Investment in on-farm reservoirs to align economic returns and ecosystem services

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Abstract:

Groundwater use plays an important role in agriculture where the lack of timely rainfall can lower yields. Policy makers strive to conserve ecosystem services, such as groundwater supply, while at the same time providing for economic growth. We develop a spatially explicit landscape level model for analyzing the ecosystem service and economic consequences of alternative crop mix patterns. The spatially explicit ecosystem service model uses initial aquifer thickness, hydro-conductivity of the aquifer, a digital elevation model, and soil characteristics, among other data, to predict the value of groundwater, surface water quality, and greenhouse gas emissions. The spatially explicit economic model incorporates site characteristics and location to predict economic returns for a variety of potential crop types. By thinking carefully about the arrangement of activities, we find crop mix and surface water storage systems that sustain high levels of ecosystem services and economic returns. Compared to the current crop mix, we show that both ecosystem service conservation and the value of economic activity could be increased substantially.

Key Words: Ecosystem services, On-farm reservoirs, Spatial-dynamic optimization

JEL classification: Q15, Q24, Q25, Q28

Introduction

Groundwater supply for agricultural production is one of a range of goods and services important for human well-being, known as ecosystem services. Numerous policies to curtail groundwater withdrawal have been proposed, including 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 quotas of groundwater stock (Provencher and Burt, 1994). Although groundwater management policies typically evaluate the consequences for economic returns, these policies should also consider the effect on the values of other ecosystem services such as surface water quality and greenhouse gas (GHG) emissions. Groundwater supply affects whether irrigation intensive crops are grown, and the type of crops grown affect how much nutrient runoff and sediment reaches water bodies. Moreover, GHG emissions depend on the crop types because crops vary in their GHG sequestration and emissions. Agricultural water management is then usefully viewed as a system and should consider the tradeoffs in economic returns and ecosystem services to determine the optimal policies for expected welfare.

One way to conserve groundwater quantity and improve surface water quality is to use on-farm reservoirs with tail-water recovery that capture the runoff leaving the field to provide irrigation later in the season and reduce pollutants that leave the farm by trapping the nutrients in the tail-water (Wailes et al., 2004). The construction of on-farm reservoirs once groundwater is sufficiently depleted may reduce soil loss by more than 80 percent (Popp et al., 2003). Farm profits can be maintained with the production of valuable irrigation-intensive crops, and the value of ecosystem services can rise from lower groundwater withdrawals and better surface water quality. However, the use of on-farm reservoirs also means tradeoffs with economic returns and GHG emissions. Valuable crop land used for on-farm reservoirs translates into a loss of farm revenue since the land cannot be used to grow crops. Also, some irrigation intensive crops such as rice that continue to be grown with the construction of on-farm reservoirs emit copious amounts of methane, a potent GHG.

Crop type decisions on farm land are based primarily on economic criteria. While crop type decisions based solely on economic returns are often detrimental to ecosystem services, securing some economic return from farm land need not be mutually exclusive with the sustainability of ecosystem services. By thinking carefully about the pattern, extent, and intensity of crop production across the landscape, it may be possible to achieve important ecosystem service objectives while also generating reasonable economic returns. The adoption of on-farm reservoirs may boost not only economic returns in the presence of groundwater scarcity but also ecosystem services or at least make the tradeoff of one for the other less severe.

We integrate spatially explicit ecosystem service and economic models to analyze the consequences of alternative crop type decisions. The ecosystem service model evaluates how groundwater supply, surface water quality, and GHG emissions are affected by a landscape given a spatially explicit pattern of crop types. The aquifer's initial thickness (amount of water available), hydro-conductivity (speed of lateral water movement), and distance to the aquifer thickness of surrounding grid cells along with irrigation needs of the crops grown predict the cost of groundwater pumping in each grid cell. The surface water quality which we measure with the level of soil, nitrogen, and phosphorous runoff depends on the land gradient and crop types grown given different tillage and irrigation practices. The GHG emissions depend on the soil texture and farm practices that vary spatially across the landscape. The ecosystem service objective is to maximize the combined values of groundwater and surface water quality, and minimize GHG emissions net of GHG sequestration which we track by summing these values over all grid cells over time.

The economic model estimates returns for each grid cell by varying crop (rice, soybeans, corn, cotton, sorghum, and wheat). Information on location, such as soil characteristics and initial depth to the aquifer affects groundwater pumping cost at each grid cell and irrigation affects yield. The groundwater pumping cost rises as the aquifer is depleted as the depth to the aquifer increases with declining ground water levels. Therefore, water conservation practices chosen influence the yield, demand for irrigation water, and production cost of the crops. The economic objective is to maximize the sum of the present value of economic returns of each grid cell given the mix of crop types produced over time.

We use results from the ecosystem service and economic models to search for efficient crop allocation using potential on-farm reservoir construction. An efficient pattern is one that generates the maximum economic returns for a given value of the ecosystem services sustained (and vice versa). By maximizing the economic returns over the entire range of possible ecosystem service values we can trace out an efficiency frontier for the landscape which demonstrates the degree of inefficiency of other crop patterns not on the frontier. The empirical application of the methodology applies to the Arkansas side of the Lower Mississippi River Basin farm production region. The current rate of pumping in the Mississippi River Valley Alluvial Aquifer is unsustainable (ANRC, 2012), and the use of on-farm reservoirs and tail water recovery systems is becoming popular to meet irrigation needs.

Background

Most prior work looking at the spatial pattern of the provision of bundles of ecosystem services describes the degree of spatial correlation given the current pattern of land use (e.g., Chan et al. 2006; Egoh et al. 2009; Raudsepp-Hearn et al. 2010) and associated bundles of services. For example, intensive agricultural production is associated with high production of agricultural products but low water quality and carbon storage while, on the other end of the spectrum, conserved forested areas, often have high carbon storage, habitat and recreation value but low commercial returns. The closest prior papers that deal with ecosystem service tradeoffs with crop land use are Nelson et al. (2009) and Polasky et al. (2008, 2011). However, none of these papers address groundwater supply in the ecosystem service objective or examine how a groundwater withdrawal policy like the cost-share of on-farm reservoir construction influences crop patterns over time.

Methods

Spatial-dynamics of the crops grown influence water quality in the farm production region and depend on the supply of water in the underlying aquifer. The time frame is a 30 year period from 2012 to 2042 chosen to observe a decline in the aquifer while staying within a farmers' planning horizon. A grid of *m* cells (sites) represents spatially symmetric cones of depression from groundwater pumping that also tracks the spatially dependent loadings of nutrients (i.e. phosphorus and nitrogen[1]) and sediment to streams. The available groundwater is based on

[1] Excessive phosphorus causes algal blooms that elevate toxins and bacterial growth that make people sick. The algae also reduce oxygen levels in streams killing fish. Nitrogen forms the nitrogen-based compound called nitrate that is harmful in drinking water. Nitrogen also contributes to the hypoxic dead zone where the Mississippi River enters the Gulf of Mexico (Carpenter et al., 1998).

initial aquifer thickness and on the pumping decisions of farms in and around the site weighted by distance (Figure 1). The pollutant loadings from runoff depend on the crops grown, the slope, soil texture, 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 crop *j* for *n* major crop types in the region (irrigated corn, cotton, rice, irrigated soybean, non-irrigated soybean, irrigated soybean grown double cropped with winter wheat, non-irrigated sorghum) and land in the Conservation Reserve Program (CRP) at the end of period *t* as denoted by L_{ij_t} for site with crop *j*. 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}. The land converted to a reservoir is used to store surface water to reduce reliance on groundwater and to store runoff to reduce non-point source pollution.

Farmers can optimally choose land allocation among various crops as well as on-farm reservoirs. They are allowed to switch their crops during different periods t, depending on the level of ecosystem services they wish to maintain and relative crop profitability as affected by the presence or absence of on-farm reservoirs and/or farm policies. For instance, farmers with declining groundwater availability may switch land out of irrigated crops into non-irrigated crops. A policy incentive targeted at minimizing net green house gas (GHG) emissions (GHG emissions – GHG sequstration) may lead to lesser acreage in rice given its high level of methane emissions. Nonetheless, no matter what crops the farmers decide to grow in the future, land use across each crop and on-farm reservoirs at any time *t* is constrained by the initial land resource availability observed in 2012.

(1) $\sum_{i=1}^{m}\sum_{j}^{n} L_{ij\_t} = \sum_{i}\sum_{j} L_{ij\_0}$, for *j*=irr.soybean, non-irr soybean, double-crop soybean, rice, cotton, non-irr sorghum, corn, on-farm reservoir and land of Conservation Reserve Program (CRP)

Spatial-dynamics of irrigation

Irrigation demand varies by crop and is given by wd_j, representing average annual irrigation needs required in addition to 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 .

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 (Eq. 2) for the acre-feet of water stored in an acre reservoir as

(2) $$(\omega_{\max} + \omega_{\min}) - \frac{\omega_{\max}}{\sum_{j=1}^{n} L_{ij}\_0} R_i\_t ,$$

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 $\left(\omega_{\max} + \omega_{\min}\right)$ when the reservoir is less than an acre in size with runoff and rainfall filling the reservoir to capacity. The model does not account for leakage and evaporative losses of water from the reservoir, and there is no tracking of within year additions and uses to the reservoir.

Typically economic papers suppose a single-cell aquifer that assumes an aquifer responds uniformly and instantly to groundwater pumping at any place in the study area (Ding and Peterson, 2012; Wang and Segarra, 2011; Wheeler et al., 2008). A limited number of papers are beginning to use a spatial aquifer to examine groundwater flow like this paper does (Brozovic, Sunding, and Zilberman, 2010; Pfeiffer and Lin, 2012). 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 hydraulic diffusivity (speed of lateral underground water movement given average soil texture and profiles observed in the region) 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. We assume the groundwater pumps are uniformly efficient with identical power units that deliver a fixed number of gallons per minute. The dynamics of irrigation and pumping cost at each site is then represented by:

$$\sum_{j=1}^{n} wd_j L_{ij}\_t \leq GW_i\_t + RW_i\_t \tag{3}$$

$$RW_i\_t \leq \left(\left(\omega_{\max} + \omega_{\min}\right) - \frac{\omega_{\max}}{\sum_{j=1}^{n} L_{ij}\_0} R_i\_t \right) R_i\_t \tag{4}$$

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

$$GC_i(t) = c^c + c^p \left(dp_i + \frac{\left(AQ_i\_0 - AQ_i\_t\right)}{\sum_{j=1}^{n} L_{ij}\_0} \right) \tag{6}$$

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).

The constraints (Equations 3 to 6) on the water availability from the ground or in the reservoirs limit the profits from the agricultural landscape. The objective of farm profitability is optimized subject to these water availability constraints.

Spatial-dynamics of greenhouse gases

The land use influences both carbon emission and carbon sequestration associated with global warming. Following Popp et al. (2011), we provide a scan-level life cycle assessment (LCA) of carbon equivalent emission and carbon sequestration for crop production on a county-level basis.

The carbon equivalent emission (E_{jt}) includes both direct and indirect GHG emissions. Direct emissions are the emissions from farm operations. For instance, carbon dioxide (CO_2) emissions from the use of diesel by tractors and irrigation equipment and the use of gasoline by farm trucks. Indirect emissions are those that come from the upstream firm which produce the input used on the farm.

We uniformly convert the multiple GHGs that lead to global warming to the carbon-equivalent (CE) emissions to obtain the C footprint.

To estimate the carbon sequestration, we first use the county level yield to derive the aboveground biomass (AGB) and the belowground biomass (BGB), and then estimate the carbon sequestrated from the aboveground biomass as well as belowground biomass across diverse soil textures using main tillage practices in Arkansas.

The aboveground biomass (AGB_{ij}) per acre for crop j in site i with tillage method q is calculated by following

(7)

$$AGB_{ij} = Y_{ij}\lambda_j(1-\alpha_j)(\frac{1}{H_j} - 1)$$

where Yij is the grain or fiber yields per acres for crop j in site i. The yields data is county-level which represents the yields across the same county are unchanged. Crop dependent yield units of measurement are converted to pounds per acre using λ_j and converted to dry matter equivalent using α_j the standard moisture content for reported yield units by crop. The harvest index, H_j, converts yield to above ground biomass by adding the weight of stems and leaves.

To convert the above ground biomass (AGB_{ij}) into kilograms of C sequestrated from above ground biomass ($CAGB_{ijq}$), one more step is needed:

(8)
$$CAGB_{ijq} = AGB_{ij} * \eta_q * \beta_j * \delta_q$$

where η_q is the proportion of plant residue incorporated in the soil depending on tillage method q, β_j is the estimated fraction of carbon in the *AGB*, and δ_q is the tillage-based fraction of carbon sequestrated in the soil.

Using the same methodology, kilograms of carbon sequestrated from the belowground biomass ($CBGB_{ijq}$) per hectare for crop j in site i under tillage method q was estimated by

(9)
$$CBGB_{ijq} = \chi_j \eta_q \left[\frac{\phi_j Y_{ij} \lambda_j (1-\alpha_j)}{\mathrm{H}_j}\right]$$

where χ_j is the fraction of carbon in the belowground biomass and ϕ_j is the shoot/root ratio.

We finally adjust both above- and belowground biomass C sequestration by an estimated soil factor, ξ_{is}, given the weighted area of soil texture in each site. The total carbon sequestration S_{ijqs}contained in both above and belowground biomass can be estimated by

(10)
$$S_{ijqs} = (CAGB_{ijq} + CBGB_{ijq})\xi_{is}$$

Spatial-dynamics of water quality

Land use dynamics influence the amounts of sediment and nutrients (in the form of chemical fertilizers applied) carried to downstream water bodies. For example, corn grown on a farm lowers regional water quality relative to rice, as corn is typically more tillage intensive and requires large nitrogen application, whereas rice is flooded, typically on flatter land and drainage of water is managed with a tail water recovery system. We focus on sediment, phosphorus, and nitrogen pollution in surface waters, which are leading causes of water quality 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 land use transitions. 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) sub-watershed within a larger study area represented by a three eight-digit HUC watershed. 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 quality model uses the entire landscape which includes public land, lakes, and residential areas. These places are excluded from the study area for the farm production problem but are included in the water quality model. The pollutants that reach the mouth of each basin from farm site *i* are therefore affected by the public land and residential areas that filter or add to the pollutant loadings in the streams. Once nutrients reach a water body the model assumes no further retention or removal (i.e. no in-stream processes) 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 runoff (measured by $\frac{R_{ij}\_t}{\left(R_{ij}\_t+1\right)}$). The use of the ratio $\frac{R_{ij}\_t}{\left(R_{ij}\_t+1\right)}$ indicates even a small reservoir, about an acre in size, can capture most of the runoff and unwanted pollutants.

The dynamics of water quality at each site is then:

$$(11)\qquad X_i\_t = X_i\_(t-1)\left(1-(1-\theta_i)\frac{R_{ij}\_t}{\left(R_{ij}\_t+1\right)}\right)+ \sum_{j=1}^{n} ps_{ij}\left(IS_{ij}\_t+DS_{ij}\_t\right)+\sum_{j=1}^{n} pc_{ij}C_{ij}\_t+\sum_{j=1}^{n} pr_{ij}FR_{ij}\_t$$

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. 11).

Equation 8 is not a constraint in the baseline model since farm profits are not influenced by water quality. Pollutant levels are tracked over time but do not influence the decisions farmers make on crop mix, water use, and reservoir construction. A later scenario includes the social value of water quality in the objective. In that scenario, the objective of social net benefits is optimized subject to the dynamics of water quality.

Model objectives

In our model, we explore the farmers' optimum land use and water extraction decision subject to two different objectives under the various scenarios. First, famers are expected to maximize their economic return by making the land and water use choice when the value of ecosystem services is neglected. Then, farmers' behavior are investigated when ecosystem services including water quality value, buffer value of groundwater, and carbon offset value are part of their intentions.

1. Economic returns 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*, comprised of $\sum_j L_{ij}\_0$ acres in field crops, and allocate the remainder to natural landscape, land for farmstead buildings, and/or 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}. We assume the stationary relift pumps are uniformly efficient with identical power units that deliver a fixed number of gallons per minute.

The problem is to maximize net benefits of farm production:

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

subject to:

(13) $$L_{ij}\_0 = L_0^{ij}, R_i\_t = 0, AQ_i\_t = AQ_0^i,$$

(14) $$FR_{ij}\_t \geq 0, L_{ij}\_t \geq 0, AQ_i\_t \geq 0$$

and the spatial dynamics of land and water use (Eqs. 1-7). The pollution levels determined by Eq. 8 result from the land and water use decisions associated with maximizing farm net benefits. The objective (Eq. 9) is to determine L_{ij}_t, FR_{ij}_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 profits of farm production over the fixed time horizon *T*. Revenue accrues from crop production constrained by the water and other inputs 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 10 represents the initial conditions of the state variables, and Equation 11 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.[2]

2. Ecosystem services objective

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[2] The problem is not linear because the groundwater pumping cost and the amount of groundwater pumped are both solved as part of the problem and are multiplied together. The CONOPT solver available in GAMS is particularly effective at solving complex non-linear programs.

Water quality value

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. Sediment lowers water quality because of the turbidity. The deterioration in water quality lowers the recreational and ecological value to the public. 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.[3] 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:

[3] A more realistic assumption is a diminishing marginal WTP for an improvement in water quality. However, there is not clear guidance in the literature as to the exact functional form that this diminishing marginal WTP would take. We therefore use the simpler linear extrapolation of the WTP for a percentage reduction in pollutant loading.

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

where X is the phosphorus or sediment loading, and where K is the basin W, B, or A.

Groundwater buffer value

Beyond the extractive value of groundwater, social value exists from water remaining in place within the aquifer. These social values include the capacity of groundwater to 1) buffer against periodic shortages in surface water supplies; 2) prevent subsidence of the land surface; 3) protect water quality by maintaining capacity to dilute groundwater contaminants; and 4) provide discharge to support recreational activities and facilitate ecological diversity. We adopt a low-end estimate of the social value by calculating only the value to buffer against periodic shortages in surface water supplies. There is insufficient information about the physical aspects of subsidence and discharge to stream beds to accurately calculate those values, and no local non-market valuation studies are available on protecting groundwater quality to include that value.

Groundwater provides farmers with a stable supply of water that represents a value beyond that of a supplement to non-irrigated crop production. The economic value of this risk management or stabilization role is called buffer value. Tsur (1990) defines buffer value *BV* as the amount a grower facing an uncertain surface water supply would be willing to pay for groundwater above the corresponding amount the grower would be willing to pay had surface water supplies been certain (or certainty equivalent).

Let the uncertain supply of surface water, *S*, be distributed according to a cumulative distribution having the mean μ and the variance σ^2. In the absence of groundwater, growers use the surface water available and enjoy the operating profit per hectare of *pF(S)*, where *F( )* represents per hectare yield response to water, and *p* is the net unit value of the crop. Tsur (1990) shows that buffer value is $BV(p,\mu) = pF(\mu) - pE\{F(S)\}$. By expanding *F(S)* about μ, *BV* can then be approximated by $BV \cong 0.5p[-F''(\mu)]\sigma^2$. This indicates the *BV* depends on the value of marginal productivity of water at μ, the degree of concavity of *F* at μ, and the variance of surface water supply σ^2. We assume the *BV* remains constant over time for each acre-foot of water left in the ground.

(16) $$BV\sum_{t=1}^{T}\delta_t\sum_{i=1}^{m}AQ_i\_t$$

Carbon offset value

The cultivation of crops is the main source of greenhouse gas (GHG) emissions in agriculture, which accounts for approximately 9% of total U.S. greenhouse gas emissions (GHG). In light of the GHGs associated global warming, it is imperative for the government to initiate the carbon offset policy to motivate farmers to reduce the agricultural related GHG emissions. Therefore, study on how the farmers as a profit maximizer will respond to the climate change incentive becomes the prerequisite of the execution of the scheme.

The climate change incentive, which rewards crop pattern that reduces the C footprint will influence the farmers' land use decision. As a result, including carbon value becomes necessary when we study farmers management behavior.

The carbon footprint is reduced by either diminishing carbon emission or increasing carbon sequestration, and the farmers will get their rewards or be charged correspondingly. Explicitly, the farmers will be charged if the net C footprint is negative and the carbon emissions is higher than carbon sequestration. In contrast, they will get their rewards if the net C footprint is positive and the carbon sequestration outweighs carbon emissions.

$$V_C = \sum_{t=1}^{T} \delta_t \left[\sum_{j=1}^{n} \sum_{s=1}^{S} (E_{jt} - S_{jstq})\, p_c\right] \tag{17}$$

E_{jt} is the carbon equivalent emission per unit of land from an existing land use j during period t. S_{jst} is the carbon sequestration per unit of land which is the function of land use j, soil type s and tillage method q.

The ecosystem services value objective is the summation of equation 15, equation 16 and equation 17 and hence the net benefits accrue from water quality improvements, the stock of groundwater as well as the mitigation of net carbon emission.

3. Optimization and solution methods

The goal of the analysis is to find land and water use patterns that maximize the economic objective (Eq. 12) for a given level of the ecosystem services (Eq. 15-17), and vice versa. By finding the maximum economic returns for a fixed value of ecosystem services, and then varying the value of the ecosystem service over its entire potential range, we trace out the efficiency frontier. The efficiency frontier illustrates what is feasible to attain from the landscape in terms of the economic and ecosystem service objectives, and the necessary tradeoffs between the objectives on the landscape. The efficiency frontier also illustrates the degree of inefficiency of other land and water use patterns not on the frontier, which shows how much the economic returns and/or ecosystem services could be increased.

The first step for generating points on the efficiency frontier involves finding the maximum present value of ecosystem services. This is any land use pattern that moves all production into

CRP and thus allows the volume of the aquifer to rise at the rate of natural recharge, sequesters the most greenhouse gas, and filters the most agricultural runoff. The second step is to solve the economic returns objective with no restriction on the final value of ecosystem services. With the highest and lowest levels of ecosystem services known, the remaining ecosystem service values for the efficiency frontier are chosen from equally spaced intervals between the lowest and the highest ecosystem service values to trace out the shape of the frontier.

The final step is to identify the land and water use patterns that maximizes the landscape economic returns while maintaining the ecosystem service values found in the second step. The value of economic returns matched to its corresponding ecosystem service value gives a combination that rests on the efficiency frontier. We perform the optimization with the Generalized Algebraic Modeling System (GAMS) 23.5.1 using the non-linear programming solver CONOPT from AKRI Consulting and Development.[4]

Data

The study area has three eight-digit HUC watersheds (L'Anguille, Big, and the Lower White)[5] 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 2,724 sites to evaluate how farmers make decisions

[4] The problem is not linear because the groundwater pumping cost and the amount of groundwater pumped are both solved as part of the problem and are multiplied together. The CONOPT solver available in GAMS is particularly effective at solving complex non-linear programs.
[5] The HUCs for L'Anguille, Big, and the Lower White are 08020205, 08020304, and 08020303.

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 irrigated corn, cotton, rice, and non-irrigated sorghum and CRP land (Table 3), and the irrigated soybean, non-irrigated soybean and double crop soybean are allocated on the basis of harvested acreage for 2010-2011 (NASS, 2012). County crop yield information for the past 5 years is used as a proxy for yields of each of the crops and not adjusted over time (Division of Agriculture, 2012).

Howarth (2009) observed that the future benefits of a public good, such as the conservation of aquifer resource, should be discounted at a rate close to the market rate of return for risk-free financial assets. This holds true even when the public good has risk characteristics equivalent to those of risky forms of wealth such as corporate stocks. The discount rate of 5% chosen for the analysis corresponds to the average yield of the 30yr Treasury Bond, a nearly risk free investment, over the last decade (US Department of the Treasury, 2012).

Groundwater

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). 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 (McKee and Hays, 2002).

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

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

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 (Barow and Leake, 2012). Specific yield, which does not vary across cells in our study area, is a dimensionless ratio of water drainable by saturated aquifer material to the total volume of that material. The product of hydraulic conductivity and saturated thickness is the transmissivity, and hydraulic conductivity is the rate of groundwater flow per unit area under a hydraulic gradient (Barlow and Leake, 2012). The hydraulic conductivity in feet per day for the Mississippi River Valley alluvial aquifer come from spatially coarse pilot points digitized in Clark, Westerman and Fugitt (2013).

The depletion of the aquifer beneath the cell is greater (i.e large ADF) if the grid cell is closer to the pumped well and the hydraulic diffusivity is bigger. We use the *ADF* to determine the proportion (or spatial weight) of the acre-feet of water pumped from a well that reduces the aquifer beneath the surrounding cells. The distance from the well and hydraulic diffusivity (based on the saturated thickness and hydraulic conductivity) of the surrounding cells influence the p_{ik} used in 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). The efficiently run groundwater pump delivers 1,800 gallons per minute and the stationary relift pump for the reservoir and tail-water recovery system delivers 2,000 gallons per minute (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, accounting for evaporation (Smartt et al., 2002). The share of nutrients and sediment captured by reservoirs (θ_i) is estimated to be 87% (or 0.87) based on the slope of the land at each site i (A. Sharpley, University of Arkansas, personal communication; AR Land Information Board, 2006) and prior modeling with Modified Arkansas Off-Stream Reservoir Analysis (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

The initial export of phosphorous and sediment to the mouth of a watershed depends largely on the crops currently grown and the slope of the land for each cell of the study area (Fig. 1c and Fig. 1d). The eastern part of the study area exports more phosphorous and sediment because the land is steeper and more corn and cotton are grown on this land.

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 from contingent valuation and travel cost studies of improved water quality, and the second study is contingent valuation survey by Hite et al. (2002) specific to pollutant reductions in the Mississippi Delta. The contingent valuation studies capture the use and non-use values of the better water quality. 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

Using monthly rainfall data for the season from June to September collected from the National Oceanic and Atmospheric Administration's (NOAA) weather station in Wynne, Arkansas for thirteen years from 2000 to 2012, the average seasonal rainfall μ is 12 inches and the variance of the seasonal rainfall, σ^2, is 19.4 inches squared (NOAA, 2013).

Several functional forms are estimated for the response of soybean yield to water input, and the natural log form is chosen to determine the concavity of soybean yield response to water input at the average rainfall for the season, $\left[-F''(\mu)\right]$, roughly 0.15 bushels per acre inch squared. The price of soybean based on a five year average of December futures prices for harvest time contracts is \$11.56 per bushel (GPTC, 2012) and the cost of production for a bushel of soybeans based on Arkansas production budgets is \$7.99 (Division of Agriculture, 2012), making the net unit value of soybeans equal to \$3.57.

The buffer value of an acre-foot of groundwater used to irrigate soybeans for an average season is then: $BV \cong 0.5p\left[-F''(\mu)\right]\sigma^2 = 0.5*3.57*0.15*19.4 = \$5.19.$ We base the buffer value of groundwater on soybean production, which is less profitable than rice, and hence this value is considered a conservative estimate. We choose a low-end for the buffer value of 1.56 and a high-end for the buffer value of 12.01 to consider the full range the buffer value of groundwater might have.

Carbon Emissions and Sequestration Value

As we discussed before, carbon equivalent emissions come from both direct and indirect GHG emissions, which include emissions from use of fertilizer, agricultural chemicals, and fuel by production practices. Due to the complexities of estimation, we employ the existing data of carbon equivalent emissions from Popp et al (2011), which provides the full information of the nitrous oxide emissions, methane emissions and other emissions. They are estimated on the basis of input use and changes for various crops. The values of the upstream emissions are from the USEPA (2007, 2009) combined with EcoInvent's life cycle inventory databased through SimaPro (Pre Consultants, 2009), and the value of other inputs are provided by Lal (2004).

Results

The efficiency frontiers with and without reservoirs are plotted out on Figure 1. Specific points along these frontiers along with their associated economic and ecosystem services returns, crop mix and environmental attributes are listed in Tables 4 to 7. Table 4 indicates that the maximum economic return (NPV) is 4,559 and 4,975 million 2013 USD without and with reservoirs, respectively. Similarly, the maximum ecosystem service value is 1,851 million dollars for both scenarios. As one moves from points A and F to points E and J (Figure 1) on the without and with reservoirs efficiency frontiers, respectively, the marginal rate of substitution between ecosystem services and economic returns increases. Table 4 indicates the points which maximizes economic returns (NPV) is associated with an ecosystem return of 14% and 7% of the maximum for the without and with reservoirs, respectively. Conversely, when maximizing ecosystem services the economic return is only 36% and 33% of its maximum for the without and with reservoirs, respectively. Thus, there appears to be a divergence between economic returns and ecosystem services. The following sections will delve into these tradeoffs.

Rice and corn are worthy of being highlighted in Table 5 as they are relatively large consumers of ES and relatively profitable. Thus, they are the two crops which see the most movement in and out of the cropping mix as you move from maximum ER to maximum ES, respectively. This movement in and out of the cropping mix is attributable to the fact that rice is the largest and corn the second largest consumers of water at 3.3 and 1.2 acre-feet per year. Couple this with the fact that rice is the most profitable crop in the mix (a profit of 348.00 per acre) and the tradeoff becomes less nebulous as you move from ES max to ER max. When looking at the ratio of profit per acre to water required we find that rice has the lowest efficiency measure ($105.45/acre-foot) across all irrigated crops. Thus, as you move towards maximizing ES, rice falls out of the mix quickly due to this inefficiency. In both scenarios cotton is not in the cropping mix for the ES maxing points (A and F) because of its heavy input (N, P, K and water) usage but enters the cropping mix when moving towards the ER maxing points (E and J).

Cropping Patterns

Table 6 illustrates the changes in cropping patterns and subsequent ER and ES along the efficiency frontier. Not surprisingly those points (A and F) which maximize ES are associated with cropping patterns that are centered on CRP acres and devoid of rice production as CRP maximizes carbon sequestration, minimizes carbon emissions, and maximizes water supply and quality. Moving along both frontiers towards maximizing ER, cropping patterns shift towards more heavily irrigated crops such as rice and corn and away from those crops which maximize sequestration and minimize irrigation such as CRP. Table 6 indicates that irrigated land in 2043 is 13% greater given the introduction of reservoirs than the scenario without as indicated by points J and E. The additional irrigation water from the introduction of the reservoirs is estimated to be applied primarily to rice and irrigated soybeans. In both scenarios (with and without reservoirs) across all points on the frontiers rice acreage falls from its initial 2013 levels and appears to the be crop most affected by the drop in water supply. This is not surprising given its large amount of water requirement and its relative profitability per acre inch of water applied.

Ecosystems services

Table 7 illustrates the suite of ES services and their dynamic changes along movements in the efficiency frontiers for the with and without reservoirs scenarios. Moving from point A (ES maximizing point) to point E (ER maximizing point) it is apparent there are significant decreases in ES. For instance, the aquifer decreases by 32,564 thousand acre feet or a 40% increase in water usage. Phosphorus, nitrogen, sediment increase by 3000, 289 and 938%, respectively when moving from maximizing ES to maximizing ER. With the addition of reservoirs and the creation of a new frontier, similar patterns unfold when moving from maximizing ES to maximizing ER. Moving from point F (ES maximizing point) to point J (ER maximizing point) it is apparent there are significant decreases in ES. For instance, the aquifer decreases by 20,117 thousand acre feet or a 20% increase in aquifer usage. Thus it would seem that the introduction of on farm reservoirs can play a major role in reducing aquifer depletion even under the ER maximization scenario. Phosphorus, nitrogen, sediment increase by 2825, 189 and 866%, respectively when moving from maximizing ES to maximizing ER. These are large increases but relatively smaller than without the introduction of reservoirs. Figure 1 indicates that along all points on the efficiency frontier that the scenario with reservoirs is second degree stochastic dominant in terms of *both* ER and ES.

Conclusion

This paper analyzes the ecosystem services and economic return of alternative crop mix patterns, considering the sustainable management of groundwater which includes spatially explicit representation of the aquifer and the potential of reservoirs to recharge the underline aquifer. We develop a spatially explicit ecosystem services model and economic model in the rice-soybean production region of Arkansas, USA. The ecosystem service model uses initial aquifer thickness, hydro-conductivity of the aquifer, a digital elevation model and soil characteristics to estimate the value of groundwater supply, surface water quality and greenhouse gas emissions. Moreover, the economic model predicts economic returns for alternative crop mix patterns under various site characteristics and locations. The social planner determines the crop allocation as well as the quantity of reservoirs to build over time to obtain high level of both ecosystem services and economic return, given the water and land constraints.

Several interesting findings emerge. First, although the value of ecosystem services is comparably smaller than the economic return, there exist tradeoffs between ecosystem service conservation and economic returns. Specifically, the results with full consideration of ecosystem services lead to losses in economic return. Vice versa, the landscape and water management results striving for economic returns harm the ecosystem service conservation. Therefore, a joint consideration of both ecosystem services and economic returns with appropriate weights is needed in search of maximum social welfare. It is further revealed that, a weighted consideration of both conflicting interests yield the largest social value, which may be of practical value for policy makers who wish to balance such interests.

Moreover, in contrast to most previous literature that generally suggests the ecosystem benefits of on-farm reservoirs, we observe a decreasing total value of ecosystem services with the construction of reservoirs. The value loss comes from farmers' change of crop types along with the availability of reservoirs in hopes of maximizing their economic profits, and such practices substantially increase greenhouse gas emission through the expansion of areas of carbon-inferior crops especially rice. Such cost is more than enough to offset the benefits in both water supply and water quality with the introduction of reservoirs. That said, it is found that with reservoirs farmers experience increasing economic profits after crop type switching mainly towards rice expansion. Furthermore, reservoirs may nevertheless be a feasible solution to groundwater scarcity in face of increasing irrigation water demand. Hence, from the social planner's perspective, a re-evaluation of the costs and benefits of reservoir construction is in need in specific localities with varying crop potentials and input constraints.

Finally, land use is found to experience changes in reply to different weights that may be placed upon the social planner's overall welfare objective to regulate farmers' production activities. to be specific, irrigated crops with high profitability expand areas in respond to reservoir availability. On the contrary, were larger incentives such as through financial means put upon farmers' production towards ecosystem service conservation, more agricultural land will shift to CRP. In the extreme, all agricultural land is converted to CRP with ecosystem as the only objective with or without reservoir. For the social planner, it would be wise to comprehensively assess such changing pattern in policy decisions that aim to meet certain land use goals.

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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)
$L_{i,rice}$, $L_{i,corn}$, $L_{i,cotton}$, $L_{i,isoy}$, $L_{i,dsoy}$, $L_{i,dsorg}$, $L_{i,dbl}$	Initial acres of rice, corn, cotton, irrigated soybean, dry land soybean, dry land sorghum, double crop irrigated soybean and winter wheat	81, 52, 10, 165, 57, 7, 47	99, 77, 40, 97, 49, 23, 73	220,624; 142,632; 25,891; 448,469; 154,946; 20,017; 128,552
$y_{i,rice}$, $y_{i,cotton}$, $y_{i,corn}$, $y_{i,isoy}$, $y_{i,dsoy}$, $y_{i,dsorg}$, $y_{i,dbl}$, $y_{i,wheat}$	Annual rice yield (cwt per acre), cotton yield (pounds per acre), corn, irrigated soybean, dry land soybean, dry land sorghum, double crop irrigated soybean, and winter wheat yields (bushels per acre)	71, 954, 163, 43, 26, 75, 34, 59	3, 12, 9, 3, 3, 6, 1, 4	-
dp_i	Depth to water (feet)	57	31	-
AQ_i	Initial aquifer size (acre-feet)	27,587	12,514	82,016
K	Hydraulic conductivity (feet per day)	226	92	-
nr_i	Annual natural recharge of the aquifer per acre (acre-feet)	0.001	0.04	547

Note: Number of sites is 2,724.

Table 2. Value of economic and reservoir model parameters.

Parameter	Definition	Value
pr_{rice} , pr_{cot} , pr_{corn} , pr_{soy} , pr_{sorg} , pr_{wht}	Price of rice ($/cwt), cotton ($/lbs), corn, soybeans, sorghum, and wheat ($/bushel)	14.00, 0.88, 5.50, 11.99, 5.23, 6.39
ca_{rice} , ca_{corn} , ca_{cotton} , ca_{isoy} , ca_{dsoy} , ca_{dsorg} , ca_{dbl}	Annual production cost of rice, corn, cotton, irrigated soybean, dry land soybean, dry land sorghum, and double crop irrigated soybean and winter wheat ($/acre)	646, 632, 742, 349, 289, 270, 656
wd_{rice} , wd_{corn} , wd_{cotton} , wd_{isoy} , wd_{dbl}	Annual irrigation per acre of rice, corn, cotton, full-season soybean, and double crop irrigated soybean (acre-feet)	3.3, 1.2, 0.8, 1.0, 0.8
$\omega_{\min}$, $\omega_{\max}$	Annual minimum and maximum capacity of a one acre reservoir (acre-feet)	1.4, 11
c^r	Estimated annual per acre cost of reservoir ($/acre)	96.7[a]
c^{rw}	Cost to re-lift an acre-foot to and from the reservoir ($/acre-foot)	22.62
c^p	Cost to raise an acre-foot of water by one foot ($/foot)	0.55
δ_t	Discount factor	0.98

[a] 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. Value of carbon model parameters.

Parameter	Definition	Value
$\lambda_{rice}, \lambda_{corn}, \lambda_{cotton}, \lambda_{isoy}, \lambda_{dsoy}, \lambda_{dsorg}, \lambda_{dbl}$	Yield converted to lbs per acre for rice (hundred weight), corn (bushels), cotton (pounds of lint), irrigated soybean (bushels), dry land soybean (bushels), dry land sorghum (bushels), and double crop irrigated soybean (bushels), wheat (bushels)	45.5, 25.4, 1.19, 27.2, 27.2, 25, 27.2
$\alpha_{rice}, \alpha_{corn}, \alpha_{cotton}, \alpha_{isoy}, \alpha_{dsoy}, \alpha_{dsorg}, \alpha_{dbl}, \alpha_{wht}$	Moisture content (wet basis) of rice, corn, cotton, irrigated soybean, dry land soybean, dry land sorghum, and double crop irrigated soybean and winter wheat	0.13, 0.155, 0, 0.13, 0.13, 0.14, 0.13, 0.135
$H_{rice}, H_{corn}, H_{cotton}, H_{isoy}, H_{dsoy}, H_{dsorg}, H_{dbl}, H_{wht}$	Harvest index of rice, corn, cotton, irrigated soybean, dry land soybean, dry land sorghum, and double crop irrigated soybean and winter wheat	0.45, 0.43, 0.45, 0.45, 0.45, 0.39, 0.45, 0.46
$\beta_{rice}, \beta_{corn}, \beta_{cotton}, \beta_{isoy}, \beta_{dsoy}, \beta_{dsorg}, \beta_{dbl}, \beta_{wheat}$	g of C per kg of dry biomass of rice, corn, cotton, irrigated soybean, dry land soybean, dry land sorghum, and double crop irrigated soybean and winter wheat.	360, 410, 420, 430, 430, 420, 430, 340
δ_{low} , $\delta_{conventioal}$	Aboveground biomass C for low tillage, and conventional tillage	0.40, 0.70
η_{low} , $\eta_{conventional}$	Belowground biomass C for low tillage, and conventional tillage	0.45, 0.40
$\chi_{rice}, \chi_{corn}, \chi_{cotton}, \chi_{isoy}, \chi_{dsoy}, \chi_{dsorg}, \chi_{dbl}, \chi_{wheat},$	Root C content of rice, corn, cotton, irrigated soybean, dry land soybean, dry land sorghum, and double crop irrigated soybean and winter wheat	350, 420, 360, 430, 430, 380, 430, 280
$\phi_{rice}, \phi_{corn}, \phi_{cotton}, \phi_{isoy}, \phi_{dsoy}, \phi_{dsorg}, \phi_{dbl}, \phi_{wheat}$	Root/shoot ratio of rice, corn, cotton, irrigated soybean, dry land soybean, dry land sorghum, and double crop irrigated soybean and winter wheat	0.16, 0.19, 0.21, 0.16, 0.16, 0.08, 0.16, 0.18

Source: Popp et al., 2011

Table 4: Ecosystem service and economic return values for selected points along the efficiency frontiers and for the 2013 landscape.

Land use pattern	Present value of economic returns ($ M)	Percentage of maximum economic return	Present value of ecosystem services ($ M)	Percentage of maximum ecosystem service value ($M)
Efficiency frontier				
Without reservoirs				
A	1654	36	1851	100
B	3234	71	1454	79
C	3960	87	1057	57
D	4376	96	660	36
E	4559	100	263	14
With reservoirs				
F	1654	33	1851	100
G	3408	69	1422	77
H	4180	84	993	54
I	4683	94	564	30
J	4975	100	135	7

Note: The values of economic returns are reported in millions of 2013 constant dollars and the volume of the aquifer in 2043 is reported in thousands of acre-feet.

Table 5: Present value of economic returns (ER) and ecosystem services (ES) with and without on-farm reservoirs. All dollar figures are reported in millions of 2013 constant dollars.

Ecosystem service or land use	Without reservoirs			With reservoirs		
	A	C	E	F	H	J
Greenhouse gas	1412	709	-28	1412	623	-207
Water Supply	420	345	292	420	367	342
Water Quality	19	3	-1	19	3	0
ES	1851	1057	263	1851	993	135
Rice	0	26	769	0	131	1150
Irrigated soybeans	0	626	674	0	612	785
Non-irrigated crop	0	297	625	0	286	441
Corn	1	1990	1940	6	2190	2172
Cotton	1	428	430	1	446	426
CRP	1652	593	121	1653	635	1
ER	1654	3960	4559	1654	4180	4975

Table 6. Land-use in 2043 for selected points along the efficiency frontier and the 2013 landscape. (in thousands of acres).

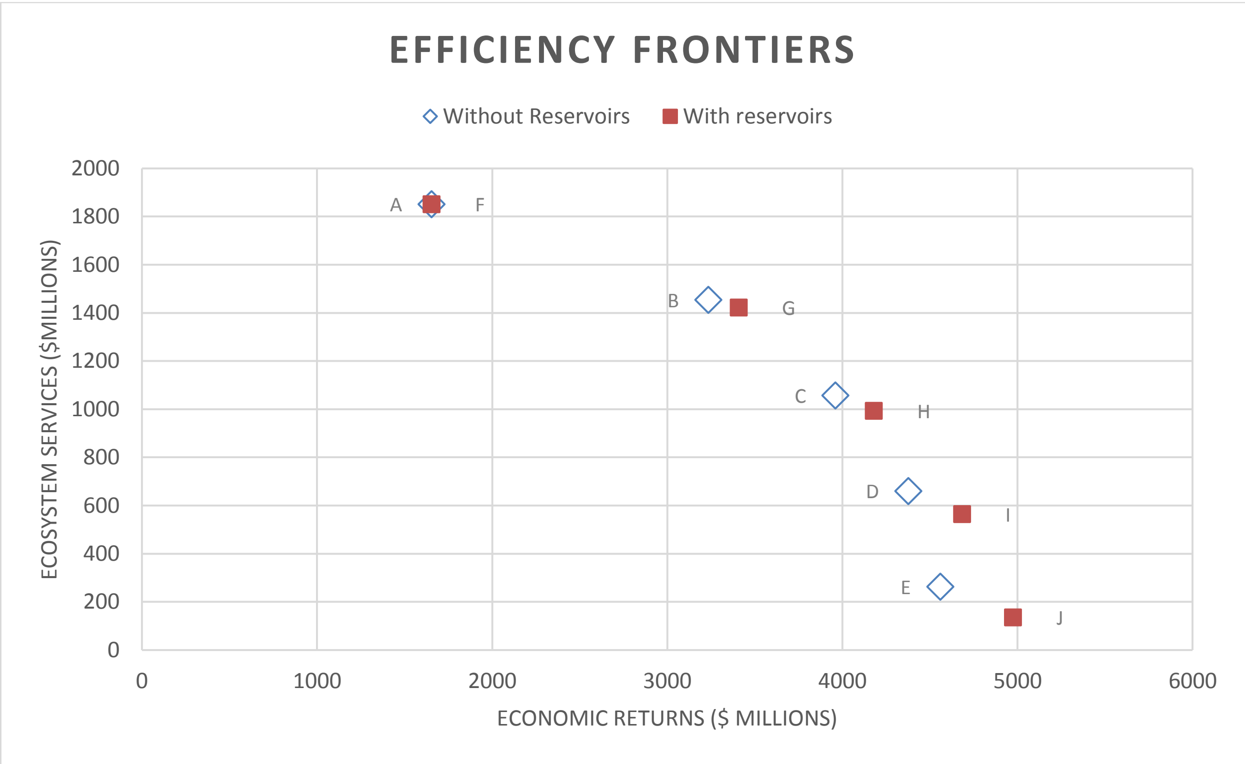
Land use	Initial, 2013	Without reservoirs, 2043			With reservoirs, 2043		
		A	C	E	F	H	J
Rice	221	0	6	164	0	20	214
Irrigated corn	143	0	384	379	0	412	411
Irrigated cotton	26	0	91	96	0	95	94
Irrigated soybeans	448	0	158	183	0	153	216
Non-irrigated soybeans	155	0	0	0	0	0	0
Non-irrigated sorghum	20	0	90	206	0	85	133
Double crop soybean	129	0	0	16	0	0	23
Reservoirs	0	0	0	0	0	19	48
CRP	0	1141	412	98	1141	357	1

Table 7. Changes along the efficiency frontiers from Point A without reservoirs and from Point F with reservoirs in the 2043 conditions of water use and supply, greenhouse gases, and water quality pollutants.

Water use and Ecosystem services	Without reservoirs			With reservoirs		
	Point A	A to C	C to E	Point F	F to H	H to J
Groundwater use (thousand acre-feet)	0	648	432	0	489	242
Reservoir water use (thousand acre-feet)	0	0	0	1	220	322
Aquifer (thousand acre-feet)	81464	-19147	-13417	91684	-13643	-6474
Net carbon emissions (thousand tons)	-479	-240	-239	-479	-269	-139
Methane emissions (thousand tons)	0	4	98	0	12	121
Nitrous oxides emissions (thousand tons)	0	61	24	0	66	24
Annual phosphorus exports (tons)	4	101	22	4	99	19
Annual nitrogen exports (tons)	124	346	13	123	221	4
Annual sediment exports (thousand tons)	101	697	251	99	604	253

Note: Net carbon emissions are positive when emissions exceeds sequestration and negative when sequestration exceed emissions

Figure 1: Ecosystem service and economic return values for selected points along the efficiency frontiers.



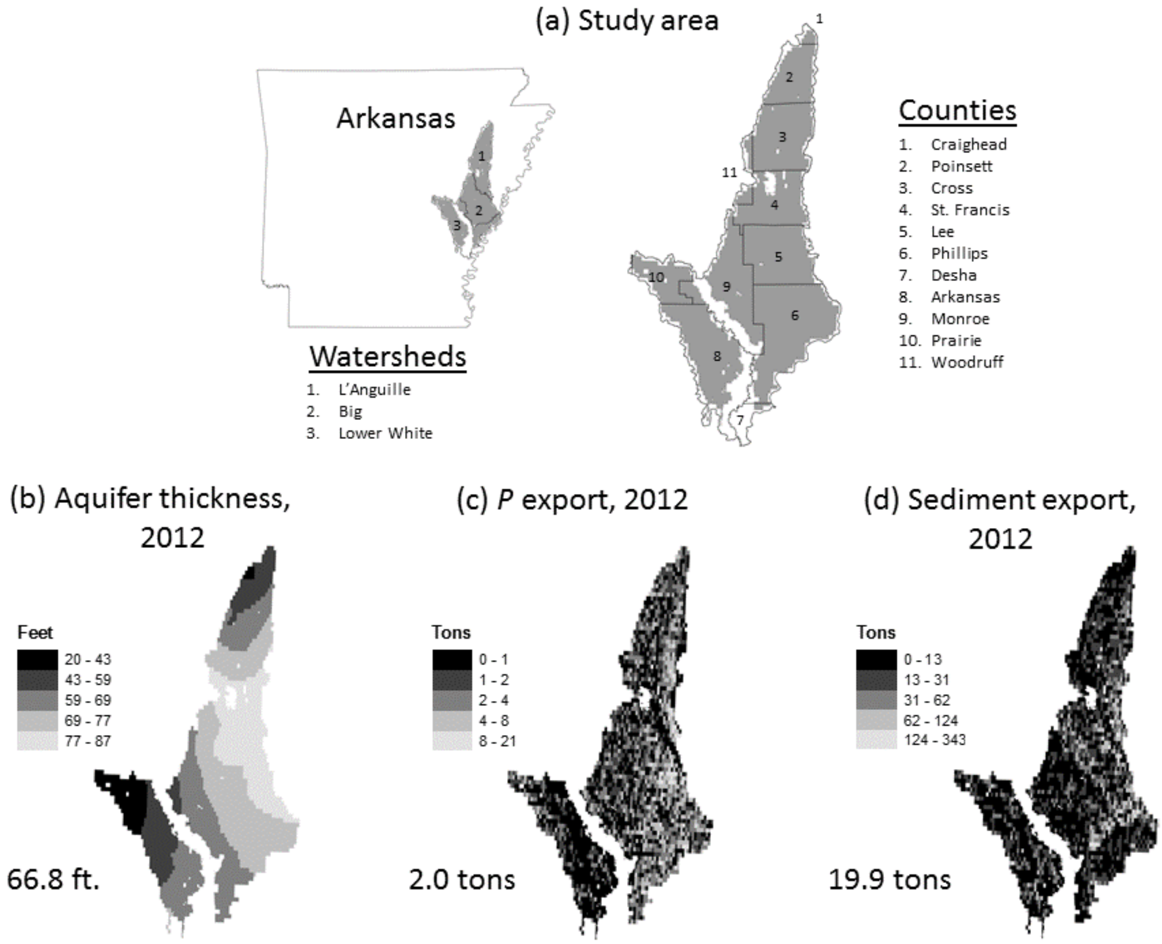


Figure 1. (a) Study area shown as grid cells. Three eight-digit HUC watersheds define the outer boundary of the study area. Public land and urban areas are excluded. Top-right map shows county lines overlay the study area. Top-left map show eight-digit HUC watershed boundaries overlay the study area. (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.

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) 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,

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

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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