Optimal Spatial-Dynamic Management of Groundwater Conservation and Surface Water Quality with On-Farm Reservoirs

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Abstract:

We examine the joint management of groundwater quantity and surface water quality using on-farm reservoirs with a spatial-dynamic model of farm profit maximization in the Arkansas Delta. Several policies for alleviating groundwater depletion and enhancing water quality are compared to find which strategies are cost-efficient for the conservation goals.

Key Words: Conjunctive surface water quality and groundwater management, On-farm reservoirs, Spatial-dynamic optimization

Introduction

Existing economic studies related to groundwater management and the quality and quantity of water focus on the quality of groundwater as an externality that affects the production of crops

(Roseta-Palma, 2002; Knapp and Baerenklau, 2006). A broader societal concern is how the contamination of surface water lowers the value of water to the public for recreation and other surface water uses. The social value of water depends as much on the quantity available for farming as the quality for the public, and both aspects should be considered simultaneously for adequate management. Another reason for joint management is that groundwater pumping and the amount of water applied to the surface depends on the types of crops grown, and this governs the contaminated runoff that reaches water bodies.

The focus of this paper is on evaluating the empirical significance of joint management of groundwater and surface water quality on agricultural landscapes when maximizing farm profits and social net returns. One way to do this is by considering on-farm reservoirs with tail-water recovery that capture runoff leaving the field to provide irrigation later in the season and to reduce the pollutants that leave the farm. Also, model outcomes change when the planner's objective includes a range of social values for surface water quality and groundwater quantity. Policy instruments to independently lower either groundwater withdrawals or non-point agricultural pollution are compared for their ability to achieve both higher groundwater levels and improved water quality.

The application of this model is the farming region of the Arkansas Delta which had more than four million acres of irrigated cropland in 2007, principally based on groundwater pumping that has significantly depleted the Alluvial aquifer (Schaible and Aillery, 2012). Spatial variation is incorporated into the model for the saturated thickness of the aquifer, groundwater flow in the aquifer, the contaminated surface water flow downstream, the yield of crops, and the costs of groundwater pumping. Spatial groundwater flow occurs between sites in response to the distance from cones of depression formed by the well pumping. Brozovic et al. (2010) show the underground flow of groundwater from pumping influences the spatial-dynamic decisions of optimal groundwater management. The water quality model determines pollutant loading by calculating the contaminated water leaving each site and routing this downstream where some of the pollutant may be filtered or additional pollutant added. The planner's decision about where to place reservoirs and the type of crops grown is influenced by the farm profits and the pollutant loadings associated with each site based on the water quality model.

There is an apparent gap in the literature related to the relationship between groundwater management for agricultural production and surface water quality. Studies of groundwater management to curtail withdrawals have explored cost-share assistance for irrigation technologies (Huffaker and Whittlesey, 1995), incentive payments to convert irrigated crop production to dry land crop production (Ding and Peterson, 2012), and tradable quota of groundwater stock (Provencher and Burt, 1994), but no research considers the effects of these policies on surface water quality. The empirical literature on the policies to control non-point pollution focuses on questions about whom to target, what to target, and what instrument to use (Shortle and Horan, 2001). Studies support the conclusion that the choice of the base (e.g. farm input related to the pollution) for the policy influences the cost-effectiveness of the agri-environmental policy (Weinberg and Kling, 1996). Helfand and House (1995) and Larson et al. (1996) find policies based on irrigation water use to be more cost-effective than policies based on nitrogen use because irrigation water is more highly correlated with nitrate leaching. The choice of the instrument and the instrument base to cost-effectively improve multiple environmental benefits is not investigated.

A small set of empirical analyses consider investment in reservoirs to sustain the groundwater resource or improve surface water quality. The thickness of the Alluvial aquifer must fall to below 30 feet before a reservoir is needed, and the optimal reservoir size depends on crop productivity and groundwater decline rate (Wailes et al., 2004). Hill et al. (2006) examine how cost-share for reservoir construction and surface water diversion may affect farm income and water use in the Grand Prairie of Arkansas. Popp et al. (2003) show building on-farm reservoirs once groundwater is sufficiently depleted may reduce soil-loss by more than 80 percent. However, none of the studies considers both water quantity and quality policies and optimal management across an agricultural landscape.

Even beyond what they contribute toward filtering sediment and nutrients from runoff, on-farm reservoir technology in Arkansas is positively linked to improved quality because the greater availability of water from reservoirs maintains more rice fields, and the rice fields filter more pollutants than other crops. The expectation then is that polices targeting either groundwater overdraft or surface water quality yield a larger aquifer and higher quality than the outcome when no policy is in place. Indeed, the findings of previous studies suggest that although policies designed for groundwater conservation are most cost-effective for raising groundwater levels, they can be cost-effective as well at reducing the loadings of nonpoint pollutants at the mouth of a watershed.

We describe the models for the spatial-dynamics of land, irrigation, and water quality and the types of management problems faced by the central (optimal) planner in the next section. Data on the groundwater and reservoir use, farm production, and the water quality and groundwater values for the model are presented in the third section. Section four discusses the results and the finding of the sensitivity analyses. We conclude by summarizing major findings and their relationship to prior work in a similar vein, and consider future research needs.

Methods

Spatial-dynamics of the crop types grown that influence water quality in the farm production region of the Delta depend on the supply of water in the underlying aquifer. The model uses a grid of *m* cells (sites) to represent spatially symmetric cones of depression from groundwater pumping and to track the spatially dependent loadings of nutrients (i.e. phosphorus and nitrogen[1]) and sediment to streams. The available groundwater is based on the pumping decisions of farms in and around the site weighted by distance. The pollutant loadings from runoff depend on the crops grown, the slope, soil type, and the surrounding vegetation types of each site as well as the proximity of the site to a stream.

Spatial-dynamics of land

We track the cumulative amount of land in use j for n land types for each of the major crops in the region (irrigated corn, cotton, rice, irrigated soybean and non-irrigated soybean) at the end of period t with $L_{ij}(t)$ site i. We assume land (in acres) can be converted to on-farm reservoirs $FR_{ij}(t)$ from an existing land use j during period t, and the cumulative amount of land converted to reservoirs at the end of period t is $R_i(t)$. Farmers can choose to switch land out of rice, corn, and cotton into irrigated soybeans in response to a growing water shortage, or land out of dry land soybeans into irrigated soybeans for the higher yield, and this is tracked with the variable

[1] Excessive phosphorus causes algal blooms that elevate toxins and bacterial growth that make people sick. The algae also reduce oxygen levels in streams, and this kills fish. Nitrogen forms the nitrogen-based compound called nitrate that is harmful in drinking water because of groundwater contamination. Nitrogen also contributes to the hypoxic dead zone where the Mississippi River enters the Gulf of Mexico.

$IS_{ij}(t)$. Due to the recent increase in the demand for corn for biofuel, farmers can switch land out of irrigated and dry land soybeans, cotton, and rice into corn, which is tracked with the variable $C_{ij}(t)$. The declining groundwater availability may lead farmers to switch land out of irrigated crops into non-irrigated soybean, and the variable tracking the land switching to non-irrigated soybean is $DS_{ij}(t)$. A switch to cotton is not considered since significant declines in the acreage of the crop in Delta have occurred over the last decade, and this trend is forecasted to continue (Adams et al. 2013). Using these definitions, we model the dynamics of land use in each site as a system of difference equations:

$$\begin{aligned}
&L_{ij}(t) = L_{ij}(t-1) - FR_{ij}(t) - DS_{ij}(t) - C_{ij}(t) - IS_{ij}(t), \text{ for } j = \text{rice, cotton} \\
&L_{ij}(t) = L_{ij}(t-1) - FR_{ij}(t) - DS_{ij}(t) - C_{ij}(t) + \sum\nolimits_{j=1}^{n} IS_{ij}(t), \text{ for } j = \text{irr. soybean} \\
&L_{ij}(t) = L_{ij}(t-1) - FR_{ij}(t) - DS_{ij}(t) - IS_{ij}(t) + \sum\nolimits_{j=1}^{n} C_{ij}(t), \text{ for } j = \text{corn} \\
&L_{ij}(t) = L_{ij}(t-1) - FR_{ij}(t) - IS_{ij}(t) - C_{ij}(t) + \sum\nolimits_{j=1}^{n} DS_{ij}(t), \text{ for } j = \text{non-irr. soybean}
\end{aligned} \tag{1}$$

$$R_i(t) = R_i(t-1) + \sum\nolimits_{j=1}^{n} FR_{ij}(t) \tag{2}$$

Each period, the amount of irrigated land in use j is reduced by the amount of land converted to on-farm reservoirs or switched into non-irrigated soybean production. For cropland in rice, the most water intensive irrigated crop, a switch to irrigated soybean and corn, less irrigation intensive crops, can also occur where the decline in rice is offset by the increase in irrigated soybeans and corn. The cumulative amount of land in non-irrigated soybean by the end of period t is the amount of land in non-irrigated soybean in earlier periods and the sum of the amount of land added to non-irrigated soybean from all land uses j less the land converted to on-farm reservoirs, irrigated soybeans, and corn during period t (Eq. 1). The cumulative amount of land in on-farm reservoirs by the end of period t is the amount of land in reservoirs in earlier periods and the sum of the amount of land added to reservoirs from all land uses j during period t

(Eq. 2). The total amount of land converted to a reservoir from land use *j* must be less than the amount of land in use *j* as of period *t*: $\sum_t FR_{ij}(t) \le L_{ij}(t)$

Spatial-dynamics of irrigation

Irrigation demand varies by crop and is given by wd_j, representing average annual irrigation needs excluding natural rainfall. The variable AQ_i (*t*) is the amount of groundwater (acre-feet) stored in the aquifer beneath site *i* at the end of the period *t*. The amount of water pumped from the ground is GW_i (*t*) during period *t*, and the amount of water pumped from the on-farm reservoirs is RW_i (*t*). The natural recharge (acre-feet) of groundwater at a site *i* from precipitation, streams, and underlying aquifers in a period is nr_i and is independent of crops grown on site *i*.

The runoff from site *i* is diverted to reservoirs through a tail-water recovery system. A reservoir, making up a small portion of acres available in site *i*, can be completely filled from the runoff collected from site *i*. A larger reservoir occupying a larger fraction of site *i* is only partly filled because the reservoir receives the same acre-feet of runoff. Hence, the acre-feet of water an acre reservoir can hold at full capacity from runoff throughout site *i* is $\omega_{\max}$. The water accumulated from rainfall into the reservoir is $\omega_{\min}$ per acre. The values for $\omega_{\max}$ and $\omega_{\min}$ are estimates because evaporation, rainfall, and the timing of rainfall during the season change by year. We define the following function for the acre-feet of water stored in an acre reservoir as $(\omega_{\max} + \omega_{\min}) - \frac{\omega_{\max}}{\sum_{j=1}^{n} L_{ij}(0)} R_i(t)$, which depends on the number acres of the reservoir $R_i(t)$ and the total acreage at site *i*, $\sum_j L_{ij}(0)$. The low-end acre-feet of water in each acre of the reservoir is $\omega_{\min}$ when the reservoir occupies the entire site *i* and only the rainfall fills the reservoir. The high-end is approximately $(\omega_{\max} + \omega_{\min})$ when the reservoir is less than an acre in size because runoff water as well as rainfall fills the reservoir.

Further, we define p_{ik} as the expected proportion of the groundwater in the aquifer that flows underground out of site *i* into the aquifer of site *k* when an acre-foot of groundwater is pumped out of site *k*, where p_{ik} is a negative quadratic function of the distance and the saturated thickness between sites *i* and *k*. The amount of water leaving site *i* is then $\sum_{k=1}^{m} p_{ik} GW_k(t)$. The cost of pumping an acre-foot of groundwater to the surface at site *i* during period *t* is $GC_i(t)$. Pumping costs depend on the cost to lift one acre-foot of water by one foot using a pump, c^p, the initial depth to the groundwater within the aquifer, dp_i, and the capital cost per acre-foot of constructing and maintaining the well, c^c. Note that we assume a producer drills a well deeper than the depth to the aquifer to allow for the eventual decline in the water table. Pumping costs vary by the energy needs required to lift water to the surface. The possibility of new well drilling, either at an existing well or in a new location, if the aquifer level drops below the initial drilled depth is captured in the capital cost per acre foot.

The dynamics of irrigation and pumping cost at each site is then represented by:

$$\sum_{j=1}^{n} wd_j L_{ij}(t) \le GW_i(t) + RW_i(t) \tag{3}$$

$$RW_i(t) \le \left((\omega_{\max} + \omega_{\min}) - \frac{\omega_{\max}}{\sum_{j=1}^{n} L_{ij}(0)} R_i(t) \right) R_i(t) \tag{4}$$

$$AQ_i(t) = AQ_i(t-1) - \sum_{k=1}^{m} p_{ik} GW_k(t) + nr_i \tag{5}$$

$$GC_i(t) = c^c + c^p \left(dp_i + \frac{(AQ_i(0) - AQ_i(t))}{\sum_{j=1}^{n} L_{ij}(0)} \right) \tag{6}$$

Each period, the total amount of water for irrigating crops grown at the site must be less than the water pumped from the aquifer and the reservoirs (Eq. 3), and the amount of water available from reservoirs must be less than the maximum amount of water that all the reservoirs built on the site can hold (Eq. 4). The cumulative amount of water in the aquifer by the end of period t is the amount of water in earlier periods plus the amount of recharge that occurs naturally less the amount of water pumped from the ground of surrounding sites weighted by the proximity to site i (Eq. 5). The cost of pumping an acre-foot of groundwater is c^p times the depth to the groundwater, which depends on how depleted the aquifer is under the site i, plus c^c (Eq. 6).

Spatial-dynamics of water quality

The land use dynamics influence the amounts of additional sediment and nutrients (in the form of chemical fertilizers applied to agricultural lands) and the ability of the lands to retain the pollutant en route to downstream water bodies. For example, corn grown on a farm lowers regional water quality relative to rice grown on the farm through both greater export and reduced sediment and nutrient retention. Here we focus on sediment, phosphorus, and nitrogen pollution in surface waters, which are leading causes of impairment in the Mississippi Delta (Intarapapong et al., 2002).

The InVEST (Integrated Valuation of Ecosystem Services and Tradeoffs; Tallis et al., 2011) water model is used to estimate the water quality change associated with the land transitions in the presence of declining groundwater supply. InVEST is a spatially-explicit model that applies a two-step process to determine the influence of land cover on water quality for the ten-digit hydrologic unit code (HUC) basins within a larger study area represented by three eight-digit HUC basins. First, the InVEST water yield model estimates the expected annual water yield in each 30-meter grid cell based on climate, geomorphological information, and land use characteristics. The model assumes that all precipitation not lost to evapotranspiration contributes to the surface water runoff and subsurface flows that constitute the water yield.

In the second step, the water yield is combined with expected pollutant loading and the filtering capacities for each crop type (see Table A-1) to calculate the annual pollutant exports from each cell. Based on a digital elevation model, pollutant export from each cell is routed downstream, where some of the pollutant may be filtered or additional pollutant added until this flows into a water body. This model structure makes results sensitive to the spatial pattern of land use in each basin. In particular, buffers of rice land may effectively filter pollutants before they reach a stream. Once the sediment and nutrients reach a stream the model assumes no additional retention or removal before delivery to the mouth of the watershed.

The InVEST water model calculates the pollutant loadings for the 2012 baseline. Average pollutant loadings are then assigned to each cell based on the location of the cell within the HUC basin. This is designated as $X_i(0)$ where X is any one of the pollutants: sediment, phosphorus, or nitrogen. The 2012 baseline is also used to calibrate the difference in the per-acre pollutant export when a crop type j in cell i switches to soybeans (ps_{ij}), corn (pc_{ij}), or a reservoir (pr_{ij}). The construction of a reservoir allows sediment and nutrients to be captured through tail-water recovery rather than leaving as runoff to a stream. The effectiveness of tail-water recovery to capture runoff depends on the slope of the land at cell i (measured by $0 \le \theta_i \le 1$) (A. Sharpley, University of Arkansas, personal communication) and the acreage of the reservoir built to collect the pollutant (measured by $\frac{R_{ij}(t)}{\left(R_{ij}(t)+\phi\right)}$ with the parameter ϕ used to adjust how much acreage influences effectiveness).

The dynamics of water quality at each site is then:

$$X_i(t) = X_i(t-1)\left(1-(1-\theta_i)\frac{R_{ij}(1)}{\left(R_{ij}(1)+\phi\right)}\right)+ \sum\nolimits_{j=1}^{n} ps_{ij}\left(IS_{ij}(t)+DS_{ij}(t)\right)+\sum\nolimits_{j=1}^{n} pc_{ij}C_{ij}(t)+\sum\nolimits_{j=1}^{n} pr_{ij}FR_{ij}(t) \tag{7}$$

More loadings are captured at sites that have larger reservoirs and flatter land. Pollutant exports that occur in later periods from crop transitions associated with the declining aquifer are calculated by multiplying the new land in soybeans ($IS_{ij}(t)$ and $DS_{ij}(t)$), corn $C_{ij}(t)$, and reservoirs $FR_{ij}(t)$ by the per-acre loading difference associated with the switch away from each crop type *j* (Eq. 7).

Farm net benefits objective

In the absence of available information on the location and size of individual farms under the direction of a particular farm manager and the location and size of existing wells, we make simplifying assumptions about the optimal construction of on-farm reservoirs subject to land and water use constraints. We set the size of each site *i* is comprised of $\sum_j L_{ij}(0)$ acres in field crops and the remainder in natural landscape, land for farmstead building, and public lands. The existing well capacity and pumping equipment only supports the current crop mix $L_{ij}(0)$ with ongoing payments made for this equipment. Investment in reservoirs and a tail-water recovery system includes additional pumping equipment for moving water from the tail-water recovery system into the reservoir and from the reservoir to the existing irrigation system at each site as well as annual maintenance costs. The overall objective is then to maximize the net benefits of farm production less the costs of reservoir construction and use over time.

Several economic parameters are needed to complete the formulation. The price per unit of the crop is pr_j and the cost to produce an acre of the crop excluding the water use costs is ca_j, which depend on the crop j and are constant in nominal terms. The yield of crop j per acre is y_{ij} at site i and are constant meaning no productivity growth trend. The net value per acre for crop j is then $pr_j y_j$ - ca_j excluding differential water pumping cost between well and reservoir water, and the reservoir construction costs. The discount factor to make values consistent over time is δ_t. Other costs constant in nominal terms include the annual per acre cost of constructing and maintaining a reservoir, c^r, and the cost of pumping an acre-foot of water from the tail water recovery system into the reservoir and from the reservoir to the field plus the capital cost per acre-foot of constructing and maintaining the pump, c^{rw}.

The problem is to maximize net benefits of farm production:

$$\text{(8)} \qquad \max_{FR_{ij}(t), RW_i(t), GW_i(t), L_{ij(t)}} : \sum_{t=1}^{T} \delta_t \left(\sum_{i=1}^{m} \sum_{j=1}^{n} \left(pr_j y_{ij} - ca_j \right) L_{ij}(t) - c^r FR_{ij}(t) - c^{rw} RW_i(t) - GC_i(t) GW_i(t) \right)$$

subject to:

$$\text{(9)} \qquad L_{ij}(0) = L_0^{ij}, R_i(0) = 0, AQ_i(0) = AQ_0^i,$$

$$\text{(10)} \qquad FR_{ij}(t) \geq 0, L_{ij}(t) \geq 0, AQ_i(t) \geq 0$$

and the spatial dynamics of land and water use (Eqs. 1-6). The objective (Eq. 8) is to determine $L_{ij}(t)$, $FR_{ij}(t)$, $RW_i(t)$, and $GW_i(t)$ (i.e. the number of acres of each crop, the number of acres of reservoirs, and water use) to maximize the present value of net benefits of farm production over the fixed time horizon T. The pollution levels determined by Eq. 7 result from the land and water use decisions associated with maximizing farm net benefits. Benefits accrue from crop production constrained by the water needed for the crops. Costs include the construction and maintenance of reservoirs/tailwater recovery, the capital and maintenance of the pumps, the fuel for the pumping of water from the reservoirs or ground, and all other production costs. Equation 9 represents the initial conditions of the state variables, and Equation 10 is the non-negativity constraint on land use and aquifer as well as non-reversibility on reservoir construction. We solve this problem with Generalized Algebraic Modeling System (GAMS) 23.5.1 using the non-linear programming solver CONOPT from AKRI Consulting and Development.

Water quality value objective

We augment Equation 8 objective to include water quality value from the percentage changes in phosphorus and sediment. Phosphorus is the limiting nutrient resulting in algal blooms observed in eutrophic water bodies (A. Sharpley, University of Arkansas, personal communication). Sediment lowers water quality because of the turbidity. The percentage change in the loadings of each basin is calculated by finding the difference of the phosphorus and sediment loadings associated with the crop cover change divided by the total baseline loading to the basin. Basins further downstream from where crop change occurs also experience a change in loadings. The three basins constitute a subset of all the sites, and these subsets are $I^W, I^B,$ and I^A respectively for the Lower White, Big, and the L'Anguille watersheds.

The willingness to pay (WTP) per household for a water quality improvement depends on the baseline water quality and median household income of the basin (wqv^W, wqv^B, wqv^A) and assumes the improvement in water quality is permanent. The WTP values per household are prorated to the percent change in pollutant loadings modeled by InVEST; for example, for a

WTP value of $50 per household for a 50% reduction, a 1% reduction in pollutant loading is prorated to $1. The WTP per basin is the multiplication of the prorated WTP per household and the number of households in the basin (hh^W, hh^B, hh^A).

The present value of the water quality improvement is:

(11) $$\sum_{t=1}^{T} \delta_t \left(\sum_K hh^K wqv^K \frac{\sum_{i \in I^K} \left(X_i(t) - X_i(t+1) \right)}{\sum_{i \in I^K} X_i(t)} \right), \text{ for all pollutants } X$$

where X is the phosphorus or sediment loading, and where K is the basin W, B, or A. The water quality value objective is then Equation 8 plus Equation 11 and hence net benefits accrue from farm production as well as the water quality improvements.

Buffer value objective

We augment Equation 8 objective to include the buffer value of groundwater. The buffer value of groundwater is an *in situ* value (i.e. value that occurs as a consequence of water remaining in place within the aquifer) defined as the ability of groundwater to buffer against periodic shortages in surface water supplies. Since surface water supplies can fluctuate, groundwater acts as important insurance to smooth overall supplies. The buffer value does not reflect the potentially large environmental benefits associated with an ample supply of groundwater. The percentage of groundwater value that is its buffer value is defined as p^{bv}. The total value of an acre-foot of groundwater is defined by v^{gw}. The NPV of the buffer value of the groundwater is:

(12) $$p^{bv} v^{gw} \sum_{t=1}^{T} \delta_t \sum_{i=1}^{m} AQ_i(t)$$

The buffer value objective is then Equation 8 plus Equation 12 and hence net benefits accrue from farm production as well as the stock of groundwater.

Policy options

Several policy options for groundwater conservation and water quality improvements are considered that include cost share for reservoir construction by modifying c^r, subsidizing reservoir pumping cost c^{rw}, taxing groundwater pumping cost GC, setting a total maximum annual load for phosphorus and sediment, and taxing the phosphorus and sediment loadings at the mouth of each basin. The cost share for irrigation reservoir construction ranges from 30% to 65% based on the rates from the Natural Resource Conservation Service's (NRCS) Agricultural Water Enhancement Program (NRCS, 2012). We choose a subsidy on reservoir pumping costs (20% or 40%) and tax on groundwater pumping costs (15% or 30%) to achieve groundwater conservation similar to the cost share on reservoir construction. The low- and high-end for the total maximum annual load are chosen as the 2042 phosphorus and sediment exports from the simulations using the low- and high-end water quality objectives, respectively. The low- and high-end tax on phosphorus is \$250 and \$1000 per ton based on a European taxes and fines on phosphorus in the range of \$750 per ton (Bomans et al., 2005). Since no taxes or fines on sediment have been found in existing regulations, we choose a tax on sediment that is one-fifth of the tax on phosphorus. This tax is large enough to induce crop management changes while also reflective of the fact that a ton of sediment is less detrimental to water quality than a ton of phosphorus.

Sensitivity Analyses

To evaluate the impact of tail water recovery/reservoir systems, the changes in water quality from crop type changes, and the depletion of the groundwater, model outcomes at different times are compared to the initial crop acreage allocation for 2012. Model runs were performed by i) allowing the building of reservoirs or not by setting $\sum FR_{ij} = 0$; ii) adding a water quality value to the objective function; iii) adjusting the effectiveness of reservoirs at capturing pollutant runoff by multiplying θ_i by 0.8 and 1.2, respectively; iv) adding a buffer value to the objective function; and v) evaluating policy options for groundwater conservation and water quality improvements.

Data

The study area has three eight-digit HUC watersheds (L'Anguille, Big, and the Lower White)[2] that represent the region of the Arkansas Delta where unsustainable groundwater use and impaired water quality is occuring (Fig. 1a). The watersheds overlap eleven Arkansas counties: Arkansas, Craighead, Cross, Desha, Lee, Monroe, Phillips, Poinsett, Prairie, St. Francis, and Woodruff. The study area is divided into 2,875 sites to evaluate how farmers make decisions about crop allocation and water use in a spatially differentiated landscape. The 2010 Cropland Data Layer (Johnson and Mueller, 2010) determines the initial acreage of corn, cotton, rice, and soybeans in each cell (Table 1), and the irrigated vs. non-irrigated soybean acreage is allocated on the basis of harvested acreage for 2010-2011 (NASS, 2012). The discount rate of 5% chosen for the analysis corresponds to the average yield of the 30yr Treasury Bond over the last decade (US Department of the Treasury, 2012). County crop yield information for the past 5 years is

[2] The HUCs for L'Anguille, Big, and the Lower White are 08020205, 08020304, and 08020303.

used as a proxy for yields of each of the crops and not adjusted over time (Division of Agriculture, 2012).

Groundwater use

The depth to the water table (from surface to the top of the water table) and initial saturated thickness (height of aquifer) of the Alluvial aquifer shown in Table 1 come from the Arkansas Natural Resources Commission (ANRC, 2012a). A thinner aquifer indicates greater depletion of the aquifer has occurred in that area (Fig. 1b). The size of the aquifer at site i is computed as the acreage, $\sum_j L_{ij}(0)$, times the saturated thickness of the aquifer. The natural recharge (nr_i) of the Alluvial aquifer is based on a calibrated model of recharge for the period 1994 to 1998 associated with precipitation, flow to or from streams, and groundwater flow to or from the underlying Sparta aquifer (Reed, 2003). Note that producers do not have access to the Sparta aquifer in this analysis because the greater depth to the Sparta aquifer makes the pumping from the Sparta prohibitively expensive, and there is controversy about compromising its use for drinking water.

Pumping of the groundwater reduces the size of the aquifer for the grid cell with the pumped well and for the cells that surround the well. After pumping, some of the water in the aquifer flows from the surrounding cells into the cell with the pumped well. The size of the underground flow of water is based on the distance from the pump and the hydraulic diffusivity of the aquifer. Jenkins (1968) introduced a term that is widely applied in aquifer depletion problems called the "aquifer depletion factor" (or *ADF*) to quantify the relation between these two variables. The depletion factor for pumping at a particular location in an aquifer is defined as

(11) $$ADF = \frac{D}{d^2}$$

where d is the shortest distance between the pumped well and the nearby aquifer, and D is the hydraulic diffusivity of the aquifer. The hydraulic diffusivity is the ratio of the transmissivity and the specific yield of the unconfined Alluvial aquifer. The product of hydraulic conductivity and saturated thickness is the transmissivity (Barlow and Leake, 2012).

The *ADF* is larger, i.e. the depletion of the aquifer beneath the cell is greater, if the grid cell is closer to the pumped well and the hydraulic diffusivity is bigger. We use the *ADF* to determine the proportion of the acre-feet of water pumped from a well that reduces the aquifer beneath the surrounding cells. It is assumed that the specific yield of the aquifer and the hydraulic conductivity do not vary spatially across the grid cells, and these physical constants do not affect the proportions calculated. Instead, only the distance and saturated thickness of the surrounding cells are used for computing the p_{ik} used for the economic model.

Farm production

Table 2 indicates the costs of production by crop from the 2012 Crop Cost of Production estimates (Division of Agriculture, 2012). Variable irrigation costs regardless of water source include fuel, lube and oil, irrigation labor, and poly pipe for border irrigation plus the levee gates for the flood irrigation of rice (Hogan et al., 2007). Capital costs associated with wells, pumps, gearheads and power units are charged on a per acre-foot basis and are incurred whether reservoirs are installed or not as wells remain to cover potential reservoir shortfalls. The average water use over the course of the growing season excluding natural rainfall is a little less than an acre-foot for cotton, about an acre-foot for soybeans and corn, and more than three acre-feet for rice (Powers, 2007). Crop prices are the five year average of December futures prices for harvest time contracts for all crops (GPTC, 2012). We assume the costs of production, crop prices, and yields do not vary over time.

The cost of pumping water from the ground and/or reservoir depends on the costs of the fuel, maintenance, and capital. The capital cost of the well, pump and gearhead, and power unit is amortized (Hogan et al., 2007) and divided by the acre-feet pumped from the well to calculate a capital cost per acre-foot applied. The reservoir and tail-water recovery system capital cost also is converted to periodic payments and depends on the reservoir acreage. The fuel cost per acre-foot of water from the aquifer depends on the depth to the water table and the corresponding fuel needed to raise water. Diesel use ranges from 13 gallons of diesel per acre foot for a 100 foot well to 26 gallons of diesel per acre foot for a 200 foot well (Division of Agriculture, 2012). The diesel needed per acre-foot for pumping water to and from the reservoir is 6 gallons (Hogan et al., 2007). We use $3.77 per gallon of diesel fuel (EIA, 2012) and add 10% to fuel cost to account for oil and lube for irrigation equipment (Hogan et al., 2007).

Reservoir use and construction

Young et al. (2004) determined 440 acre feet is the maximum a reservoir can be filled using a tail-water recovery system from the average rainfall runoff on a 320 acre farm. This suggests that an acre of land can yield 16.5 acre-inches for holding at the reservoir. This is the minimum amount of water (ω_{min}) we estimate an acre of reservoir can hold without the collection of runoff from a tail-water recovery system. The use of a tail-water recovery system allows a reservoir to fill to an estimated maximum capacity of 11 acre-feet per acre over the course of a year (Smartt et al., 2002). The reservoir's capacity is one and a half times the storage height less what is lost to evaporation because runoff collected during the year refills the reservoir. The effectiveness of reservoirs at capturing nutrients and sediment (θ_i) depends on the slope of the land at each site *i* (AR Land Information Board, 2006) and prior modeling with MARORA (Popp et al., 2003).

On-farm reservoir/tail-water recovery construction and maintenance costs for various size reservoirs were estimated using MARORA (Smartt et al., 2002) for different size operations to obtain capital cost estimates. Subsequently, total system cost was regressed against acres occupied by the reservoir to determine per acre investment cost for different size reservoirs. Since a majority of the construction cost for a reservoir rests on the cost to move one cubic yard of soil, this cost was updated from $1 per cubic yard to $1.2 per cubic yard to reflect changes in fuel cost since 2002 when MARORA costs were updated last. The remainder of the investment and maintenance cost is based on estimates provided within MARORA and includes a pump for tail-water recovery and a pump for irrigation.

Note that while reservoirs already exist in the study region, we assume zero reservoirs in the baseline to highlight the potential for reservoirs. This is because of the scarcity of spatially explicit data on existing reservoirs as well as the objective to highlight how construction of surface water reservoirs for irrigation use matters for farm profitability and conservation.

Water Quality Value

We use two studies to identify household WTP for lower pollutant loadings. One study is by Johnston et al. (2005) who develop a national meta-analysis of WTP estimates of improved water quality, and the second is a WTP study by Hite et al. (2002) specific to pollutant reductions in the Mississippi Delta. Following the guidelines in Johnston and Besedin (2009) we adapted parameters in the WTP function from Johnston et al. (2005) to reflect appropriate geographic area, water body type, and mean household income. The model estimates WTP as a function of changes in water quality relative to baseline conditions, with water quality described by the Resources for the Future (RFF) water quality ladder (Vaughan, 1981). To establish baseline water quality for each HUC basin, we use the 2008 list of impaired water bodies from the Arkansas Department of Environmental Quality (ADEQ, 2012). Based on consultation with local water quality experts, a 50% reduction in pollutant loading relates to a two-point increase along the RFF water quality ladder. Combining these water quality parameters with the Johnston et al. (2005) WTP function, the estimates of annual WTP for the 50% reduction are $41.97 to $73.29 per household in 2012 constant dollars.

These results are compared to WTP values from Hite et al. (2002) who report an average value of $137.91 per household per year in 2012 constant dollars for a 50% reduction in pollutant loadings. The WTP estimates from Hite et al. (2002) are 2-3 times greater than WTP values from the Johnston et al. (2005) meta-analysis, so we use each estimate as an upper and lower end on WTP for modeled pollutant reductions. The WTP per basin is the multiplication of the household WTP and the projection of the number of households in the basin in each period (Cole, 2003).

Buffer Value

After a review of the literature on groundwater's buffer value (Tsur and Graham-Tomasi, 1991), p^{bv} was set to 38% as a midpoint of available estimates of a low- (5%) to a high-end (84%) of the value of groundwater. The total value of an acre-foot of groundwater, v^{gw}, was defined as its average pumping cost or the cost to raise water by 57 feet at a cost of $0.55 per foot.

Results

Figures 1b-d illustrate initial aquifer thickness and phosphorus and sediment exports in the three studied watersheds. On average, regions with larger saturated thickness in the aquifer are experiencing higher phosphorus and sediment exports. Thus, building on-farm reservoirs may not accomplish both conservation goals simultaneously. Table 3 summarizes crop allocations, water conditions, reservoir adoption and farm profits with and without reservoirs over time when profit maximization is the only objective. In scenarios where no reservoir construction is permitted (‘without reservoirs’) on the roughly 1.2 million acres of available cropland, nearly 46 percent (543,000 acres) of the land shifts out of rice, irrigated soybeans and non-irrigated soybeans and into irrigated corn by 2022. This reallocation from 2012 to 2022 increases annual farm net returns by $25 million, drops annual groundwater irrigation use by 436,000 acre feet, and the aquifer declines to a little less than 71 million acre feet. However annual losses of nutrients from farm practices increase substantially, nitrogen by 76% and phosphorus by 112%, while sediment increased 18%. Between 2022 and 2042 a smaller percentage of additional acreage move out of rice and irrigated soybean and into irrigated corn and non-irrigated soybean. This further reduces annual groundwater irrigation use by 70,000 acre feet, and the final aquifer level is 54.6 million acre feet. By 2042, annual sediment exports increase overall by 18% and annual phosphorus exports nearly double compared to 2012. These increases are experienced nearly uniformly in the watersheds with the exception of far lower areas of the L’Anguille and far upper reaches of the Big (Fig. 2a). Losses in revenue and higher costs of irrigation cause annual farm net returns to fall 12.5% from 2022, but the annual net returns are still greater than in 2012.

Allowing reservoir construction reduces nutrient and sediment loss, slows aquifer depletion, and improves annual farm net returns compared to the 2012 and the ‘without reservoir’ conditions.

Even though available production acres fall by 75,000 acres by 2042 to create reservoirs (many of which are placed in the Lower White watershed), annual farm net returns are higher with reservoirs because more acreage remains in profitable rice, low revenue non-irrigated soybeans are eliminated, and the costs associated with pumping water from the aquifer are greatly reduced. While groundwater levels continue to fall, groundwater levels only decrease by 8% when reservoirs are allowed compared to 31% without reservoirs. Similarly, percentage increases in annual phosphorus and nitrogen loadings are much lower with reservoirs compared to without reservoirs (Fig. 2b) because more acreage remains in rice, and sediment loadings actually decrease with reservoirs compared to 2012.

The next scenarios have water quality value in the profit objective so that social net returns are optimized (Table 4). As stated above, the social value of water quality is based on the percent change in phosphorus and sediment loadings in the three watersheds at each period. Nitrogen loadings shown in the tables affect water quality, but the lack of good damage estimates from nitrogen prevent us from including this in water quality value. The water quality value in the objective for the scenarios without reservoirs causes more land to go into non-irrigated soybeans rather than corn, to reduce phosphorus and sediment loadings compared to the pure profit scenarios; however phosphorus and sediment loadings by 2042 are still higher than the baseline.

As is found in Table 3, allowing the construction of reservoirs generates greater annual farm net returns, lowers annual nutrient and sediment exports, and reduces groundwater use more than the scenarios without reservoirs. Because of water quality value, even more land (up to 92,000 acres in 2042) goes to reservoirs leading to even less phosphorus and sediment export than in the pure profit scenarios. Without reservoirs allowed, the 30 year present value of farm net returns is estimated as $2,521 to $2,554 million, which is less than the $2,616 million with the pure profit objective. The decline in water quality (from higher nutrient and sediment exports) over time represents a cost to society. Therefore the 30 year present value of social net returns are $9 to $21milllion lower than their associated farm net returns, for low end and high end water quality values, respectively. Allowing the construction of reservoirs, however, leads to a reduction in overall sediment exports after 2012. The reduction in sediment loadings and the lessened increase of phosphorus loadings represents a benefit to society and therefore the present value of social net is approximately $1 million higher over farm net returns for both low and high end water quality values.

Two additional analyses (Table 5) have scenarios that allow reservoirs and use an objective with high-end water quality value. The first examines how changes in the efficiency (+/-20%) in the pollution capture rate of reservoirs affect crop mix, water use, nutrient and sediment loadings, as well as farm and social net returns. By 2042, a 20% lower capture efficiency increases exports of nutrients and sediments and generates social costs of $4 million. However, when pollution capture rates improve by 20%, the export of sediment and phosphorus falls, and this generates social benefits of $4 million. Before the improvement in the capture rate, farm profits are $2 million higher and social benefits are only $1 million. The efficiency improvement therefore causes a redistribution of gains as well as an increase in social net returns.

In addition to valuing water quality, social values may also include the reserves in the aquifer (for use as a buffer in case of surface water shortages). In these cases, by 2042, reservoirs cover at least 88 thousand acres; ending aquifer levels are higher with greatest percentage increases in thickness found in the upper L'Anguille and some parts of the Lower White (Fig. 3). Nutrient and sediment exports are lower than in other scenarios with average decreases in exports occurring primarily in the Big and lower regions of the L'Anguille. In the low-end buffer value scenario, the social benefits reach $240 million, a result of the valuation of the entire aquifer in the social objective. When high-end buffer values are considered, land in reservoirs greatly increase to 117,000 acres and provide 1.25M acre feet of water. This reduces annual draws (by 85% compared to 2012) on the aquifer to 269,000 acre-feet and helps the aquifer recharge nearly 10% (79.6M to 87.4M acre-feet) by 2042.

Table 6 shows the results of the pure profit motive change under conservation policies that: 1) subsidize the cost of reservoir construction or reservoir water pumping, 2) tax ground water use or phosphorus and sediment loadings, or 3) set maximum allowable annual loads for phosphorus and sediment. Compared to 2012, farm net returns increase, nutrient and sediment exports decrease, and aquifer water levels are greater when policies are enacted to subsidize construction or pumping costs. As expected, the greater the respective subsidy, the greater the improvement in farm net returns but also the greater the losses in government revenue. Subsidies on pumping improve farm net returns more than those for reservoir construction, but they also led to greater losses in government revenue. Policies that tax groundwater use or pollutant loadings are effective in reducing exports and water withdrawals compared to 2012 levels. These taxes decrease net farm income and increase government revenue compared to the baseline; the higher the tax rate, the greater the changes in farm income and government revenue. Policies that set total maximum annual loads are effective at reducing nutrient and sediment loads, as well as modestly increasing the aquifer level, compared to 2012 but they also reduce net farm income without generating any government revenue.

The policy cost is the sum of the changes in net farm income and government revenue with and without the policy (Table 7). All policies generate a real cost to society ranging from $5 to $25 million. Any gains in farm net returns created through a policy are outweighed by losses in government revenue. Similarly, any gains in government revenue are outweighed by losses in farm net returns. However, in choosing the most appropriate policies, policy efficiencies can be considered. Policy efficiency can be evaluated through an examination of what it costs to achieve one unit of desired conservation. In general, no one policy is the most cost effective at achieving groundwater, phosphorus, and sediment conservation concurrently. Taxes on groundwater for example are the most cost efficient at achieving ground water conservation but highly inefficient at conserving phosphorus and sediment. Setting total maximum daily loads reduces phosphorus and sediment losses in a cost efficient manner but was a relatively inefficient policy means to address groundwater conservation. These results suggest a suite of policies may be necessary to address conservation issues if cost efficiency is desirable.

Conclusion

The results suggest that the joint management of groundwater and surface water with on-farm reservoirs can increase social benefits. Currently, the focus for building reservoirs with tail-water recovery is to conserve groundwater resources; however, with proper management of the reservoirs, there is the opportunity to significantly improve water quality across the agricultural landscape. When only pure profit motive is considered, the use of reservoirs allows farm profits to rise, final aquifer levels to increase, and pollutant loadings to decrease. Thus even without the value of water quality or groundwater in the planner's objective, the construction of reservoirs both increase farm profit and enhance conservation.

By comparing an objective with low-end water quality value with the pure profits when no reservoirs are allowed on the landscape, farm profits are less, and there is a small improvement in aquifer levels and water quality. The construction of reservoirs on the other hand increases farm profits and significantly improves aquifer levels and water quality. This suggests the focus of conservation groups should be more on technology adoption than advocacy of environmental values.

Adopting reservoirs on a farm landscape mitigates the decline in farm profits and enhances water quality more when environmental values are in the objective.

There is an alignment of conservation goals for surface water quality and groundwater conservation when the environmental value of either is used by the planner. More collaboration is thus needed among traditionally separate organizations for natural resources such as the U.S. Geologic Survey and environmental quality such as the Environmental Protection Agency.

Policies to achieve conservation goals can be evaluated based on how much redistribution of income with the government occurs and how cost effectively the policy addresses each conservation goal. At the low-end of policy intervention, the subsidy of the reservoir pumping is very cost effective for every conservation goal, although a large redistribution occurs from tax payers to the government. At greater levels of policy intervention, the subsidy on reservoir construction handles all the goals cost effectively with a modest redistribution of income; however, a tax on groundwater use is more cost-effective for groundwater conservation, and the total maximum annual load is more cost-effective for water quality improvement. Several policies then may be preferable for cost-efficiency, although research is needed to understand the effect of multiple policies with overlapping outcomes on the farm landscape.

The model for groundwater and water quality dynamics and control can be extended and further refined. Other strategies to increase aquifer levels may include the adoption of sprinkler systems like center pivot or improving the efficiency of furrow irrigation with surge valves or multiple/side inlet. Surface water quality can be improved by widening riparian buffers and maintaining highly erodible land in accordance with NRCS standards (ANRC, 2012b). Each of these strategies for conservation has their own advantages, but reservoirs with tail-water recovery are particularly well-suited to jointly manage groundwater and water quality. Other ecosystem services from the farm landscape can be considered in addition to water conservation. Some of these services include the maintenance of site productivity, regulation of insect pests, climate change mitigation, soil conservation, aesthetics, and socioeconomic values (Zhang et al. 2007). Crop types differ in the contribution to soil organic matter which enhances soil fertility and their resilience to insect invaders. Tourism in the Delta is heavily influenced by flooded rice fields that attract millions of migratory waterfowl to stop and winter.

An important caveat to our model framework is the assumption of a central planner. This is based on the seminal paper by Gisser and Sanchez (1980) that shows competitive pumping differs only slightly from optimal pumping in an application to the Pecos Basin in New Mexico. However, later papers indicate that the inefficiency from competitive pumping could be large because there are several externalities inherent in groundwater use. Negri (1989) shows there is a strategic externality to pumping, and Provencher and Burt (1993) reveal a risk externality to pumping when farmers are risk averse and there is uncertain revenue. Accounting for these externalities will likely mean groundwater depletion occurs faster than we find in this paper, and this may lead to more reservoirs and better water quality.

Groundwater overdraft and non-point source pollution substantially increase environmental damage unless management actions are taken. The Delta has several critical groundwater areas and impaired water bodies from sediment and nutrients as designated by the Arkansas Natural Resource Commission. The model we develop to jointly manage these water issues provides insight into the optimal spatial-dynamic path of management and social values from on-farm reservoirs adoption.

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Table 1. Descriptive statistics of the model data across the sites of the study area

Variable	Definition	Mean	Std. Dev.	Sum (thousands)
$L_{i,corn}$	Initial acres of corn	18	47	52
$L_{i,rice}$	Initial acres of rice	124	116	356
$L_{i,cotton}$	Initial acres of cotton	27	79	79
$L_{i,irr\ soy}$	Initial acres of irrigated soybean	185	103	530
$L_{i,non\text{-}irr\ soy}$	Initial acres of dry land soybeans	59	63	170
$y_{i,corn}$	Annual corn yield (bushels per acre)	156	8	-
$y_{i,rice}$	Annual rice yield (cwt per acre)	69	3	-
$y_{i,cotton}$	Annual cotton yield (lbs per acre)	963	74	-
$Y_{i,irr\text{-}soy}$	Annual irrigated soybean yield (bushels per acre)	42	3	-
$y_{i,non\text{-}irr\ soy}$	Annual non-irrigated soybean yield (bushels per acre)	28	4	-
dp_i	Depth to water (feet)	57	31	-
AQ_i	Initial aquifer size (acre-feet)	27,698		79,633
nr_i	Annual natural recharge of the aquifer per acre (acre-feet)	0.001	0.04	547
$nitr_i$	Annual export of nitrogen (kilograms)	555	485	1,596,000
$phos_i$	Annual export of phosphorus (kilograms)	202	214	580,000
$sedm_i$	Annual export of sediment (tons)	20	25	57,229
θ_i	Share of pollutant runoff captured by reservoir	0.87	0.07	-

Number of sites: 2,875

Table 2. Value of model parameters.

Parameter	Definition	Value
pr_{corn}	Price of corn ($/bushel)	5.07
pr_{rice}	Price of rice ($/cwt)	14.06
pr_{cotton}	Price of cotton ($/lbs)	1.02
pr_{soy}	Price of soybeans ($/bushel)	11.56
ca_{corn}	Annual production cost of corn ($/acre)	644.7
ca_{rice}	Annual production cost of rice ($/acre)	692.3
ca_{cotton}	Annual production cost of cotton ($/acre)	759.7
$ca_{irr\ soy}$	Annual production cost of irrigated soybeans ($/acre)	354.3
$ca_{non\text{-}irr\ soy}$	Annual production cost of non-irrigated soybeans ($/acre)	299.1
wd_{corn}	Annual irrigation per acre corn (acre-feet)	1.16
wd_{rice}	Annual irrigation per acre rice (acre-feet)	3.34
wd_{cotton}	Annual irrigation per acre cotton (acre-feet)	0.84
$wd_{soybean}$	Annual irrigation per acre soybean (acre-feet)	1.00
$\omega_{\max}$	Annual maximum capacity of a one acre reservoir (acre-feet)	11
$\omega_{\min}$	Annual minimum holding of a one acre reservoir (acre-feet)	1.375
c^r	Estimated annual per acre cost of reservoir ($/acre)	96.7*
c^{rw}	Cost to re-lift an acre-foot to and from the reservoir ($/acre-foot)	22.62
c^p	Cost to raise an acre-foot of water by one foot ($/foot)	0.55
δ_t	Discount factor	0.95
bv_{low}, bv_{high}	Low- and high-end buffer value of groundwater ($/acre-foot, low-end)	1.56, 12.01
wqv^W_{low}, wqv^B_{low}, wqv^A_{low}	Low-end willingness to pay for a 50% reduction in phosphorus or sediment at the mouth of the Lower White, Big Creek, and L'Anguille HUC08 watersheds, respectively ($/household)	41.9, 53.5, 73.3
wqv^W_{high}, wqv^B_{high}, wqv^A_{high}	High-end willingness to pay for a 50% reduction in phosphorus or sediment at the mouth of the Lower White, Big Creek, and L'Anguille HUC08 watersheds, respectively ($/household)	98.1, 137.9, 172.8
hh^W, hh^B, hh^A	Estimates of the number of households in 2032 in the White, Big Creek, and L'Anguille HUC08 watersheds, respectively	27362, 33716, 66799

* This is the amortized cost to construct an additional acre of reservoir. The first acre of the reservoir constructed is more expensive, and the last acre of reservoir constructed is less expensive.

Table 3. Initial, 2022, and 2042 crop allocations, water conditions, reservoir adoption, and farm profits with and without reservoirs. The objective includes no buffer value for the groundwater and no water quality value.

Crop and water conditions	Initial, 2012	Without reservoirs		With reservoirs	
		2022	2042	2022	2042
Rice (thousand acres)	356	81	59	169	166
Irrigated corn (thousand acres)	52	595	604	515	516
Irrigated cotton (thousand acres)	79	79	78	79	79
Irrigated soybeans (thousand acres)	530	382	378	353	351
Non-irrigated soybeans (thousand acres)	170	50	68	0	0
Reservoirs (thousand acres)	0	0	0	71	75
Annual reservoir water use (thousand acre-feet)	0	0	0	797	833
Annual groundwater use (thousand acre-feet)	1,846	1,410	1,340	768	726
Aquifer (thousand acre-feet)	79,633	70,896	54,624	77,133	73,057
Annual phosphorus exports (tons)	580	1,017	1,036	737	738
Annual nitrogen exports (tons)	1,596	3,390	3,463	2,458	2,429
Annual sediment exports (tons)	57,229	67,296	67,631	45,830	45,773
Annual farm net returns (millions in 2012$)[1]	111	136	119	150	146
30yr PV farm net return (millions in 2012$)[1]	--	2,616		2,959	

[1] The groundwater buffer value of the aquifer and the water quality value are not counted in the farm net returns.

Table 4. Initial and 2042 crop allocations, water conditions, reservoir adoption, and net returns with and without reservoirs for low-end and high-end water quality value. The objective includes no buffer value for groundwater.

Crop and water conditions	**Initial, 2012**	**2042 Without reservoirs**		**2042 With reservoirs**	
		Low-end water quality value	**High-end water quality value**	**Low-end water quality value**	**High-end water quality value**
Rice (thousand acres)	356	54	57	167	172
Irrigated corn (thousand acres)	52	537	551	511	500
Irrigated cotton (thousand acres)	79	75	76	78	79
Irrigated soybeans (thousand acres)	530	375	380	349	344
Non-irrigated soybeans (thousand acres)	170	146	123	0	0
Reservoirs (thousand acres)	0	0	0	82	92
Annual reservoir water use (thousand acre-feet)	0	0	0	903	988
Annual groundwater use (thousand acre-feet)	1,846	1,241	1,278	656	570
Aquifer (thousand acre-feet)	79,633	56,107	55,591	73,970	74,980
Annual phosphorus exports (tons)	580	989	993	689	661
Annual nitrogen exports (tons)	1,596	3,229	3,237	2,288	2,182
Annual sediment exports (tons)	57,229	66,531	66,437	43,256	41,614
Annual farm net returns (millions in 2012$)[1]	111	107	111	145	144
30yr PV farm net return (millions in 2012$)[1]	--	2,521	2,554	2,954	2,946
30yr PV social net returns (millions in 2012$)	--	2,512	2,533	2,955	2,947

[1] The groundwater buffer value of the aquifer and the water quality value are not counted in the farm net returns.

Table 5. Initial and 2042 crop allocations, water conditions, reservoir adoption, and net returns for sensitivity analyses on pollutant capture by reservoirs and groundwater buffer value. All models allow on-farm reservoirs and use an objective with high-end water quality value.

Crop and water conditions	**Initial, 2012**	**Pollutant capture by reservoir[2]**		**Groundwater buffer value**	
		Low-end capture	**High-end capture**	**Low-end buffer value**	**High-end buffer value**
Rice (thousand acres)	356	170	173	176	168
Irrigated corn (thousand acres)	52	502	499	497	491
Irrigated cotton (thousand acres)	79	78	79	78	77
Irrigated soybeans (thousand acres)	530	341	343	347	334
Non-irrigated soybeans (thousand acres)	170	0	0	1	0
Reservoirs (thousand acres)	0	96	93	88	117
Annual reservoir water use (thousand acre-feet)	0	1,019	1,011	962	1,254
Annual groundwater use (thousand acre-feet)	1,846	525	546	615	269
Aquifer (thousand acre-feet)	79,633	76,025	75,391	75,732	87,448
Annual phosphorus exports (tons)	580	701	616	648	524
Annual nitrogen exports (tons)	1,596	2,302	2,071	2,151	1,856
Annual sediment exports (tons)	57,229	46,132	37,199	40,514	32,289
Annual farm net returns (millions in 2012$)[1]	111	143	143	145	142
30yr PV farm net return (millions in 2012$)[1]	--	2,937	2,944	2,947	2,850
30yr PV social net returns (millions in 2012$)	--	2,933	2,948	3,187	4,861

[1] The groundwater buffer value of the aquifer and the water quality value are not counted in the farm net returns. [2] The low- and high-end of the pollutant capture by reservoir is determined by multiplying [illegible] by 0.8 and 1.2, respectively.

Table 6. Water conservation policies influence on reservoir adoption, water use, water quality, and private returns. All models allow on-farm reservoirs, and the objective includes no buffer value for the groundwater and no water quality value.

Policy	Level	Reservoir acres (thousand) 2022	Reservoir acres (thousand) 2042	Phosphorus exports (tons) 2042	Sediment exports (tons) 2042	Aquifer (thousand acre-feet) 2042	Farm net returns[1] ($ millions) 30yr NPV	Government revenue ($ millions) 30yr NPV
Baseline	--	71	75	738	45,773	73,057	2,959	0
Cost share reservoir construction costs[2]	30%	81	95	692	44,022	75,821	2,999	-50
	65%	92	97	641	41,526	76,641	3,062	-117
Subsidize reservoir pumping costs[3]	20%	86	88	667	42,276	75,537	3,039	-87
	40%	104	105	585	39,166	78,157	3,123	-203
Tax on groundwater use[3]	15%	81	84	708	43,459	76,559	2,912	40
	30%	89	90	686	41,755	79,184	2,877	63
Total maximum annual load[4]	Low end	74	82	689	43,243	74,039	2,954	0
	High end	75	81	661	41,610	73,850	2,954	0
Tax on pollutant loadings[5]	Low end	70	103	708	44,380	76,459	2,929	5
	High end	74	86	680	40,487	74,591	2,920	18

[1] The farm net returns include the payments to or receipts from the government because of the policy. [2] The cost share for irrigation reservoir construction ranges from 30% to 65% based on the rates from the Natural Resource Conservation Service's (NRCS) Agricultural Water Enhancement Program (NRCS 2012). [3] We choose a subsidy on reservoir pumping costs (20% or 40%) and tax on groundwater pumping costs (15% or 30%) to achieve groundwater conservation similar to the cost share on reservoir construction. [4] The low- and high-end for the total maximum annual load are chosen as the 2042 phosphorus and sediment exports from the simulations using the low- and high-end water quality objectives, respectively. [5] The low- and high-end tax on phosphorus is $250 and $1000 per ton based on a European taxes and fines on phosphorus in the range of $750 per ton (Bomans et al. 2005). Since no taxes or fines on sediment have been found in existing regulations, we choose a tax on sediment that is one-fifth of the tax on phosphorus because this tax is large enough to induce crop management changes while also less than the tax on phosphorus because a ton of sediment is less detrimental to water quality than a ton of phosphorus.

Table 7. Policy cost and the cost of conservation for groundwater and reductions in phosphorus and sediment

Policy	**Level**	**Policy cost[1] ($ millions)**	**Groundwater conservation cost[2] ($ per acre-foot)**	**Phosphorus conservation cost ($ per kg P)**	**Sediment conservation cost ($ per kg Sediment)**
Cost share reservoir	30%	10	3.62	217	5.71
construction costs	65%	14	3.91	144	3.30
Subsidize reservoir	20%	7	2.82	99	2.00
pumping costs	40%	39	7.65	255	5.90
Tax on groundwater	15%	7	2.00	233	3.03
use	30%	19	3.10	365	4.73
Total maximum	Low end	5	5.09	102	1.98
annual load	High end	5	6.31	65	1.20
Tax on pollutant	Low end	25	7.35	833	17.95
loadings	High end	21	3.97	362	13.69

[1] This is farm net returns in the baseline less the farm net returns plus government revenue for each policy scenario.
[2] Groundwater conservation cost is calculated as the policy cost divided by the change in aquifer level between the policy option and the baseline. Similar calculations with the policy cost are used to arrive at the phosphorus and sediment conservation costs.

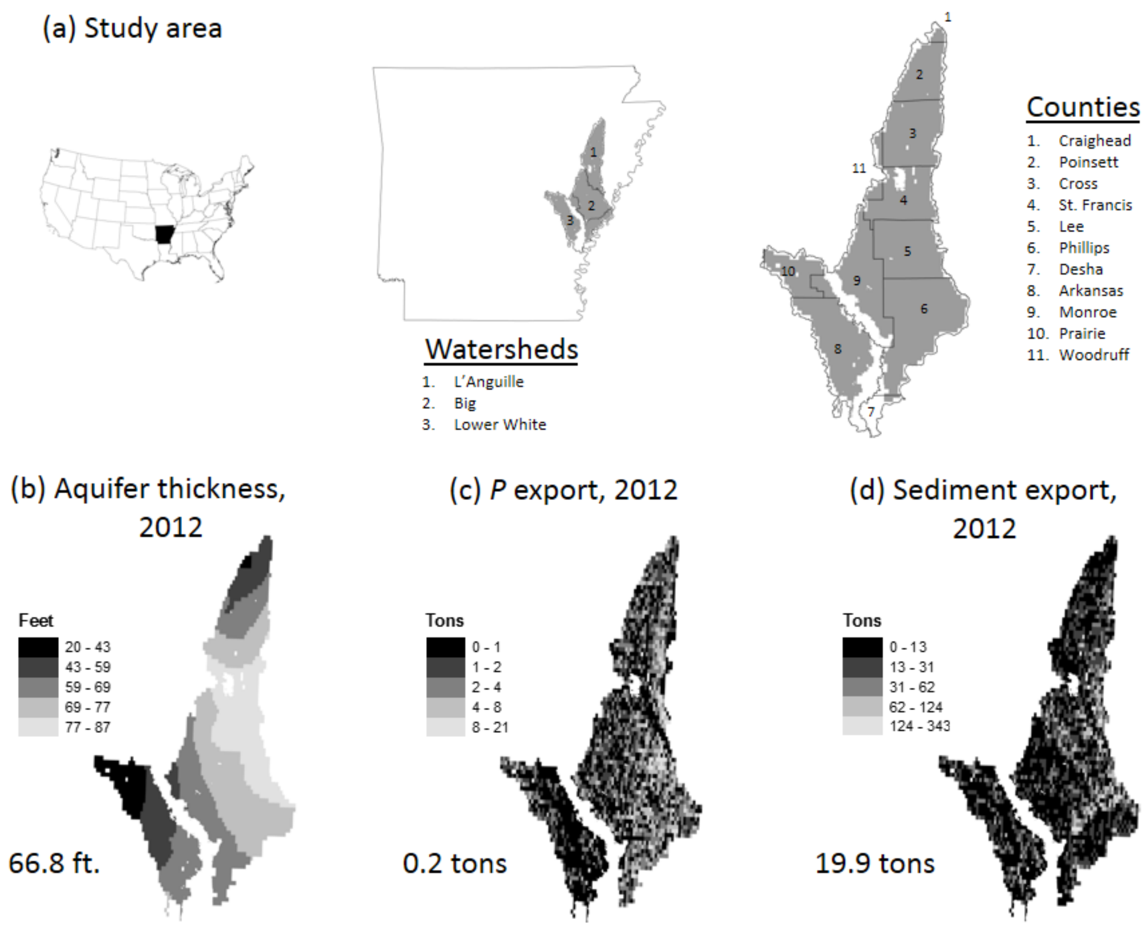

Figure 1. (a) Study area shown as grid cells. Three eight-digit HUC watersheds define the outer boundary of the study area. Public land and urban areas are excluded. Top-right map shows county lines overlay the study area. Top-left map show eight-digit HUC watershed boundaries overlay the study area. (b) Alluvial aquifer shown as feet of thickness in 2012. Lighter shades indicate the groundwater resource is more abundant. (c) Phosphorus exports shown as tons in 2012. (d) Sediment exports shown in tons in 2012. Lighter shades indicate greater exports of phosphorus and sediment. The numbers by the side of each map indicate the average.

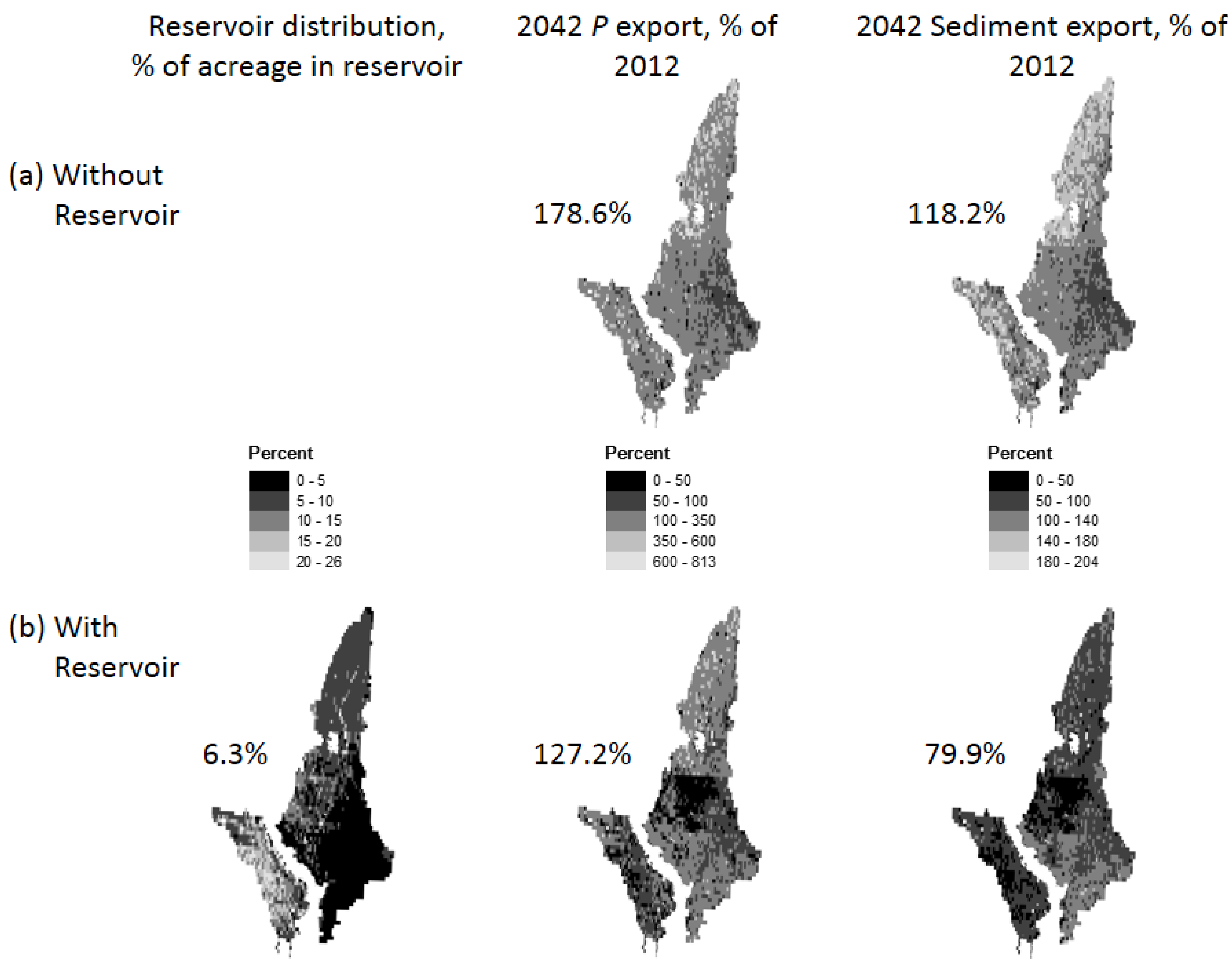

Figure 2. Reservoir locations and the change in phosphorus and sediment exports under the cases without and with reservoirs. The model runs do not have groundwater buffer value or water quality value. The numbers by the side of each map indicate study area averages.

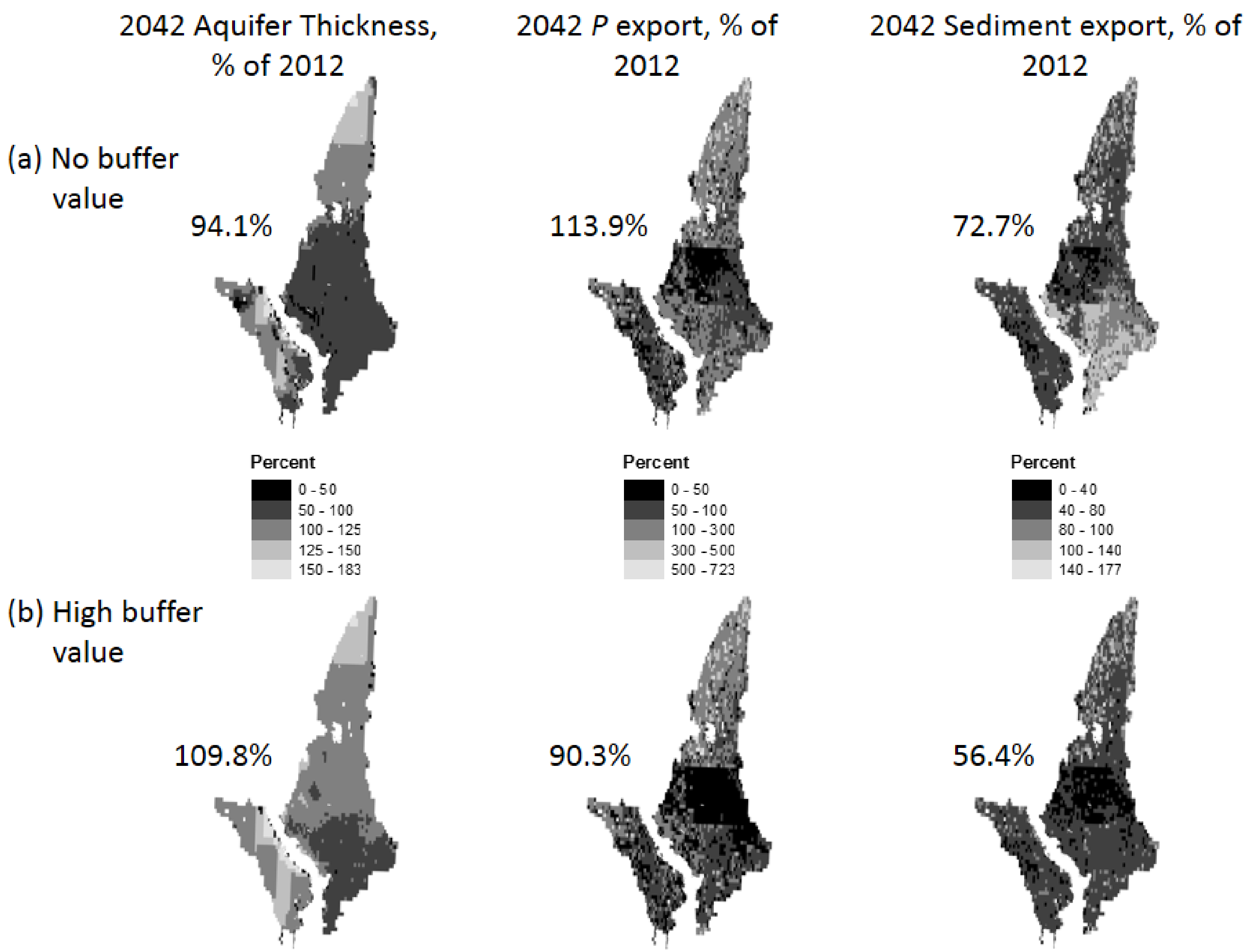

Figure 3. Aquifer decline and the change in phosphorus and sediment exports under the cases without and with groundwater buffer value. All the model runs include high-end water quality value and allow on-farm reservoirs. The numbers by the side of each map indicate study area averages.

Appendix

Water yield and nutrient runoff models

The following model descriptions are adapted from Tallis et al. (2011). For each scenario we determined water yield and total phosphorus/nitrogen loadings for ten-digit HUC watersheds within the study area of three eight-digit HUC basins. First, we model water yield, which approximates the absolute annual water yield across the basin, and is calculated as the difference between precipitation and actual evapotranspiration on each grid cell. We used maps of 30-year mean annual precipitation (Prism Climate Group, 2010)[3] and potential evapotranspiration (Ahn and Tateishi, 1994), soil depth and plant available water content (USDA-NRCS, 2013), as well as data on the coefficients of rooting depth (Schenk and Jackson, 2002) and evapotranspiration (adapted from Allen et al., 1998) for each LULC type (See Table A-1).

The water yield model is based on the Budyko curve, developed by Zhang et al. (2001), and annual average precipitation. We determine annual water yield (Y_{jx}) for each grid cell on the landscape (indexed by $i = 1,2,\ldots,I$) as follows:

$$Y_{ji} = \left(1 - \frac{AET_{ij}}{P_i}\right) \cdot P_i$$

where, AET_{ij} is the annual actual evapotranspiration on grid cell i with LULC j and P_i is the average annual precipitation on grid cell i. The evapotranspiration portion of the water balance, $\frac{AET_{ij}}{P_i}$, is an approximation of the Budyko curve (Zhang et al., 2001).

$$\frac{AET_{ij}}{P_i} = \frac{1 + \omega_i R_{ij}}{1 + \omega_i R_{ij} + \frac{1}{R_{ij}}}$$

where, R_{ij} is the Budyko Dryness index on a grid cell i with LULC j, which is the ratio of potential evapotranspiration to precipitation (Budyko, 1974). ω_i is an annualized ratio of plant accessible water storage to expected precipitation.

$$\omega_i = Z \frac{AWC_i}{P_i}$$

where, AWC_i is the volumetric plant available water content measured in mm and is estimated as the difference between field capacity and wilting point. AWC_i is defined by soil texture and effective soil depth, which establishes the amount of water capacity in the soil that is available for use by a plant. Z is the Zhang constant that presents the seasonal rainfall distribution. Finally, with R_{ij} is calculated by the following,

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[3] Unit: supplied in mm*100, and have been converted to mm; Resolution: resampled to 30m*30m cell size

$$R_{ij} = \frac{k_{ij} \cdot ETo_i}{P_i}$$

where, ETo_i is the reference evapotranspiration on grid cell x and k_{ij} is the plant evapotranspiration coefficient associated with the LULC j on pixel i. ETo_i represents an index of climatic demand while k_{ij} is largely determined by a grid cell's vegetative characteristics (Allen et al., 1998).

Second, we determine the quantity of phosphorus/nitrogen retained by each grid cell in the watershed using information on nutrient loadings based on export coefficients and filtering characteristics of each LULC (see Table A-1; Reckhow et al., 1980), the water yield output noted above, and a Digital Elevation Model (Arkansas Land Information Board, 2006). Adjusted Loading Value for grid cell i, ALV_i, is calculated by the following equation:

$$ALV_i = HSS_i \cdot pol_i$$

where, pol_i is the export coefficient at grid cell i and HSS_i is the Hydrologic Sensitivity Score for grid cell i and is calculated as:

$$HSS_i = \frac{\lambda_i}{\overline{\lambda}}$$

where, $\overline{\lambda}$ is the mean runoff index for the basin, and λ_i is the runoff index for grid cell i and is calculated by the following:

$$\lambda_i = Log\left(\sum_U Y_U\right)$$

where, $\sum_U Y_U$ is the sum water yield of all grid cells along the water flow path above and including grid cell i.

Once we determine ALV_i, we then estimate how much of the load is retained by each grid cell downstream of a neighboring cell, as surface runoff moves phosphorus/nitrogen across the landscape and towards the mouth of the watershed. Using a GIS, we model the route of surface water down flow paths as determined by the slope of a grid cell. Each grid cell downstream is allowed to retain phosphorus/nitrogen based on its land-use type. Finally, the model aggregates the phosphorus/nitrogen loading that reaches the stream from each grid cell to determine the total loading for the entire watershed.

Table A-1. Estimates for nutrient loading, evapotranspiration, rooting depth, available water capacity, and vegetation filtering.

LULC	Evapotranspiration	Rooting depth	Phosphorus loading	Phosphorus filtering	Nitrogen loading	Nitrogen filtering
Corn	1200(e)	900(c)	2210(a)	25(b)	12420(a)	50(d)
Cotton	1200(e)	1000(j)	4310(a)	25(b)	9310(a)	25(b)
Rice	1200(e)	550 (i)	450(f)	80(h)	600(f)	90(l)
Soybeans, Dbl Crop Winter Wht/Soybean	1150(e)	740(c)	1907(k)	62(k)	4712(k)	70(k)
Sorghum, Sunflower, Winter Wheat, Oats, Millet, Safflower, Other Crops, Peas, Peaches, Pecans, Squash, Dbl Crop Winter Wht/Corn, Dbl Crop Soybeans/Oats, Cabbage	600(b)	700(b)	2320(a)	62(k)	5630(a)	70(k)
Fallow/Idle Cropland	200(b)	500(b)	100(b)	50(b)	3400(b)	50(b)
Pasture/Hay	850(b)	1000(b)	100(b)	25(b)	3100(b)	25(b)
Open Water	1000(b)	1000(b)	1(b)	5(b)	1(b)	5(b)
Developed/Open Space, Developed/Low Density, Developed/Medium Density, Developed/High Density	100(b)	10(b)	500(b)	5(b)	4000(b)	5(b)
Barren	200(b)	10(b)	1(b)	5(b)	4000(b)	5(b)
Deciduous Forest, Evergreen Forest, Mixed Forest, Shrubland	1000(b)	7000(b)	35(a)	70(g)	2862(a)	80(b)
Grassland Herbaceous	650(b)	2000(b)	50(b)	60(g)	4000(b)	40(b)
Woody Wetlands, Wetlands	1000(b)	7000(b)	50(b)	80(b)	2000(b)	80(b)

Source: a) Reckhow et al., 1980; b) Tallis et al., 2011; c) Dwyer et al., 1998; d) Simpson et al., 2008; e) Allen et al., 1998; f) Manley et al., 2009; g) Zaines & Schultz, 2002; h) Moore et al., 1993; i) Mishra et al., 1997; j) Phocaides, 2007; k) USDA, 2012; l) Reddy, 1982.

Sediment Retention Model

Sediment export and retention for the Arkansas ten-digit HUC watersheds within the study area of three eight-digit HUC basins is likewise determined for each scenario. InVEST applies the Universal Soil Loss Equation (USLE) (Wischmeier and Smith, 1978) at the pixel scale to model soil loss and sediment transport across the study area. The USLE integrates information on land use patterns and soil properties, as well as a DEM, rainfall, and climate data. We determine $USLE_{ij}$ for each grid cell as follows:

$$USLE_{ij} = R_i \cdot K_i \cdot LS_i \cdot C_{ij} \cdot P_{ij}$$

where, R_i is rainfall erosivity, K_i is the soil erodibility factor, LS_i is the slope-length gradient factor, C_{ij} is the crop/vegetation and management factor, and P_{ij} is the support practice factor. The C_{ij} factor is used to determine the effectiveness of a given crop and tillage method in terms of preventing soil loss, while the P_{ij} factor reflects the effectiveness of support practices such as cross-slope cultivation relative to straight-row farming up and down slope. We use data for C_{ij} and P_{ij} factors for each LULC type obtained from the USDA NRCS Arkansas RSULE cropping management regions (See Table A-2). We use rainfall erosivity data R_i, digitized from USDA maps and published by the EPA (EPA, 2013) and soil erodibility data, K_i, obtained from the USDA SSURGO dataset (USDA-NRCS, 2013).

The Slope Length Factor is the most crucial parameter in the USLE for determining sediment export and retention. Slope length is essentially the distance that a drop of rain or sediment would flow until its energy dissipates, either through deposition or joining concentrated flow. It represents a ratio of soil loss under given conditions compared to a site with standard reference conditions. We determine LS_i for each grid cell as follows:

For low slopes:

$$LS_i = \left(\frac{(flowacc_i \cdot cellsize_i)}{22.13}\right)^{nn_i} \left(\left(\frac{sin(slope_i \cdot 0.01745)}{0.09}\right)^{1.4}\right) \cdot 1.6 nn_i = \begin{cases} 0.5, & slope_i \geq 5\%, \\ 0.4, & 3.5 < slope_i < 5\% \\ 0.3, & 1 < slope_i \leq 3.5\% \\ 0.2, & slope_i \leq 1\% \end{cases}$$

where, $flowacc_i$ is accumulated water flow to each cell and $cellsize_i$ is the pixel size or grid resolution (30m in our case).

For high slopes:

$$LS_{ij} = 0.08 \lambda_i^{0.35} pcnt\_slope_i^{0.6}$$

λ = *{cellsize, flowdir = 1, 4, 16, or 64/1.4cellsize, other flowdir}*

where, $pcnt\_slope_i$ is the pixel's percent slope and $flowdir_i$ is the flow direction of the pixel.

The model estimates the ability of the vegetation to retain sediment by comparing erosion rates on a pixel with vegetation data to erosion rates on that same pixel with no vegetation present (bare soil). The bare soil estimate is calculated as follows:

$$RKLS_i = R_i \cdot K_i \cdot LS_i$$

While erosion from the pixel with vegetation is calculated using the USLE equation:

$$USLE_{ij} = R_i \cdot K_i \cdot LS_i \cdot C_{ij} \cdot P_{ij}$$

Subtracting $USLE_{ij}$ from $RKLS_i$ calculates the amount of erosion that was avoided, or sediment retention. In addition to preventing sediment from eroding where it grows, vegetation also serves to trap sediments that have eroded upstream. We model the flow path of surface water as determined by the slope of a grid cell and estimate how much sediment eroded will be trapped downstream based on the ability of vegetation in each pixel to retain sediment. The model aggregates the sediment loading that reaches streams for each grid cell to determine the total sediment loading for the watershed.

Table A-2. Estimates for crop/vegetation and management factor, support practice factor, and sediment filtering.

LULC	Crop/vegetation and management factor	Support practice factor	Sediment filtering
Corn	130(c)	400(c)	25(a)
Cotton	170(c)	400(c)	25(a)
Rice	90(c)	400(c)	25(a)
Soybeans, Dbl Crop Winter Wht/Soybean	120(c)	400(c)	25(a)
Sorghum, Sunflower, Winter Wheat, Oats, Millet, Safflower, Other Crops, Peas, Peaches, Pecans, Squash, Dbl Crop Winter Wht/Corn, Dbl Crop Soybeans/Oats, Cabbage	170(c)	400(c)	25(a)
Fallow/Idle Cropland	8(c)	200(c)	5(a)
Pasture/Hay	20(a)	250(a)	40(a)
Open Water	1(a)	1(a)	80(a)
Developed/Open Space, Developed/Low Density, Developed/Medium Density, Developed/High Density	1(a)	1(a)	5(a)
Barren	250(a)	10(a)	20(a)
Deciduous Forest, Evergreen Forest, Mixed Forest, Shrubland	3(b)	200(b)	60(a)
Grassland Herbaceous	8(c)	200(c)	40(a)
Woody Wetlands, Herbaceous Wetlands	10(a)	200(a)	60(a)

Source: a) Tallis et al., 2011; b) Wischmeier & Smith, 1978; c) USDA-NRCS, 2004

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