained)	23,250 (58,125)	23.1	0.46 (0.40)	23.2	4 (3,525)
Grassland used for hay	13,402 (33,505)	13.3	0.34 (0.30)	9.8	4 (3,525)
Grassland used for pasture (overgrazed)	6,542 (16,355)	6.5	0.81 (0.71)	11.5	2.15 (1,895)
Row crop (grazeout wheat in rotation with green beans)	2,625 (6,563)	2.6	2.31 (2.03)	13.2	1.3 (1,145)

Table 2. Simulated Practices for Each Agricultural Land Use in the Eucha-Spavinaw Watershed

Land Use Change? (Yes / No)			
Alum-Treated Litter		Non-Treated Litter	
With Commercial N	Without Commercial N	With Commercial N	Without Commercial N
— Litter Application Rates[a] —			
	1.50 of existing		1.50 of existing
	1.20 of existing		1.20 of existing
	existing		existing
0.85 of existing	0.85 of existing	0.85 of existing	0.85 of existing
0.75 of existing	0.75 of existing	0.75 of existing	0.75 of existing
0.50 of existing	0.50 of existing	0.50 of existing	0.50 of existing
0.25 of existing	0.25 of existing	0.25 of existing	0.25 of existing
no litter application	no litter application	no litter application	no litter application

[a] Existing application rate for each of the land uses was as given in table 1 above.

options were considered for applications with alum treated litter and with non-treated litter. Chemical treatment of poultry litter with alum prior to its land application was another simulated practice. Alum (aluminum sulfate) reacts with soluble phosphorus in the soil to create non-soluble forms. When litter is treated with alum, soluble phosphorus runoff is reduced by 75% (Moore, Daniel, and Edwards, 1999). The cost of treating litter with alum was assumed at 2% reduction of net income, based on arguments put forward in some previous studies (Moore, Daniel, and Edwards) and interviews with local extension staff.

Another possibility for reducing phosphorus loading from agricultural sources is by land use conversion, which was simulated in SWAT by converting land uses with high phosphorus loading potential (overgrazed pastures and row crops) into land uses with lower phosphorus loading potential (well-maintained pasture and hay) (table 1). An intuitive possibility would be to simulate withdrawal of land from production activities, as for example under the Conservation Reserve Program (CRP) (Babcock et al., 1996; Feather and Hellerstein, 1997; Johansson and Randall, 2003). However, the shortage of land available for litter application is the crux of the problem in this particular watershed. There is too much litter produced and too little land on which to apply it. Land retirement would exacerbate the phosphorus loading problem, since more litter would be applied to remaining land available, creating conditions for even more phosphorus loading. This example has actually been reflected in practice, with only 50 acres being retired under the CRP in the period ending 2003, at a rental price of $41.73 per acre (USDA/Farm Service Agency, 2004).

Another intuitively appealing method for reducing phosphorus loading would be to reduce the number of birds raised in the watershed. With profit margins for broiler production estimated at 6¢ per bird (Cunningham, 2003), a simple calculation shows that reducing the quantity of broiler litter in the watershed by reducing the number of birds would amount to 2.2¢ per pound of litter. Thus, to reduce the litter produced in the watershed by 20% (from 84,000 to 67,200 tons), it would cost $756,000 in terms of foregone profits from broiler production.[3] Although not considered as an option in this study, it can be used as an indicator for backstop costs.

Net income (quasi-rent) for each of these alternative management practices for the individual agricultural land areas in the watershed was calculated using the Oklahoma Enterprise Budget digital worksheets (Oklahoma Cooperative Extension Service, 2003). The calculations were based on simulated yields, biomass production, and grazing intensity for each HRU in the SWAT model, as well as on the level of applied commercial fertilizer (table 2).

Transportation Cost Data

The empirical model incorporated poultry litter transportation within and outside of the Eucha-Spavinaw watershed. Costs of hauling litter *within the watershed* were estimated based on road distances between sub-basins, and on litter demand based on crop requirement in a sub-basin, and litter supply based on number of poultry houses in a sub-basin. The litter produced in the watershed either had to be applied to land in the watershed or transported out. The distances necessary to haul litter *out of the watershed* were determined by locating counties to the east, north, west, and south of the watershed having sufficient capacity to accept manure phosphorus application, based on a method described by Gollehon et al. (2001). Costs of litter transportation within and outside the watershed were incorporated as transportation activities in the mathematical program.

Point Source Abatement Cost Data

The costs of abatement at the point source were estimated using engineering cost data. An abatement system that could utilize various levels of chemical treatment to precipitate phosphorus in the effluent was modeled and costs were calculated for various levels of abatement. Chemical treatment using a secondary process was chosen for modeling due to its relative simplicity and cost-effectiveness for comparably small wastewater

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[3] For each pound of live weight, there is roughly half a pound of litter produced. Each bird has a live weight of about 5.4 pounds. This gives approximately 1.3 tons of litter per 1,000 birds. We are grateful to a reviewer for pointing out that the costs of reducing the number of birds could be treated as an indication of backstop costs.

treatment plants. The data for cost calculations were obtained from several publications, including Metcalf & Eddy (2003) and Klute and Hahn (1994). Costs were calculated for 34 levels of abatement, gradually reducing the phosphorus concentration in the effluent from the current 6.6 mg/liter to 1.0 mg/liter, which is considered to be an acceptable value (Metcalf & Eddy). These costs were subsequently used for constructing abatement activities for the point source in the mathematical program.[4]

Environmental Damage Cost Data

To empirically estimate the environmental damage cost function, $D(Z)$, two major classes of environmental damage costs were identified in the watershed: direct costs of additional treatment of drinking water, and the loss of recreational values. The impacts on ecological values in terms of habitat change or alteration of species composition for this watershed are not well documented, and therefore were not included in the analysis.[5]

Damage to Drinking Water Quality

Cost data for the additional drinking water treatment were obtained through direct communication with the Tulsa Municipal Utility Authority (TMUA). Monthly cost data covered the period from July 1995 through December 2002. To control the odor- and taste-causing chemicals, TMUA uses additional filtration with powdered activated carbon. The powdered activated carbon is effective in removing odor and improving the taste of drinking water (American Water Works Association, 2001), but is quite costly (its price is around $0.2/kg).

Regression analysis was used to estimate the relationship between the level of phosphorus pollution in the watershed and the costs to the TMUA for additional water treatment. Initially, a lagged model was estimated, where the observed monthly costs of water treatment were regressed on the lagged phosphorus concentration in the water. Phosphorus concentration measures were obtained from TMUA for the period covering the cost data (1995–2002). The interpretation of the results was complex and their use in the further calculations was limited. Therefore, the model was respecified using annually grouped cost data. Variability in water treatment costs over the years was explained by the variability in phosphorus loading in the watershed. Annual phosphorus loading estimates were obtained from seven sets of SWAT simulations corresponding to seven alternative average phosphorus discharge levels (Storm, White, and Smolen, 2002).[6] For each of these seven levels, phosphorus discharges were simulated for eight years (1995–2002, inclusive) to match cost data.

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[4] The following exponential abatement cost function could be fitted through those 34 abatement points: $PSC = 34{,}544\, e^{0.0002\, ZPS}$, where PSC denotes point source abatement costs, and ZPS denotes phosphorus abated at the point source. This function is provided here for illustrative purposes only. The abatement cost points themselves were used in the mathematical program as abatement activities for the point source.

[5] Mathews, Homans, and Easter (1999) conducted a study estimating the impacts of pollution on ecological values of the Minnesota Valley National Wildlife Refuge. The study area treated in the current article is quite different in that it does not contain any wildlife parks, refuges, or other established conservation areas. Because of this, even though the ecological values are undoubtedly present, there is not a meaningful and practical way to measure the damages to ecological services caused by phosphorus pollution.

[6] There were seven levels of average simulated phosphorus discharge expressed in tons per year (18, 20, 25, 30, 35, 40, and 46). Each of these was associated with an alternative set of assumptions about the watershed, most notably the intensity of poultry litter application. See Storm, White, and Smolen (2002) for more detail.

For each individual level of average phosphorus discharge, the observed annual costs were regressed on the simulated annual phosphorus loadings:

$$CT_{kt} = b_0 + b_1 Z_{kt} + \mu_{kt}, \tag{6}$$

where CT_{kt} denotes the observed annual cost to the Tulsa Municipal Utility Authority in year t ($t = 1, ..., 8$) under the kth level of average phosphorus discharge ($k = 1, ..., 7$), Z_{kt} denotes the average phosphorus discharge in the watershed, and μ_{kt} is a normally distributed random error term. Estimation was conducted using ordinary least squares. The parameter b_0 was estimated as −226,394 with a t-ratio of −5.36, and the parameter b_1 was estimated as 11.14 with a t-ratio of 10. The R^2 was 0.971. This specification of the functional form was influenced by the range of the available data. Had the data for more extreme levels of phosphorus discharge and the costs of ameliorating them been available, another (in all likelihood nonlinear) specification would have been appropriate.

Damage Due to Loss of Recreational Values

The costs of recreational losses were estimated within the travel cost method framework. Data on annual visitations in the two state parks in the watershed, as well as data for possible substitute sites, were obtained through direct communication with the Oklahoma Tourism and Recreation Department. Annual visitation data were for the period 1990–2001. The origin of the visitors to the state parks was determined using data published in a survey conducted by the Oklahoma Conservation Commission (1997). Population data were obtained from the *U.S. Census*, and published income data were used to value the recreation time (U.S. Department of Commerce, 2000).

Visitation data from Eucha and Spavinaw State Parks were used in a zonal travel cost method (ZTCM) to determine the demand for recreation and the loss of recreational values as a result of phosphorus pollution in the lakes. The two state parks are close to each other (approximately 6.5 miles) and their site characteristics are quite similar. Furthermore, Spavinaw State Park significantly dominates Eucha State Park in terms of recreational visitation, accounting for 94.5% of the visits during the sample period. These characteristics prevent the potential problems with recreation demand aggregation across sites (Shaw and Shokwiller, 2000) and the need for using a price index for multiple sites (Phaneuf, Kling, and Herriges, 2000).[7]

Four iso-travel cost zones were constructed, and travel costs from each zone were calculated using road distances, average gasoline consumption, and gasoline prices. An average length of a visitor day to the state parks was assumed to be 10 hours for all visitors, so that visitors from farther away spend more time traveling and less on actual recreation than the visitors who live nearby. The value of time (McConnell, 1992) was incorporated in the travel cost estimates using a rate of one-third of the estimated hourly earnings (Shaw and Feather, 1999, p. 592).

[7] We are grateful to associate editor Paul Jakus for pointing out these potential issues.

An inverse demand function for recreation was estimated according to the following:[8]

$$TC_{lm} = \sum_{m=1}^{12} INT_l^m D_l^m + dQ_{lm} + \varepsilon_{lm}, \tag{7}$$

where TC_{lm} denotes the travel cost to the state park from the lth zone when the average annual phosphorus concentration in the lakes was m. Here the average annual phosphorus concentration was used as a class variable. D_l^m is a dummy variable for each level of phosphorus concentration, while INT_l^m denotes the vertical intercept of the inverse demand function. For each level of phosphorus concentration, these intercepts may be interpreted as maximum willingness to pay for recreation under the given environmental circumstances. This design was pursued to represent the effects of quality change in a recreational site on the demand for trips to that site, where the water quality (phosphorus concentration) was modeled as an intercept shifter. Q_{lm} is the observed number of visits from zone l under phosphorus concentration m, d is the slope parameter to be estimated, and ε_{lm} is a normally distributed random error term.

Maximum likelihood estimation was conducted and the results are presented in appendix table A1. These results are expressed in terms of phosphorus concentration over the 12 years for which observations were available. However, the estimates had to be translated in terms of phosphorus loading to be complementary with the SWAT estimates, and to enable derivation of welfare measures for the seven levels of simulated phosphorus loadings, as presented in appendix table A2. This was done by first establishing the relationship between the estimated intercepts, $\widehat{INT}^m$, and the phosphorus concentration:

$$\widehat{INT}^m = 72.7 - 788.5PC^m, \tag{8}$$

and then, by converting the measures of phosphorus concentration to phosphorus loading (Zmax), according to the formula: $PC = 0.0105 + 0.00000066Z\text{max}$, using information published by the Oklahoma Water Resources Board (2002, pp. 120–121).[9] This relationship enables the calculation of consumers' surplus under the seven levels of considered phosphorus loading in the watershed. The estimates of consumers' surplus and the change in consumers' surplus at the various levels of phosphorus loading were derived using standard procedures and are reported in appendix table A2.

[8] A standard demand function for a ZTCM was initially considered, but the inverse demand function was chosen for ease of interpretation and better visualization of the intercept shifts as a result of changing phosphorus concentration over time. Since the model is linear, the two expressions (the standard and the inverse demand functions) yield the same results in terms of welfare measures. Another nonstandard feature of the model in equation (7) is the inclusion of phosphorus concentration as a class variable, rather than as a standard continuous variable. Both of those representations can act as intercept shifters, which was the intention here. The estimated model with phosphorus concentration as a continuous variable was:

$$\underset{(3.73)}{TC_l = 72.2} - \underset{(-1.59)}{778.15Pconc} - \underset{(-20.9)}{0.0016Q_l},$$

with t-ratios in parentheses. The coefficient on *Pconc* is statistically insignificant at $\alpha = 0.10$. This led us to estimate a model with phosphorus concentration as a class variable (reported in appendix table A1). The differences between the two models were minor. Average difference between the calculated intercepts for the two models was 0.094 (with a minimum of 39.08 and a maximum of 43.16 for the model reported in table A1, and a minimum of 40.96 and a maximum of 42.87 for the model with continuous phosphorus concentration). The coefficient on number of visits was the same for the two models.

[9] Phosphorus concentration was used in the travel cost method because it is a water quality parameter that has more direct impact and could therefore more directly affect visitors' perceptions, as opposed to largely unobservable phosphorus loading.

Estimates of Total and Marginal Environmental Damage Costs

Estimated cost for additional drinking water treatment and estimated cost of recreational losses were summed together for each of the seven considered phosphorus loading levels. A total damage cost function $D(Z)$ was fitted through these cost points (with t-ratios in parentheses):

$$D(Z) = \underset{(10.25)}{585{,}446.9} - \underset{(-15.45)}{59.93Z} + \underset{(25.18)}{0.0015Z^2}. \tag{9}$$

The marginal environmental damage cost is then

$$MDC(Z) = -59.93 + 0.003Z. \tag{10}$$

Method

The method consisted of two steps. SWAT simulations and estimation of the point and nonpoint sources abatement costs, $A(Z)$, and the environmental damage costs, $D(Z)$, were first conducted. The alternative agricultural management practices (table 2) were simulated using SWAT. The resulting estimates (yield, phosphorus discharges) from the SWAT simulations were then integrated in a mathematical program to derive efficient targets for each of the considered scenarios.

Finding the Optimal Solutions

There are two alternative avenues to derive efficient environmental targets using mathematical programming. One approach is to derive a marginal abatement cost curve by parametrically varying exogenous phosphorus discharge limits. The shadow price on the pollution constraint in each solution represents a point on the marginal abatement cost curve [equation (5)], which could be traced by repeatedly solving the program and connecting the resulting points. The efficient target will be found at the point where the estimated marginal damage [equation (10)] and marginal abatement cost curves cross, as required by the equilibrium condition set in equation (2). An alternative approach is to segment the estimated total environmental damage cost function [equation (9)] and use it to design environmental damage cost activities in the mathematical program itself.

Mathematical Programming Framework

The objective function of the mathematical program was to maximize net agricultural income (quasi-rents) from all areas (HRUs) in the watershed less the point source abatement costs and the cost of litter transportation within and out of the watershed, by choosing alternative agricultural management practices with various potential for phosphorus loading in each HRU. Five scenarios were simulated within the programming framework. A summary of the scenarios and their main characteristics is given in table 3. For each scenario, choice variables in each of the HRUs were specified as described in tables 2 and 3. The considered scenarios were cumulative, so that one scenario was contained in the next. This is also presented in the left-most column of table 4. Scenarios 2 and 5 simulated a policy that restricts litter application to soils with soil test phosphorus (STP) less than 120 pounds per acre.

Table 3. Summary of the Considered Scenarios and Their Characteristics

	Scenario				
Characteristic	1	2	3	4	5
Variable litter application rate	Yes	Yes	Yes	Yes	Yes
STP < 120 criterion	No	Yes	No	No	Yes
Alum use	No	No	Yes	Yes	Yes
Land use change	No	No	No	Yes	Yes

Table 4. Efficient Phosphorus Targets for the Eucha-Spavinaw Watershed Under Various Abatement Scenarios

Scenario	Efficient Phosphorus Emission Target [Zmax] (kg/year)	Value of the Objective Function at the Optimum ($/year)	Shadow Price[a] [MAC = MDC] ($/kg P)	Abatement Costs to the Agric'l. Sources ($/year)	Abatement Costs to the Point Source ($/year)	Total Damage Costs ($/year)	Sum of Total Abatement Cost and Damage Cost ($/year)
Scenario 1 (variable litter application)	34,115	5,048,053	43.57	65,067	149,390	286,685	501,142
Scenario 2 (Scenario 1 + STP < 120)	43,367	3,380,050	70.17	1,698,100	184,360	807,508	2,689,968
Scenario 3 (Scenario 1 + alum use)	28,834	5,254,412	25.56	98,573	130,133	104,525	333,231
Scenario 4 (Scenario 3 + land use change)	24,697	5,610,182	14.16	134,672	33,113	20,268	188,053
Scenario 5 (Scenario 4 + STP < 120)	25,000	3,673,977	15.07	1,404,173	3,065	24,670	1,431,908

[a] The shadow price stated in the table represents the equilibrium point where marginal costs of phosphorus abatement (MAC) are just equal to marginal damages from phosphorus pollution (MDC).

These various scenarios were run in order to demonstrate the dependency of the efficient environmental targets on the abatement options available to farmers under various policy settings. It was expected that efficient targets will be lower (more abatement) as the set of available options to farmers becomes less restricted. In this sense, scenario 4 was the least restrictive and scenario 2 was the most restrictive.

The model was continuous (non-integer), so that fractions of activities in an HRU were possible. The program was subject to the usual convexity and nonnegativity constraints, as well as to the accounting constraints regarding litter treatment with alum and the transportation of litter. The program was repeatedly solved using an exogenously parameterized phosphorus discharge constraint. The parameterized discharge levels were

$$Z = \left\{ Z_{\max} \middle| Z_{\max} \in \left[18, 20, 25, 30, 35, 40, 46\right],\ \forall Z_{\max} \right\}$$

in tons per year, identical to the previously mentioned seven average phosphorus loading levels. This process was conducted only for scenarios 1, 3, and 4, which did not contain the STP criterion, because under the condition of STP < 120, not all parameter values for the phosphorus constraint were feasible.

Results

The results from programming runs are summarized in table 4. For each of the scenarios the reported shadow prices represent both the marginal costs of abating an additional unit of phosphorus in the watershed and the marginal damage caused by an unabated unit of phosphorus. The graphs of derived marginal abatement curves for scenarios 1, 3, and 4, and the marginal damage curve are presented in figure 2.

For scenario 1, the determined efficient target for phosphorus loading was a little over 34 tons/year, where the total annual costs of abatement and environmental damages were estimated at around $500,000. The optimal solution for this scenario was also characterized with intensive transport of litter within the watershed, but not outside of the watershed, reflecting the relatively lax phosphorus constraint.

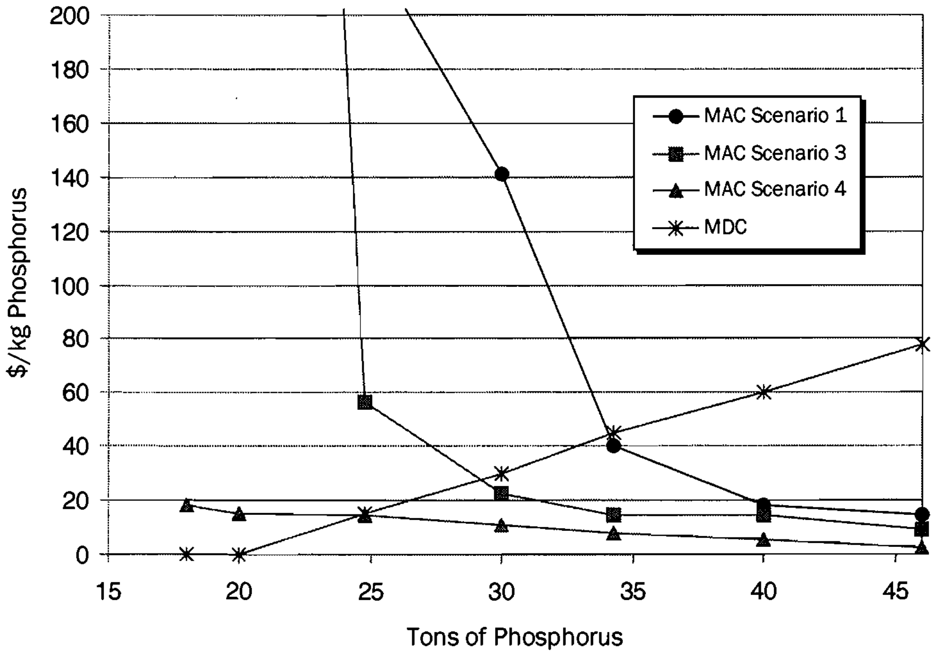
For scenario 2, litter could only be applied to soils where STP was less than 120 pounds per acre. The results showed it was optimal to apply as much litter as possible on the land where its application was allowed. This perverse incentive was due to the high costs of transportation of litter outside the watershed, which had to be undertaken to meet the STP criterion. This scenario had highest total costs at the watershed level and the lowest amount of phosphorus abatement (highest discharges) of all considered scenarios, reflecting the restrictiveness of this stringent regulatory approach.

The efficient target for scenario 3 was determined at about 29 tons/year. For this scenario it was optimal to use alum-treated litter to reduce soluble phosphorus runoff. Some 53,800 out of 84,000 tons of total poultry litter were treated with alum under this scenario.

An efficient phosphorus target of about 24.7 tons/year was estimated for scenario 4. Under this scenario it was found optimal to convert all overgrazed pasture to well-maintained pasture. Well-maintained pasture has a reduced stocking rate, higher commercial nitrogen application rate, and a higher biomass threshold before grazing is allowed. In addition, approximately 42% of the current 2,625 hectares of row crops did convert to hay under this scenario.

For scenario 5, the efficient target was estimated at approximately 25 tons/year. This scenario was characterized by perverse incentives similar to those found for scenario 2, but their impact was moderated by the possibility for land use change. This resulted in significant phosphorus abatement, but at much higher cost as compared to scenario 4, which was due to the significant litter transport outside the watershed under this scenario.

The optimal amount of abatement at the point source varies inversely with the set of abatement options available to agricultural nonpoint sources. These results are also presented in table 4. For scenario 1, the point source optimally abated 9.3 tons of phosphorus at a cost of $149,390, while the agricultural sources abated only 1.7 tons at a cost of $65,067 per year. For scenario 2, the point source abated 10 tons at a cost of $184,360. Despite the high abatement at the point source, the cost of inevitable transport of the excess litter outside the watershed reduced the agricultural income by $1.8 million under this scenario. For scenario 3, the point source optimally abated 8.5 tons per year at a cost of $130,133, while the agricultural sources abated 7.5 tons at a cost of $98,573. Under scenario 4, only 0.5 tons were abated by the point source at a cost of $33,113; the majority of abatement occurred at agricultural sources, which abated 20.5 tons at a cost of $134,672. Similarly for scenario 5, only 0.2 tons of phosphorus were



Note: Because of the nature of the regulatory policy simulated, the mathematical programs for Scenarios 2 and 5 were not run with an explicit phosphorus constraint. Consequently, estimates for marginal cost of abatement were not simulated for these scenarios.

Figure 2. Traced marginal abatement cost curve (MAC) for Scenarios 1, 3, and 4, and the marginal damage cost curve (MDC)

optimally abated at the point source. Allowing for land use changes as an abatement option for the agricultural sources not only shifted the optimal allocation of abatement from point toward nonpoint sources, but also considerably reduced the marginal abatement cost.

Summary and Conclusion

The main message this article conveys is that economics should play a greater role in setting environmental targets because it can be instrumental in determining an efficient level of pollution/abatement in a watershed. All too often economists provide advice on meeting already set environmental targets at least cost, without questioning their efficiency. Even though the dedication to cost-effectiveness is compelling, the profession should make use of newly available techniques and methods to determine efficient environmental targets. This study has presented one method for derivation of such endogenous pollution abatement targets. The method was applied to the case study area of the Eucha-Spavinaw watershed in Oklahoma. Rather than just meeting exogenously set targets at least cost, the empirical study endogenously determined efficient targets.

Emission-based abatement cost estimates for both point and nonpoint sources of phosphorus loading in the case study area were presented. One limitation of the proposed model is that the estimated discharges from agricultural sources were treated as nonstochastic, in an attempt to keep the analysis on the level of applicability for actual policy design. The method proposed by Teague, Bernardo, and Mapp (1995) could be employed to incorporate stochastic emissions estimates in the model, because the biophysical simulator used (SWAT) has the capability of computing the variability of estimated parameters (Eckhardt, Breuer, and Frede, 2003). This capability of the simulator can also be used in computations of trading ratios for a possible point versus nonpoint tradable permits scheme.

Several conclusions could be drawn from the results derived from the case study. First, it appears that an efficient environmental target for phosphorus loading in the Eucha-Spavinaw watershed could be set in a range of 25 to 30 tons per year (abatement of 15 to 21 tons/year), dependent on the abatement options available to the agricultural sources. Further, the simulated use of the stand-alone STP criterion (scenario 2) resulted in highest costs of reducing phosphorus loading and lowest abatement. If an STP policy is to be implemented, then it should be used in conjunction with other abatement options (scenario 5, table 4). This policy would result in a low phosphorus target, but at higher cost than necessary. Instead, land use changes (scenario 4) seem to be an efficient long-term solution to the problem of phosphorus pollution in this watershed. Such land use conversion would, however, require considerable changes in the economic structure of the agricultural production in the watershed and some innovative policy design.

The determined efficient targets under the various simulated scenarios are dependent on a specific choice of policy. If the policy choice in the case study watershed was to only encourage optimal litter application rates, an efficient phosphorus target should be set at 34 tons per year (scenario 1, table 4). If, in addition to this policy, alum use is encouraged (e.g., through a subsidy on the cost of alum), then the efficient phosphorus target should be 28.8 tons per year (scenario 3, table 4). If the policy goes further and provides incentives for land use changes, the efficient phosphorus loading target for the watershed should be set at 24.7 tons per year (scenario 4, table 4).

It follows that the efficient target is not a single number for any given watershed. Rather, its value will be dependent on the abatement options available to farmers, and on the policies targeted at farmers' adoption of those options. Understanding efficient pollution targets in this manner has important implications for policy making. Instead of imposing exogenous environmental targets, policy makers can consider the costs of environmental damages and the abatement options available to the polluters. The targets can be set whereby the total costs—i.e., the sum of abatement and damage costs—are minimized.

In relation to point versus nonpoint source abatement, the results suggest that significant phosphorus abatement at the point source would be optimal in this watershed, especially considering the longer time required for agricultural land use changes. The optimality of point source abatement is accentuated by the uncertainty surrounding the estimates of pollution emissions from agricultural nonpoint sources. The results do indicate that the optimal amount of point source abatement is inversely related to the number of abatement options available to nonpoint sources.

The process of determining the standards and targets for water quality at the watershed level often lacks economic input. This investigation sheds light on how economics can assist policy makers in choosing watershed policies that maximize social welfare at a time when water quality impairments are at the forefront of the national environmental debate.

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Appendix: Results from Estimation of Environmental Damages as a Loss of Recreational Values

Table A1. Results from Estimation of the Demand Equation for Recreation in the Eucha and Spavinaw State Parks [dependent variable = price of recreation (*Travel Cost*)]

Effect [a]	Year	P Concentration Level	Estimate	Standard Error	DF	t-Value	Pr > | t |
Q			-0.0016	0.000079	35	-19.85	< 0.0001
INT^1	1990	0.037675	43.1634	1.4812	35	29.14	< 0.0001
INT^2	1991	0.038232	42.4313	1.4706	35	28.85	< 0.0001
INT^3	1992	0.038719	41.8975	1.4634	35	28.63	< 0.0001
INT^4	1993	0.039133	41.8838	1.4633	35	28.62	< 0.0001
INT^5	1994	0.039477	42.4304	1.4706	35	28.85	< 0.0001
INT^6	1995	0.039749	41.3470	1.4565	35	28.39	< 0.0001
INT^7	1996	0.039887	39.0826	1.4333	35	27.27	< 0.0001
INT^8	1997	0.039950	42.3921	1.4701	35	28.84	< 0.0001
INT^9	1998	0.040042	39.6035	1.4379	35	27.54	< 0.0001
INT^{10}	1999	0.040080	41.7904	1.4621	35	28.58	< 0.0001
INT^{11}	2000	0.040126	41.7886	1.4620	35	28.58	< 0.0001
INT^{12}	2001	0.040139	41.4425	1.4577	35	28.43	< 0.0001

[a] Q = number of visits per 1,000 population (slope parameter estimated); INT^m = intercept parameters estimated.

Table A2. Estimated Maximum WTP, Consumer Surplus (CS), and Change in Consumer Surplus (relative to 46,000 kg/year) from Each Iso-Travel Cost Region

Average P Loading – Zmax (kg/year)	Calculated Intercept on the Recreation Demand Function [a]	– REGION 1 – Tulsa Metro		– REGION 2 – Siloam Springs and Fayetteville, Ark.		– REGION 3 – Other Okla.		– REGION 4 – Local	
		CS	ΔCS	CS	ΔCS	CS	ΔCS	CS	ΔCS
18,000	55.02	33,251	33,251	109,209	104,511	127,618	118,534	363,145	247,076
20,000	53.97	26,778	26,778	97,193	92,495	114,601	105,517	340,947	224,877
25,000	51.35	13,658	13,658	70,218	65,520	85,121	76,037	288,513	172,444
30,000	48.73	4,913	4,913	47,617	42,919	60,016	50,932	240,454	124,385
35,000	46.11	544	544	29,392	24,694	39,287	30,203	196,771	80,702
40,000	43.49	0	0	15,542	10,844	22,933	13,849	157,463	41,394
46,000	40.34	0	0	4,698	0	9,084	0	116,069	0

[a] Calculation based on text equation (8).