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Hydro-economic Modeling of Managed Aquifer Recharge in the lower Mississippi

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Research Impact Statement: Managed aquifer recharge alone does not stop groundwater depletion in the Mississippi Embayment region due to the change in land use toward more water-intensive crops.

ABSTRACT: The Mississippi Embayment aquifer is one of the largest alluvial groundwater aquifers in the United States. It is being excessively used, located along the lower Mississippi River covering approximately 202,019 km^2 (78,000 square miles). Annual average groundwater depletion in the aquifer has been estimated at 5.18 Gm^3 (4.2 million acre-feet) for 1981-2000. However, since 2000 annual groundwater depletion has increased abruptly to 8 Gm^3 (2001-2008). In recent years, multi-state efforts have been initiated to improve the Mississippi Embayment aquifer sustainability. One management strategy of interest for preserving groundwater resources is managed aquifer recharge (MAR). In this study, we evaluate the impact of different MAR scenarios on land and water use decisions and the overall groundwater system using an economic model able to assess profitability of crop and land use decisions coupled to the Mississippi Embayment Regional Aquifer Study (MERAS) hydrogeologic model. We run the coupled model for 60 years by considering the hydrologic conditions from the MERAS model from the years 2002-2007 and repeating them 10 times. We find MAR is not economically attractive when the water cost is greater than \$0.05/m^3. Groundwater storage is unlikely to improve when relying solely on MAR as groundwater management strategy but rather should be implemented jointly with other groundwater conservation policies.

(**KEYWORDS**: coupled hydro-economic model; managed aquifer recharge; groundwater depletion; land use)

INTRODUCTION

Groundwater is a major water source for agriculture and municipalities especially during periods of drought. Water users tend to augment their water needs from groundwater when the surface water resources do not fulfill their needs during the dry season. Within the United States, groundwater in some places can be the foremost water source for irrigation, drinking water or industrial/municipal uses rather than surface water resources. For example, irrigation of cropland in Arkansas, Louisiana, Mississippi, and Tennessee depend to 61 – 84% on groundwater that they extract from the Mississippi Embayment Aquifer (Konikow, 2013).

The extensive reliance on the groundwater extraction from the Mississippi Embayment has several adverse economic and environmental impacts including the loss of the aquatic ecosystems, water quality reduction, and land subsidence (NRC 1997). Some area-specific adverse effects from Clark et al., (2011) are that by 2007, a water level decline of more than 25 feet (7.62 m) had occurred in nearly 36% of the alluvial aquifer extent within the Mississippi Embayment aquifer. On average, the groundwater water level in the alluvial aquifer declines at a rate of approximately one foot per year (Clark and Hart, 2009).

After an extended dry season, groundwater aquifers become less resilient to future droughts. Therefore, there is a high need for management procedures to protect and enhance the sustainability and resilience of groundwater aquifers. One approach for improving groundwater sustainability is artificial or managed aquifer recharge (MAR) that can take place via either direct injection or surface water inundation over a preselected recharge site. Applying those techniques requires the consideration of many factors such as the geological, geochemical, hydrological, biological, and engineering aspects (Bouwer, 2002; Reba et al., 2015; Levintal et al. 2022).

Recharge feasibility has been tested with several pilot projects (Fitzpatrick, 1990; Hays, 2001; Reba et al., 2015; Rigby, 2017), and there is high potential to effectively enhance the aquifer sustainability. However, many studies only use groundwater models or geospatial data analysis to determine suitable MAR sites without embedding any economic factors or considerations (Rahman et al. 2013; Russo et al. 2015; Ringleb et al., 2016; Marwaha et al. 2021). Many of these MAR studies are often pilot studies to figure out how feasible individual MAR scenarios are in terms of reducing water level decline (S1, supplemental material).

Motivated by multi-state efforts to save the Mississippi Embayment aquifer as a viable water source for future generations, our main goal is to evaluate the effects of different MAR scenarios on crop and land-use decisions, groundwater levels and storage, irrigation water supply and demand, and the spatial variability in the economic costs and returns using a coupled hydro-economic model in Eastern Arkansas. Unlike previous hydro-economic studies in the region (Kovacs et al. 2015; Tran et al. 2019; Tran et al. 2020a,b; Tran & Kovacs 2021), here we couple the distributed economic model developed by Kovacs et al. (2015) with the MERAS (Mississippi Embayment Regional Aquifer Study) MODFLOW finite-difference groundwater flow model of the Mississippi Embayment Aquifer (Clark and Hart, 2009). We use a modular approach in coupling the hydrologic and economic models, where each model is calibrated separately and then the information is passed between the two models via either a response function or data exchange platform (MacEwan et al. 2017). The economic model used in this study is built using GAMS (General Algebraic Modelling System; 2021). Python scripts were developed to facilitate data exchange between the GAMS and MODFLOW platforms.

The economic optimization model has been used to investigate MAR issues in the Mississippi Embayment, initially in Tran et al. (2019), who assessed the interaction of crop choice, surface reservoir storage, MAR, and groundwater conservation policies using a landscape level non-linear optimization approach. Later applications of the economic model focused on the effects of drought risk and drought severity on groundwater use (Tran et al. 2020a), siting of MAR locations dependent on factors such as natural recharge, proximity to surface water sources, and agronomic conditions of crops (Tran et al. 2020b), and climate adaptation willingness by farmer risk preference (Tran and Kovacs 2021). However, all of these studies use a simplified groundwater flow modeling approach consisting of a combination of Darcy’s law and Laplace’s continuity equation (Wang & Anderson, 1982), which assumes that there are simplistic surface-groundwater interactions and groundwater flows, to allow for a computationally tractable optimization. None of the previous studies coupled the economic model to a sophisticated groundwater flow model such as the USGS MERAS model; in fact, Tran et al. (2021) conclude that “An economic model coupled with a more sophisticated hydrologic model such as MODFLOW would better account for the complexity of groundwater flows”. Therefore, we developed the coupled hydro-economic model to assess the feasibility of different MAR scenarios to restore the sustainability of the Mississippi Embayment aquifer. Our study is among a handful of other studies that couples a MODFLOW groundwater model with an economic optimization model (Wesley et al. 2021; Rouhi Rad et al. 2020; Hrozencik et al. 2017; Niswonger et al. 2017; Morway et al. 2016; Kuwayama and Brozovic 2013; Varela-Ortega et al. 2011), which will add to the general knowledge base on coupled hydro-economic models (Harou et al. 2009). In addition, the focus of previous studies has been on groundwater overdraft and well production (Varela-Ortega et al. 2011; Rouhi Rad et al. 2020) and on surface water deliveries (Morway et al. 2016), but this study is a first to consider MAR scenarios.

LITERATURE ON COUPLED HYDRO-ECONOMIC MODELING

MacEwan et al. (2017) provided a comprehensive comparison of different coupled hydro-economic models in terms of interaction type, how the models are numerically linked, their calibration methods, and type of response function used. They distinguished two major coupling approaches: holistic and modular. The holistic approach combined water resources and economic models into a single and consistent mathematical programming, which is typically adopted to address combined environmental and economic issues (Cai, 2008). It is more convenient to use this approach when both economic and hydrologic variables are interacting and governing the decision-making process (Mulligan et al., 2014), however the downside of the holistic approach is that it is very computationally demanding and involves extended data collection and calibration efforts (Booker et al., 2012).

In the modular approach, both the hydrologic and economic models are run separately and are then followed by information exchange between the models (MacEwan et al., 2017): The gain when using the modular approach is that the individual models are standard models that are very well established and trusted according to their main users. Also, the information exchange is not as intense as in the fully coupled or holistic approach. The caveat is the extra needed effort to automate the data exchange and to ensure the consistency between the two models, since they might be written in two different languages (MacEwan et al., 2017), which we overcome in our study through the use of the programming language Python as an interface.

Many of the coupled hydro-economic modeling studies published to date use groundwater models that are uniformly and instantaneously responsive to pumping, which is fundamentally unrealistic as shown by Brozović et al., (2010). Kuwayama and Brozović (2013) developed an economic optimization model to manage the agricultural groundwater use by considering stream depletion analytically. The model was developed to simulate real field conditions to evaluate various groundwater pumping regulations. Mulligan et al. (2014) took the hydro-economic coupling effort a step further by incorporating a numerical groundwater flow model with a water utilization model to better assess the efficiency of different management procedures. Other studies implemented a coupled model by combining a physical hydrodynamic model and an economic model to evaluate the effect of drought on groundwater overdraft (Maneta et al., 2009; Medellín-Azuara et al., 2015).

Over the past 20 years, groundwater models have been increasingly coupled with other flow models including unsaturated flow models (e.g., Niswonger et al., 2006; Xu et al., 2012), farm process model such as the One Water Hydrologic Flow Model (Hanson et al. 2014), or the Process-based Adaptive Watershed Simulator (Shen and Phanikumar, 2010), ParFlow (Kollet and Maxwell 2008). In MODFLOW, many land surface or streamflow processes are simulated by separate packages such as the Stream Flow Routing (SFR1) package that is fully integrated into the MODFLOW code (Prudic et al., 2004). The SFR1 package allows simulation of runoff, evapotranspiration, and stream discharge by manipulating the flow depth at the mid-point of each stream reach. The more recent update of the SFR package (SFR2, Niswonger and Prudic, 2005) even estimates unsaturated flow between the groundwater aquifer and the land surface or the bottom of a stream especially whenever the unsaturated zone is extensive.

Several studies have coupled MODFLOW groundwater models with optimization models. For example, Morway et al., (2016) coupled MODFLOW-NWT (Niswonger et al., 2011) -which is a 3-D numerical model that simulates groundwater flow including surface water-groundwater interaction and unsaturated flow- with MODSIM (Labadie, 2010) to simulate basin hydrogeology and river/reservoir operations simultaneously. MODSIM was used to optimize the allowed water deliveries (e.g., river diversions and reservoir releases) that meet the water demands while complying with local restrictions and policies.

Harou et al., (2009) highlighted the significance of hydro-economic modeling and reviewed the key components, limitations, and procedures implemented in over 80 hydro-economic models from more than 20 countries over 45 years. Varela-Ortega et al., (2011) developed a hydro-economic model that was capable of representing the different interactions (environmental, social, and economic) of manmade and natural water systems. They used it to study the impacts of different policies and climatic scenarios on farm types and aquifers on a short and long-term planning horizon. Rouhi Rad et al., (2020) integrated three models (hydrologic, agronomic, and economic) into their hydro-economic model named MOD$$AT. Their model evaluates the economic impacts of extended groundwater overdraft and various policies by simulating changes in groundwater storage and well productions. Their MOD$$AT model was implemented to a case-study of Finney County in the southwest of Kansas, USA, but the authors explained how to apply it to other study areas.

METHODS

The basic method used to attain the above objective is a coupled hydroeconomic model. The model assesses the feasibility of different MAR scenarios on the Mississippi Embayment Regional Aquifer Study (MERAS) hydrogeologic model (MODFLOW 2005 based) developed by Clark and Hart (2009). In the following we provide a brief description of the original numerical MERAS model. Then, we follow that by a brief description for the economic model and our study area located in Eastern Arkansas. For more details about the MERAS MODFLOW model (e.g. development, calibration, validation) please see the original 2009 (Clark and Hart, 2009) and 2013 reports (Clark et al, 2013).

Study Area and Data of the Coupled Model

The study area is located in Eastern Arkansas, USA, and covers three eight-digit hydrologic unit codes (HUC). The study area overlays eleven counties in the State of Arkansas, USA (Figure 1), which is also one of the most critical groundwater areas of the state, where aquifers experience significant depletion and degradation (ANRC, 2018). The climate in the study area is moderate with an average annual rainfall that ranges from 56 inches in the south to 48 inches in the north (Kleiss et al, 2000) most of which occurs in the winter and spring. The mean annual temperature ranges from 18.8 °C in the southern part to 14.4 °C in the northern part (Cushing et al, 1970).

The main land use within the study area is agriculture and the major water source for irrigation is groundwater, especially in Arkansas, Louisiana, and Mississippi (Hutson et al, 2004; Clark et al, 2013; USDA-NASS, 2022). The primary irrigated crops are soybean, rice, corn, and cotton while dryland crops such as soybean are planted as well (USDA-NASS, 2022). Most of the groundwater pumping in Arkansas is used for irrigation, with surface irrigation such as furrow being the main application method in Arkansas, Missouri, and Mississippi (Hutson et al, 2004; Edward, 2016).

INSERT Figure 1 here

MERAS Hydrogeologic Model

The Mississippi Embayment aquifer system is a large groundwater aquifer system located along the lower Mississippi River covering approximately 78,000 square miles (202,019 km^2) over eight states including Mississippi, Louisiana, and Arkansas (which is the largest national groundwater user). The system is largely depleted in Arkansas, Louisiana, Mississippi, and Tennessee due to excessive pumping from the shallow alluvial aquifer (that is used for irrigation) and from the Claiborne aquifer (that is used for industrial and public water supply purposes). A significant portion of the groundwater storage losses took place in the alluvial aquifer in Arkansas and Mississippi as depicted in Figure 2. Maupin and Barber (2005) reported that approximately 42.2 million cubic meters (Mm^3) per day were withdrawn from the alluvial aquifer in 2000. From 1870 through 2007 more than 87% of the total groundwater pumping within the entire Mississippi embayment aquifer was being withdrawn from the alluvial aquifer (Clark and Hart, 2009).

INSERT Figure 2 here

To evaluate groundwater availability within the Mississippi embayment the USGS developed the Mississippi Embayment Regional Aquifer Study (MERAS) model (Clark and Hart, 2009). The finite difference MERAS model consists of 414 rows, 394 columns, and 13 groundwater layers. Cells are uniform in size; 1 mile by mile (2.59 km^2), while the layer thicknesses vary by cell and by layer. The northwestern corner of the model grid is positioned at 37º 27’ 28” North latitude and the West longitude is located at 93º 57’ 19”. Each layer of the model contains over 164,000 cells. For the study area located in Eastern Arkansas, the MERAS model contains 6 (out of the 13) aquifer layers and three confining units with two primary aquifers, the Mississippi River Valley Alluvial and the Middle Claiborne aquifers. The alluvial aquifer is mainly represented by layer 1 but other geological units are also present in the same layer. Those units include Pleistocene deposits and other formations covering the Vicksburg-Jackson confining unit in Louisiana and southern Mississippi (Figure S2.1, Clark & Hart, 2009). Layer 2 represents the Vicksburg-Jackson confining unit when it is present, otherwise the properties of layer 2 are revised to correspond to the alluvial aquifer. The upper Claiborne aquifer is represented in layer 3 where present, while beyond the upper Claiborne aquifer extent, the alluvial aquifer is extended. Layer 4, mainly, signifies the middle Claiborne confining unit when it is present, and the surficial unit where the middle Claiborne confining does not exist. The middle Claiborne aquifer is represented in layer 5 and varies from 3 to 6 layers according to its spatial location. Layers 8, 9, and 10 represent the lower Claiborne confining unit, the Winona-Tallahata, and the lower Claiborne aquifer, respectively. Layers 11 and 12 represent the middle Wilcox aquifer and the lower Wilcox aquifer, respectively. Layer 13 represents the lower Wilcox aquifer or the Old Breastworks confining unit where present (Fig. S2.1, Clark & Hart, 2009). More description for the hydrogeologic units of the Mississippi Embayment aquifer can be found in Hart et al. (2008). The model was originally calibrated from January 1, 1870, to April 1, 2007, for a total of 137 years and 69 stress periods. The predevelopment conditions are simulated in the first stress period as steady state; stress periods 2-27 representing years 1870 through 1986 have fluctuating lengths while the rest of the stress periods (28-69, representing years 1986 through 2007) have a length of 6 months each to represent the yearly seasons (spring-summer and fall-winter).

Areal recharge varies spatially depending on hydrogeology, land use, vegetation type, soil moisture, and slope (Figure 3). Main sources of areal recharge are rainfall and leakage from streams and irrigation return flow. The average recharge within the study area is 1 cm/year (0.01 m/year) but recharge varies regionally between 0.01 and 14.55 cm/year (0.0001 and 0.1455 m/year) (Arthur, 2001). Ackerman, (1989) anticipated the hydraulic head in the alluvial aquifer in the beginning of the 20th century to mimic the land surface and slope toward major rivers. Areal recharge is implemented in the MERAS model using the Recharge Package within MODFLOW-2005. Pumpage is obtained from site-specific 5-year water-use reports. The pumpage of the different (irrigation, municipal, and industrial) wells is simulated using the Multi-Node Well Package.

Streams within the model area are simulated via the Stream Flow Routing (SFR) package within MODFLOW to consider the groundwater - surface water interaction. Streams that either have more than 28.3 m^3/s (1000 ft^3/s) in discharge or streams that were verified in previous studies to interact with the groundwater aquifers were included in the model. Based on these criteria, 43 streams are simulated via the SFR package within the MERAS model domain. Surface runoff is entered to the SFR package for the selected streams based on the 30-year average runoff (Williamson et al, 1990).

The flow through the circumference of the model is assumed to be negligible and similarly the leakage through the base is very small compared to the volumetric flow within the aquifers above it. Therefore, the model boundaries as well as the model base are characterized as no-flow boundaries.

INSERT Figure 3 here

Economic Model

The economic model used in this study operates on an annual time step. The optimization occurs for each period but not across time periods. The model accounts for spatially heterogeneous natural and economic conditions in the study area. The economic model representing the study area in Eastern Arkansas has 3000 cells with the same cell size as the MERAS model (i.e., 1 mile^2 or 2.59 km^2) per cell. We define index $s \in \{1, 2, \ldots, 3000\}$ for the cells. We distinguish six-crops, i.e., rice, irrigated soybean, corn, and cotton, non-irrigated soybean, and double-cropped irrigated soybean with winter wheat. These crops may use*, i,* irrigation practices, e.g., conventional (contour-levee flood for rice and furrow for other irrigated crops), conservation furrow (poly-pipe hole selection method, and soil sensors), and zero-grade leveling flood for rice (Hignight et al., 2009; Henry et al. 2016; MSU, 2017). Farmers in the region have been using on-farm reservoirs to reduce the groundwater dependency (Smartt et al., 2002; Young et al., 2004). Thus, we also consider two other land uses in the model namely land fallowing and construction of on-farm reservoirs. Land balance constraints require that the sum of all land uses in each cell/site/farm, *s*, is less than or equal to the total cropland acreage of that cell. If a unit of land is allocated to on-farm reservoirs, the unit of land remains a reservoir for the rest of the simulation period. The amount of water used in cell, *s*, in time *t* must not exceed the sum of groundwater use, on-farm reservoir water use, and MAR water use in year, *t*.

Each year, t, a producer at site, s, decides to allocate, $A_{sli}(t)$in acres (1 acre = 0.4 ha) to land use, l, and irrigation practice, i. Therefore, a site, s, can have more than one crop and irrigation practice at each time, t. The possible land covers are the six crops, fallow, and the on-farm reservoirs. We define the objective function in equation (1) to maximize the sum of the total net return through optimal use of land, groundwater, MAR and other related inputs in the planning time horizon, t. The costs include production costs of crop, l, with irrigation practice, i, c_{li}, cost of MAR water, $C^{mar}(t)$, cost of on-farm reservoir water, $C^{rw}(t)$, and cost of groundwater pumping, $C^{gw}(t)$. The revenue is the price of the crop, l, p_l multiplied by the yield of the crop, l, planted at the site, s, with irrigation practice, i, y_{sli}. Each year, t, the model maximizes the total net return over, n, farm sites, which have variation in land use, hydrologic conditions (e.g., depth to water level, saturated thickness, and hydraulic conductivity), groundwater pumping rate and the costs of surface water conveyance for MAR.

$$\max_{\substack{¿A_{sli}(t),RW_s(t)\\ ¿,GW_s(t),MAR_s(t)}} \sum_{s}^{n}\sum_{l}^{m}\sum_{i}^{k}\left(\left(p_l y_{sli}-c_{li}\right)A_{sli}(t)-C^{mar}(t)-C^{gw}(t)-C^{rw}(t)\right) \qquad (1)$$

Where the total cost of MAR water is $C^{mar}(t)=\sum_{s}^{n}\left(c_s^{mar_{fix}}+c_s^{mar_{var}}\right)MAR_s(t)$.

The total MAR cost consists of fixed, $c_s^{mar_{fix}}$, (e.g., pipeline, and other infrastructure and equipment) and variable cost, $c_s^{mar_{var}}$, components (e.g., energy costs to transport the water to recharge wells). We assume all locations with MAR share the fixed costs, and these fixed costs are spread evenly throughout the study region and time. For the MAR cost we assume that farmers use bank filtration to extract surface water from streams through extraction wells for groundwater recharge. The concept is to induce flow through the streambed into the aquifer and capture that water, rather than groundwater from storage, and use a pipeline to transfer water to recharge wells in overdrafted areas. For this study, water is extracted nearby the main rivers (e.g., White and Arkansas rivers) and injected into sites/cells that can maximize the total net return (see S3, supplemental material). We assume farmers collectively maximize profits on the landscape in each period rather than individually maximize profits in each period. The actual degree of coordination among farmers is somewhere on the continuum between the social planner and the individual profit maximization.

$C^{gw}(t)=\sum_{s}^{n}\left(c_s^{gw}H_s(t)+c_s^{cw}\right)GW_s(t)$ is the total cost of groundwater pumping, comprised of the cost to lift one unit of water by one unit of depth, $c_s^{gw}(t)$, multiplied by the depth, $H_s(t)$, plus the capital costs per unit of water extracted for the well, $c_s^{cw}(t)$, which also accounts for new well drilling in response to aquifer decline. The capital costs may not be linear in the groundwater pumping if the equipment is more prone to maintenance and repair at high usage. However, we have not come across data to support the idea of non-linear capital costs.

$C^{rw}(t)=\sum_{s}^{n}\left(c_s^{rw}RW_s(t)+c_s^{cr}A_{sr}(t)\right)$ is the total cost of pumping from an on-farm reservoir, and includes the cost of irrigating with a unit of water from a reservoir, c_s^{rw}, multiplied by the volume of reservoir water, $RW_s(t)$, and the construction cost of a unit of reservoir land, c_s^{cr} multiplied by the size of the reservoir, $A_{sr}(t)$.

Maximization of the objective function (Eq. 1) is subject to non-negativity constraints and land availability, water balance, on-farm reservoir and maximum water injection rate constraints shown as in equation 2 to 4.

$$\sum_{l=1}^{m}\sum_{i=1}^{k} A_{sli}(t)=A_s \qquad (2)$$

Where $A_s$is the total acreage of cropland in cell, s. The total cropland in cell, s, $\sum_{l=1}^{m}\sum_{i=1}^{k} A_{sli}(t)$, must equal to A_s. The study area is a heavily developed agricultural region, and there are few opportunities to expand the amount of land in agriculture.

The irrigation water in the region comes from two sources, including groundwater as the primary source and on-farm water from constructed reservoirs (Reba et al. 2017). The irrigation water applied per area (acre) of the land crop, l, at the site, f, with irrigation practice, i, in time,t, is wr_{flti}. We assume producers switch to less intensively irrigated crops rather than deficit irrigating a high water demand crop. Empirical evidence from Moore et al. (1994) and Wang & Segarra (2011) suggests that perfectly inelastic demand for irrigation water is a reasonable assumption even in the long run. The total amount of water needed for irrigation at the site,s, is, $\sum_{l=1}^{m}\sum_{i=1}^{k} wr_{sli} A_{sli}(t)$, which equals the sum of groundwater use,$GW_s(t)$, and on-farm reservoir water use, $RW_s(t)$. The well injection would operate over six months (October through April) when surplus water is available, no irrigation occurs, and obtaining water rights is the most flexible (Fitzpatrick, 1990; ANRC, 2014). Groundwater wells are assumed to be dual-purpose, useful for both recharging surplus surface water in the winter and early spring, followed by pumping groundwater during periods of irrigation. MAR at sites without an extraction well would be farther away from the agriculture that would later utilize the groundwater. Economies of scale are unlikely to make a difference since the main constraint to high-volume injection is the ability of the water to permeate into the aquifer through the soil. In addition, previous studies showed that more irrigation-intensive crops are grown if the variable costs of irrigation decline through greater irrigation efficiency or the use of MAR (Ward and Pulido-Velazquez, 2008; Pfeiffer and Lin, 2014; Tran et al., 2019). Thus, the MAR water being recharged to the aquifers in a year, t, immediately becomes groundwater available for pumping in the same year. As a result, the water balance constraint (Eq. 3) is written as:

$$\sum_{l=1}^{m}\sum_{i=1}^{k} wr_{sli} A_{sli}(t) = GW_s(t) + RW_s(t) \tag{3}$$

Water for the on-farm reservoirs comes from two sources: recovery of runoff from irrigation and rainfall-runoff. The formulation reflects the total amount of water per one unit of an on-farm reservoir in time, t:

$$RW_s(t) = (\omega_{sr} + \omega_{sw}) - \omega_{sw}\left(\frac{A_{sr}(t)}{A_s}\right) \tag{4}$$

Where ω_{sr} and ω_{sw} are the water amounts per one unit of on-farm reservoir, claimed from precipitation and tail-water recovery, respectively, and $A_{sr}(t)$ is the area of the reservoir. Tail-water recovery ω_{sw} becomes negligible when the reservoir's size increases, at which point the amount of water coming from precipitation is the only source of water when the reservoir occupies the entire field (Kovacs et al. 2015).

We rely on two sources of information to estimate the maximum injection rate into a groundwater well: the hydrogeologic properties of the aquifer that affect the rate (Theis 1935; Cooper 1946) and results from actual recharge tests in the region (Fitzpatrick 1990; Kresse et al., 2014). Cooper (1946) simplified the Theis equation for large values of time, t, and/or small values for the well radius, r:

$$h_{sw}-h_{s0}=\frac{2.3\,MAR_s(t)}{4\pi T_s}\log\frac{2.25\,T_s t}{r^2 S_s} \qquad (5)$$

Where h_{sw} is the hydraulic head undergoing injection and h_{s0} is the initial hydraulic head before injection. T_s is the transmissivity. Solving for MAR on the right-hand side of Eq. 5, when $\frac{DTW_s}{h_{sw}-h_{s0}}$ is set equal to 1 (Gibson et al., 2018), indicates the maximum annual injection rates by site. DTW_s is the depth to the static water level below the ground surface.

In summary, the economic model maximizes equation (1) subject to the constraint set by equations (2)–(5). In order for equation (1) to be solved, excess water from rivers need to be moved to recharge sites at a water conveyance cost defined in the supplemental material.

Coupled Model

The hydro-economic model used in this study consists of the entire MERAS hydrogeologic model area (414 rows, 394 columns, 13 groundwater layers, Fig. S2.1) coupled to the economic model in 3000 cells in the Eastern Arkansas region (Figure 4). In other words, only 3000 cells of the MERAS model are selected to evaluate different MAR scenarios to improve the sustainability of the groundwater aquifer. Each cell in the coupled model has its own ID adopted from the MERAS model domain, covering rows 112 to 239 and columns 133 to 195 in the MERAS model domain. Vertically, the alluvial aquifer, comprised of the first two layers of the MERAS model represent the groundwater aquifer in the economic model.

INSERT Figure 4 here

The coupled model simulates processes in a transient way that allows data exchange between the hydrogeological and economic parts of the model after each stress period. The simulation period spans 60 years and 120 stress periods from 2007 to 2067. All stress periods have the same length of 6 months each (spring-summer and fall-winter) to mimic the irrigation and dormancy seasons in the study area. Each stress period is split into 2 equal time steps (3 months each). The total modeling period evaluated in this study is created by sequentially repeating the last six years of the original MERAS model (2002-2007) ten times for a total of 60 years. Since the model is run into the future, some model inputs needed to be predicted, while natural and measured boundary conditions imposed in the original MERAS model were maintained in the 60-year modeling period. The repeated boundary conditions include the natural recharge (mainly rainfall), stream flows, and pumping rates for the wells outside the economic model boundary extent.

The final heads of the original MERAS model obtained in 2007 were used as initial heads for the first stress period in the 60-year simulation period. The hydro-economic model is updated each time-step by parsing the output heads of the hydrogeologic model at the end of a given stress period to the economic model to obtain the depth to groundwater to estimate the pumping cost. The economic model first estimates the current pumping cost based on the final heads in a given stress period and then optimizes the groundwater pumping rates by determining the most economical crops to be planted and their water use as well as MAR rates depending on water availability, hydrologic conditions, and water conveyance cost. The optimized groundwater pumping and MAR rates estimated with the economic model along with the ending heads of the current stress period are then used as initial boundary conditions for the next stress period in the hydrogeologic model. The pumping rates are passed to the hydrogeologic model as inputs to the MNW package (Konikow et al., 2009) where MODFLOW determines the available storage for pumping. This procedure is repeated for all stress periods as shown in Figure 4.

The economic model written in GAMS is a two-dimensional horizontal model with one groundwater aquifer while the MERAS model is a three-dimensional model with multiple aquifer layers. This difference in model structure creates an inconsistency between the two models in terms of the input-output data exchange. For example, when parsing data from the economic model to the hydrogeologic model, a decision must be made from which layer (out of the 13 layers of the MERAS model) groundwater is pumped to meet crop water demand in the economic model. Since the economic model only assumes one groundwater aquifer (the alluvial aquifer), while the MERAS model represents the alluvial aquifer with two model layers, groundwater heads in the hydrogeologic model are extracted from the uppermost wet (non-dry) cell of either of the two model layers and exported to the economic model to be used for the water depth estimation. This procedure was selected because pumping from the uppermost water carrying aquifer is expected to deliver the extracted water via wells. The same procedure applies when assigning the output pumping rates from GAMS to the hydrogeologic model cells, where pumping is assigned to the uppermost wet cell in the hydrogeologic model.

Optimality condition for MAR choice

We consider how net returns in year *t*, $nr(t)$, from equation 1 depend on MAR. The first-order condition for MAR is:

$$\frac{\partial nr(t)}{\partial MAR_s(t)} = c_s^{gw} \frac{\partial H_s(t)}{\partial MAR_s(t)} GW_s(t) - c_s^{mar_{fix}} - c_s^{mar_{var}}. \tag{6}$$

Only the insertion of $C^{gw}(t)=\sum_{s}^{n}\left(c_s^{gw}H_s(t)+c_s^{cw}\right)GW_s(t)$ into equation 1 is necessary for deriving equation 6. The marginal benefit of MAR is the cost savings in the well pumping due to the higher water table that occurs because of MAR. The relationship between MAR and the water table, $\frac{\partial H_s(t)}{\partial MAR_s(t)}$, depends on the water balance within the MERAS hydrogeologic model. If the water table responds more to MAR due to the hydrogeologic properties of the site, then the marginal benefit of MAR is greater. The marginal cost of MAR is the added cost, either fixed, $c_s^{mar_{fix}}$, or variable, $c_s^{mar_{var}}$, required to recharge the aquifer. For sites with lower fixed or variable costs, then all else equal, more MAR occurs there.

Data Sources and Model Assumptions

The land use data to initialize the land-use input for the economic model originates from the 2017 Cropland Data Layer (USDA-NASS 2022). The average county crop yields from 2017 to 2021 are used as crop yields in the economic model (UARK, 2022). Crop prices come from the average of prices for each crop over the past five years (UARK, 2022). The construction and operation and maintenance costs for irrigation technologies, on-farm reservoirs, MAR, wells, and production costs for the crops are assumed to be constant over time in real terms (S3, supplemental material). We select a 2% real discount rate determined from a 5% thirty-year Treasury bond yield minus a 3% inflation expectation (USDT, 2022). Tran et al. (2019) compared the influence of low and high discount rates using an economic model with a simple hydrologic model, and they find that MAR increases substantially with a lower discount rate. In this study, we decided to not conduct a sensitivity analysis on the real discount rate to keep our focus on the influence of the MAR cost. The irrigation cost includes labor, fuel, lube and oil, and poly pipe for border irrigation plus the levee gates for the flood irrigation of rice, purchase and maintenance costs of wells, pumps, gearheads, and energy cost to lift a volume of a unit of irrigation water (Hogan et al., 2007). The annual on-farm capacity and cost of a unit of on-farm reservoir are defined based on The Modified Arkansas Off-Stream Reservoir Analysis (MARORA) tool (Smartt et al., 2002; Young et al., 2004)[32]. Additional descriptions of on-farm reservoir use and construction are given in S3.

Spatial hydrologic data, including the depth to the water table, initial saturated thickness, and hydraulic conductivity of the alluvial aquifer in the economic model come from Arkansas Natural Resources Commissioners (ANRC, 2018). The natural recharge and storativity values come from the U.S. Geologic Survey (Reitz et al. 2017). We use the distance from rivers to recharge sites to estimate the cost of MAR water conveyance. The distances are estimated by comparing recharge well locations, which are assumed to be at the center of each recharge site, to the closest river using the National Hydrography HUC 12 Dataset (NHD) (USGS 2022). Additional information on how water distribution costs are estimated are given in S3.

The current fraction of producers that use more efficient irrigation practices is less than 20% in the study area, and this fraction increases by about 1% per year (Edward, 2016). We consider furrow irrigation as the conventional irrigation practice for corn, soybean, and cotton, and contour-levee flood irrigation as the conventional practice for rice. Alternative irrigation practices (e.g., center pivot, surge irrigation, precision leveling, and poly-pipe with computerized hole selection) often reduce water use (Henry et al., 2016; MSU, 2017) and the lower costs associated with water pumping have the potential to increase net returns if the capital costs of the alternative irrigation practices are not too high. Adjustment coefficients to the costs of production and water use by crops relative to conventional irrigation practices depend on various agronomic sources (Hignight et al., 2009; Henry et al., 2016; MSU, 2017). Additional information on alternative irrigation practice adoption and the adjustment coefficients are given in S6.

MAR Water Cost Scenarios

MAR using injection wells has not been implemented at a large scale in Eastern Arkansas. The variable and capital cost (e.g., infrastructure, equipment, and interest) of MAR water in the region are highly variable and not well documented. For this study, we vary the costs per unit of MAR and off-farm water based on the costs of irrigation projects in Arkansas and the costs provided by Agricultural Research Service personnel and Eley-Barkley Engineering and Architecture, Cleveland, MS. The MAR variable cost depends on the required volume of water conveyed for MAR and the distance from an excess surface water source (e.g., a nearby river) to the recharge site. Additional information on the water conveyance costs is given in S3. To capture the range in MAR cost observed in the region, we explore four water costs scenarios for implementing MAR ranging from \$0.02 (MAR20), \$0.03 (MAR40), \$0.05 (MAR60), to \$0.16 (MAR200, baseline) per cubic meter (\$20-\$200 per acre-foot), respectively. The baseline scenario price is set at a high enough level to ensure no MAR occurs on the landscape. Initial model runs suggested that MAR is not economically attractive when the water cost is equal to \$0.16 per cubic meter. Thus, we select this scenario as the baseline scenario.

RESULTS AND DISCUSSION

We analyze the hydrologic and economic outcomes of MAR through the change in hydraulic head, change in groundwater storage, land and water use, and total net returns over the entire 60-year simulation period. First, we evaluate how the cost of MAR affects pumping and whether MAR contributes to an increase in groundwater pumping. Next, we evaluate how much MAR contributes to an increase in water intensive crops and the economic impact of MAR water use. We conclude by analyzing the extent to which MAR affects the dynamics of land and water use and groundwater storage.

Optimal Use of Water for MAR and Irrigation

As shown in Figure 5, among the four water cost scenarios considered, MAR is only economically attractive when its cost is less than \$0.05/m^3 (\$60/ac-ft). MAR water is only substantially increasing over time when its cost reaches \$0.02/m^3 (\$20/ac-ft). At this cost, MAR is offsetting some of the groundwater storage loss that is occurring due to extended groundwater pumping for irrigation in the region. For all other cost scenarios, MAR water use appears to be uniform over time. At a MAR water cost of \$0.05/m^3 (\$60/ac-ft) or higher, MAR use is less economically attractive. Only about 1.5 Mm^3 of water are recharged when the MAR water cost is \$0.05/m^3 (\$60/ac-ft), which is much lower than the 12 and 85 million cubic meters of water that are recharged when the MAR water costs are \$0.03/m^3 (\$40/ac-ft) and \$0.02/m^3 (\$20/ac-ft), respectively. This finding corroborates previous studies showing that little MAR has been used in the Eastern Arkansas region due to its high cost (Hays, 2001; Kresse et al., 2014).

INSERT Figure 5 here

Figure 6 shows the cumulative pumping rates for the four cost scenarios. Groundwater pumping reduces over time regardless of whether MAR is implemented. Since the study area is already impacted by groundwater level declines (ANRC, 2017) high groundwater use over the

60-year simulation period further diminishes the saturated thickness of the aquifer thereby increasing the cost of pumping which decreases overall groundwater use. Groundwater pumping is to some extent influenced by the cost of MAR water – a high cost of MAR water means a lower level of pumping (Figures 5 & 6), while a lower cost of MAR water increases groundwater storage and raises water levels and allows a greater portion of the MAR water to be pumped back up for irrigation. Previous studies have examined the rebound in groundwater pumping in the presence of MAR (Tran et al., 2019; 2020). However, the use of MAR water alone is unlikely to reduce groundwater storage depletion significantly in the region. There are, however, unexplored considerations in the change in cost and value of MAR. Namely, the cost of MAR will fall during periods of flood, and the value of MAR will rise during periods of drought. In addition, to couple the economic model with MODFLOW, the economic model assumes that the optimization occurs by myopic groundwater users that can augment their pumping wells. The limitation of this simplification is that the model likely predicts MAR is less feasible than MAR actually is because technology like MAR often requires high initial investments and/or fixed costs.

INSERT Figure 6 here

The Economic Impacts of MAR Water Use

Table 1 shows average land and water use over the 60-year simulation, and total net returns for the four MAR cost scenarios. The results demonstrate that MAR use increases the total net return by stabilizing the acreages allocated to irrigated crops such as corn, soybeans, and rice, though MAR use only reduces groundwater depletion marginally over the 60-year simulation due to a change toward more water-intensive crops such as rice (Table 1). When the cost of MAR water is equal to \$0.02/m^3 (\$20/ac-ft), MAR and groundwater use are considerably higher than when MAR water cost is \$0.03/m^3 (\$40/ac-ft) or higher. At the lowest MAR water cost (\$0.02/m^3 or \$20/ac-ft), average annual MAR and groundwater uses are 83 and 1,708 Mm^3, respectively, compared to only 12 and 1,641 Mm^3 when the cost of MAR water is \$0.03/m^3 (\$40/ac-ft). When more MAR occurs, the irrigated crop acreage increases to plant more profitable crops such as rice and soybeans compared to the baseline (no MAR) scenario.

INSERT Table 1 here

In cases of higher MAR water costs, MAR water use is less economically attractive compared to other forms of groundwater conservation such as planting dryland crops (e.g., dryland soybeans and CRP) and on-farm reservoirs. At a MAR water cost of \$0.05/m^3 (\$60/ac-ft) or higher, the total land allocated to on-farm reservoirs is almost two times higher than observed in the two lower MAR water cost scenarios (Table 1). In general, we find that using MAR water alone is unlikely to alleviate groundwater depletion in the region even if the cost of MAR water is optimistically cheap. This finding is in agreement with previous studies (Tran et al. 2019; 2020), who used a hydro-economic models with simplified groundwater flow components (e.g., Darcy’s law and the Laplace equation) to study the trade-off between using MAR water and surface reservoirs. Hybrid-groundwater conservation strategies, such as combining MAR with other water conservation or use measures could provide more flexible and appropriate groundwater conservation strategy.

Overall, we find that using MAR to reduce groundwater depletion can increase the total net return by stabilizing irrigated crop acreages and reduce the dependency on fossil groundwater resources whenever the cost of MAR water is economical. However, MAR water use is unlikely to stop groundwater depletion in the region even if the cost of MAR water is low as \$0.02/m^3 (\$20/ac-ft). MAR leads to a larger acreage of irrigated crops and smaller acreage of dryland farming and on-farm reservoirs than not implementing MAR. We also find that as MAR increases groundwater pumping increases (due to easier access to groundwater), which partially or fully offset the benefits of MAR to groundwater storage. This finding differs from other hydro-economic studies that have used MODFLOW in a coupled modeling approach to evaluate the potential impacts of MAR water use on groundwater storage (Niswonger et al., 2017; Scherberg et al., 2014; 2018). Our findings indicate that the cost of MAR is a limiting factor to adoption of MAR and that additional measures, such as restrictions on water use, might therefore be needed for groundwater conservation (Ward and Pulido-Velazquez, 2008; Grafton et al., 2018). Other considerations that would be expected to change the findings on a basin-by-basin basis across the study region and other regions include climate, hydrogeology, farming practices, inter-basin transfers, and reservoir storage, among others.

Effects of the Use of MAR on the Dynamics of Land and Water Use and Groundwater Stock

Table 2 depicts the change in crop mix, MAR use, groundwater pumping, and DTW for the years 2037, 2057, and 2077 for the MAR water cost scenarios. The results show that the irrigated acreage tends to decrease over time regardless of MAR water cost, but the dynamic patterns of MAR water use and crops choice depend on the costs of MAR: higher MAR water use is associated with higher total irrigated acreages and pumping compared to low or no MAR water use. When MAR water cost is 0.02/m^3 (\$20/ac-ft), MAR water use increases over time and coincides with higher irrigated acreages, resulting in a lower use of on-farm reservoirs and more dryland farming. When MAR water cost is 0.03/m^3 (\$40/ac-ft) or higher, more on-farm reservoir use and dryland farming are observed. High utilization of MAR stabilizes groundwater levels in the first twenty years of the simulation period after which the use of MAR slowly declines, while the acreages allocated to dryland crops and on-farm reservoirs increases. Specifically, a MAR water cost of 0.02/m^3 ($20/ac-ft) would lead to a reduction in total irrigated crop area of 62,208 ha, but an increase in total non-irrigated crop area and on-farm reservoir of 57,821 and 4,388 ha, respectively. However, when MAR water cost is equal to 0.16/m^3 ($200/ac-ft), a reduction in irrigated crop area of 94,461 ha, and a rise in non-irrigated crop area and on-farm reservoirs of 85,838 and 8,624 ha, respectively is observed. The difference in land use between the two scenarios results in a difference of about 10% in groundwater pumping (1,552 vs 1,405 Mm^3).

INSERT Table 2 here

These results imply that the cost of pumping decreases the more MAR is used, but MAR water use alone is unlikely to stabilize the groundwater levels. Even with an optimistic cost of MAR water such as $0.02/m^3 ($20/ac-ft), the groundwater storage in the study area still decreases over the 60-year simulation period. For example, for the MAR20 scenario, DTW increases by 2.81 m compared to 3.31 m for the MAR40 scenario. These results highlight that MAR water use is unlikely to stop groundwater depletion in the region, but MAR water use can slow the rate at which groundwater levels are declining over time unless MAR is combined with other groundwater conservation policies such as on-farm reservoirs, dryland farming and/or restrictions on groundwater use.

Water Budget Analysis

We analyzed the various water budget components for the different MAR cost scenarios to better understand the coupled model behavior within the study area. We select the MAR60 scenario to analyze the water budget in detail while providing summaries for all other scenarios. The results will be shown for the first two layers of the MERAS model that overlap with the 3000 cells of the economic model. Figure 7 shows the various groundwater budget components for the MAR60 scenario. The groundwater flow budget specifies the changes in the inflows into and outflows from the model domain for the entire 60-year simulation period.

INSERT Figure 7 here

Inflows are represented by positive values and outflows are represented by negative values. Figure 7 shows balanced total inflows versus total outflows for all stress periods. The MERAS model water budget includes five components, three of which may contribute to inflows (if they are positive) or outflows (if they are negative). Those three components are: local inflow (flow from/to the model domain to/from the neighboring cells), stream leakage, and storage withdrawal/accretion. The areal recharge is always considered an inflow while pumping is considered an outflow. As shown in Figure 7, pumping has the highest values in the water budget, which causes an increase in the withdrawal from groundwater storage and continuous gain (i.e., local inflow) from neighboring cells to the study area. This pattern holds except for four stress periods within the first seven stress periods when a minimal loss to the neighboring cells (i.e., less than 0.3 Gm^3) occurred. The pumping rate is decreasing over time (from 1.89 billion cubic meters (Gm^3) to 1.17 Gm^3 per stress period) which decreases the rate at which groundwater storage is declining over time from 2.2 Gm^3 to 0.7 Gm^3. Overall, we see minimal contribution from the stream leakage and relatively smaller contribution from areal recharge to the water budget compared to the pumping and storage withdrawal rates.

The net water quantities for all the water components for all scenarios are shown in Figure 8. Among the several water budget components, the pumping values represent the actual water amount withdrawn from the first two layers of the groundwater system which does not necessarily equal the input pumping value from the economic model. For example, when extracting the actual pumping value for a specific cell and time step in the MERAS model, it might be less than the requested pumping value, which can happen if the cell does not possess enough storage to meet the requested pumping amount, which will reduce the actual pumping to a value that is less than the requested pumping.

INSERT Figure 8 here

It is clear from Figure 8 that the stream component contributes with approximately equal values to all scenarios while the local flow component contributes with similar water amounts for all scenarios with minor differences (i.e., 3.7 Gm^3 max difference). Recharge is contributing with similar values for all scenarios except for the MAR20 scenario, which has significantly higher MAR amounts than all other scenarios as shown in Figure 5. On the other hand, the pumping is varying among the scenarios noticeably with the lowest pumping observed in the baseline scenario (about 88.6 Gm^3) and the highest pumping observed in the MAR20 scenario (about 94.5 Gm^3) for the 60-year simulation period. Pumping in the MAR60 scenario is slightly higher than in the baseline scenario (88.8 Gm^3 and 88.6 Gm^3, respectively) but overall, the pumping amounts that could be accommodated from storage in the MERAS model are about 92% of the amounts requested by the economic model for both scenarios (96.3 Gm^3 and 96 Gm^3, respectively). The difference in pumping amounts between all scenarios is mirrored in the storage withdrawals except for the MAR20 scenario which, among all the scenarios, has the largest pumping and smallest storage depletion. The lower storage decline is due to the additional influx of water from MAR. Figure 8 confirms that the pumping is withdrawn mainly from storage and neighboring cells.

Figure 9 shows the cumulative plots of individual water budget components over the 60-year simulation time. Pumping in all MAR cost scenarios starts high and declines over time as groundwater storage becomes more depleted (Figure 9a). The MAR20 scenario has the highest pumping rates over time due to having more water available for pumping from recharge compared to all other scenarios (see Figure 5 for MAR amounts). All scenarios except for the MAR20 scenario show similar recharge amounts in Figure 9b and significantly higher recharge amounts for the MAR20 scenario. This is because recharge amounts were determined by the economic model (see Figure 5 for total MAR amounts for each scenario).

INSERT Figure 9 here

The stream leakage shown in Figure 9c indicates a steady influx of stream water into the groundwater system with incidental backflows from the groundwater system to the streams. Such backflows cause the intermittent declines in the cumulative leakage for all scenarios, which originates from the 12 stress periods from the MERAS model that were repeated in the 60-year simulation period. Local flows, as depicted in Figure 9d, are not varying much from one scenario to another as all of them show almost the same behavior as they mirror the pumping behavior. Figure 9e shows the change in storage which is steadily declining over time but at a decreasing rate over the 60-year simulation period. The declining rate at which groundwater storage declines over time is largely influenced by the pumping. As shown in Figure 9a, pumping starts high and drops over time, which is influencing the local inflow and change in storage in the same way.

Figure 10 shows the difference groundwater heads between the end and start of the 60-year simulation period. While the MAR20 scenario resulted in some noticeable enhancement in groundwater heads none of the other MAR scenarios resulted in significant improvement within the study area.

INSERT Figure 10 here

The MAR20 scenario caused some head increases in the southeastern parts of the study area compared to the baseline scenario. Water tables declined on average by 5 m in the MAR20 scenario compared to an average decline of 10 m in the baseline scenario. A slight improvement is also noticed in the northern part of the study area, where groundwater tables decline by only 15 m in the MAR20 scenario compared to about the 20 m decline in the baseline scenario. Both the MAR40 and MAR 60 scenarios resulted in lesser improvements in heads as both experienced modest head increases in the central part of the model relative to the baseline scenario (~15 m vs 20 m decline in baseline scenario).

Our study did not incorporate potential effects of long-term climate change on land use decisions or the water budget of the alluvial aquifer. Historical streamflows in the region indicate that there is ample water for MAR even in the driest year. However, our present model does not allow for climatic change that could lead to no surplus water for MAR in some years.   Future studies could incorporate risk and uncertainty into this coupled model to evaluate the impacts of drought on the hydrology and economics of MAR (Fatichi et al. 2011; Collados-Lara et al. 2018; Steinschneider et al., 2019). The use of synthetic drought scenarios based on historical drought indices such as the Palmer Drought Severity Index (PDSI) could allow for the assessment of the impacts of drought on the hydro-economic outcomes of MAR use (Tran et al., 2019; 2020).

The MERAS model was released in 2009 while the economic model simulates land use decisions in the early 2020s, resulting in some data availability, accuracy and spatial resolution discrepancies that needed to be overcome when coupling both models. For example, the horizontal grid size of the MERAS model is 1 mile by 1 mile (2.59 km^2) which is fairly coarse and is impacting some of the hydrologic features as well as land use and crop acreage in the coupled model. Consequently, some of the obtained heads from the coupled model did not match historical heads observed in the Eastern Arkansas area. Future studies should attempt to use a more refined grid to capture land use and hydraulic features at a higher resolution. Since we relied on the boundary conditions (pumping, areal recharge, etc.) from the last six years (2002 – 2007) of the MERAS model and repeated them 10 times, future studies may implement different climatic scenarios and test their sensitivities. One of the next steps is to enlarge/vary the economic model domain in order to test a larger spectrum of economic variables.

CONCLUSIONS

We built a coupled hydro-economic model for the alluvial aquifer in Eastern Arkansas, USA to evaluate the hydrologic and economic benefits of implementing MAR as groundwater conservation strategy. Among the four water cost scenarios for MAR evaluated, we find that only the cheapest water cost scenario (\$0.02/m^3 or \$20/ac-ft) results in significant amounts of water being recharged, although not enough to prevent groundwater levels from further decline. We show that more MAR water use results in less use of other groundwater conservation strategies such as dryland farming and/or on-farm reservoir usage, and larger areas planted with water-intensive crops. As a result, MAR also increases groundwater pumping compared to the no MAR scenario. We find that the increase in groundwater pumping is likely to offset the groundwater storage gain from MAR, however, it is expected to increase the total farm net return regardless of the MAR water cost and pumping patterns. Among the four different MAR scenarios tested $(\$0.02/\mathrm{m}^3$, $\$0.03/\mathrm{m}^3$, $(\$0.05/\mathrm{m}^3$, $(\$0.16/\mathrm{m}^3)$, neither resulted in a significant improvement of groundwater heads. Improvements were limited quantitatively and spatially to only certain areas within the study region. This indicates that groundwater storage takes a long time to recover and that it might be more prudent to take mitigating measures (such as restraining strategies) to limit groundwater overdraft.

AUTHORSHIP CONTRIBUTION STATEMENT

AAA: Conceptualization, Methodology, Formal analysis, Writing - original draft. DQT: Conceptualization, Methodology, Writing - original draft. KFK: Conceptualization, Data curation, Writing - original draft. HED: Conceptualization, Writing - review & editing.

SUPPORTING INFORMATION

Additional supporting information may be found online under the Supporting Information tab for this article: it includes descriptive statistics of the model data across the study area and model parameters. It provides additional information regarding on-farm reservoir use and construction.

DATA AVAILABILITY STATEMENT

The data that support the findings of this study are available from the corresponding author upon reasonable request.

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Figures and Tables

Table 1. Average annual land and water use for each MAR water cost scenario.

Crop	**Initial**	**$\$0.02/\text{m}^3$**	**$\$0.03/\text{m}^3$**	**$\$0.05/\text{m}^3$**	**Baseline ($\$0.16/\text{m}^3$)**
Irrigated corn (ha)	56,251	49,495	48,959	47,635	47,527
Irrigated cotton (ha)	17,401	31,608	31,608	31,607	31,607
Fallow (ha)	-	504	568	570	561
Irrigated soybean and winter wheat (ha)	8,094	-	-	-	-
Dryland soybean (ha)	55,442	53,046	61,683	62,223	62,941
Irrigated soybean (ha)	238,360	268,031	267,824	270,720	270,128
On-farm reservoir (ha)	-	1,548	1,727	2,780	2,870
Irrigated rice (ha)	118,573	89,889	81,751	78,587	78,488
Total irrigated crop area (ha)	438,680	439,024	430,144	428,549	427,749
Total non-irrigated crop area (ha)	55,442	53,550	62,251	62,793	63,502
On-farm reservoir (ha)	-	1,548	1,727	2,780	2,870
MAR use (MCM)	N/A	82.85	12.38	1.47	0.00
Groundwater pumping (MCM)	N/A	1,708.03	1,641.01	1,605.01	1,600.00
DTW (m)	16.93	18.81	19.73	20.01	20.04
Total net return[a]	N/A	4,246.02	3,593.03	3,581.00	3,588.01

Note: [a] Total net return is in 2022 million dollars. The average annual land and water use corresponds to the average over the 60-yr simulation periods for each MAR water cost scenario. The results for the year 2018, one year after the model starts, differ slightly from the initial year (2017). The results of irrigated crops in 2018 increase by 2.6% at the expense of non-irrigated crops compared to 2017. Using the results for the year 2018 instead of the initial year 2017 unlikely alter the main conclusions of this study. Also, the results for the year 2018 might not reflect the status quo. Our optimization model reflects the best-case scenario and might miss some of the forces already occurring in the economy that are likely to either magnify or alleviate some of the pains associated with groundwater overuse and/or changing climate conditions. Comparing the simulated results to the initial year allowed us to better highlight where the status quo is unsustainable and, therefore, where management actions are most needed.”

Table 2. Change in crops planted overtime for each MAR water cost scenario.

	$\$0.02/\text{m}^3$			$\$0.03/\text{m}^3$			$\$0.05/\text{m}^3$			$\$0.16/\text{m}^3$ (Baseline)		
Crop	**2037**	**2057**	**2077**	**2037**	**2057**	**2077**	**2037**	**2057**	**2077**	**2037**	**2057**	**2077**
Irrigated corn (ha)	-14,003	-15,760	-21,099	-14,120	-15,356	-21,199	-17,666	-22,933	-29,844	-18,389	-22,997	-29,915
Irrigated cotton (ha)	+37,193	+37,023	+36,792	+37,193	+37,023	+36,792	+37,193	+37,023	+36,792	+37,193	+37,023	+36,792
CRP (ha)	+1,165	+1,165	+1,506	+1,379	+1,462	+1,511	+1,383	+1,466	+1,515	+1,368	+1,446	+1,495
Double-cropping[a] (ha)	-20,000	-20,000	-20,000	-20,000	-20,000	-20,000	-20,000	-20,000	-20,000	-20,000	-20,000	-20,000
Dryland soybean (ha)	-23,899	+17,900	+56,315	-4,256	+43,072	+80,388	-4,177	+43,934	+81,780	-2,102	+46,641	+84,342
Irrigated soybean (ha)	+70,254	+67,488	+53,621	+70,051	+68,244	+59,156	+86,453	+81,706	+73,317	+85,084	+79,345	+70,773
On-farm reservoir (ha)	+3,887	+4,106	+4,388	+4,077	+4,690	+5,454	+6,644	+7,532	+8,341	+6,860	+7,790	+8,624
Irrigated rice (ha)	-54,597	-91,921	-111,523	-74,323	-119,134	-142,102	-89,830	-128,727	-151,902	-90,013	-129,248	-152,112
Total irrigated crops (ha)	+18,848	-23,170	-62,208	-1,199	-49,224	-87,352	-3,850	-52,931	-91,637	-6,125	-55,877	-94,461
Total non-irrigated crops (ha)	-22,734	+19,065	+57,821	-2,878	+44,534	+81,899	-2,794	+45,400	+83,296	-735	+48,087	+85,838
Reservoir (ha)	+3,887	+4,106	+4,388	+4,077	+4,690	+5,454	+6,644	+7,532	+8,341	+6,860	+7,790	+8,624
MAR use (MCM)	52.11	102.98	140.88	3.88	2.24	2.15	3.68	1.63	1.48	0.00	0.00	0.00
Groundwater pumping (MCM)	1,760.71	1,638.81	1,552.02	1,698.76	1,552.21	1,456.73	1,650.45	1,509.90	1,411.05	1,645.91	1,503.67	1,405.41
Change in DTW (m)	+1.52	+2.31	+2.81	+2.05	+3.31	+4.54	+2.05	+3.25	+4.70	+2.13	+3.39	+4.73

Note: [a] Double-cropping means irrigated soybean and winter wheat are planted in one year. Positive numbers indicate the increases in crop area while negative numbers indicate the decreases in cropland area relative to initial areas. Positive numbers indicate the increases in DTW while negative numbers indicate the decreases in DTW relative to initial DTW.

Figure 1. Watersheds (HUC-8) and county boundaries within the study area in Eastern Arkansas, Mississippi Delta region.

Figure 2. Cumulative groundwater pumping in the entire MERAS model domain (Figure 5, Clark et al. 2011). Note, 1 million acre-feet are 1.23 km^3.

Figure 3. MERAS model domain showing recharge zones (Clark and Hart, 2009) and area of interest (white outline in left panel) for the hydro-economic study with initial predominant land use categories for each cell.

Figure 4. Flow diagram showing data requested between the groundwater and economic models using Python and API GAMS. Note: we use Local Polynomial Regression Fitting (i.e., **LO**cally **WE**ighted **S**catter-plot **S**moother [LOESS]), a local weighted regression approach, to fit a smooth curve through the MAR use over time with span parameter of 0.75. The dependent variable is MAR use, while year is set to be the independent variable. LOESS can capture the relationship between the two variables, while making minimal assumptions about the relationship.

Figure 5. MAR use by the cost of MAR water. Note: we use Local Polynomial Regression Fitting (i.e., **LO**cally **WE**ighted **S**catter-plot **S**moother [LOESS]), a local weighted regression approach, to fit a smooth curve through the groundwater pumping over time with span parameter of 0.75. The dependent variable is MAR use, while year is set to be the independent variable. LOESS can capture the relationship between the two variables, while making minimal assumptions about the relationship.

Figure 6. Groundwater pumping for each MAR water cost scenario. MAR20, MAR40, MAR60, and MAR200 (baseline) are equal to a MAR cost of \$0.02, \$0.03, \$0.05, \$0.16 per cubic meter, respectively. Solid lines show the mean values for each of the type-specific fitted polynomial functions. The shading around the lines represents 95% confidence intervals.

Figure 7. Groundwater flow budget components for the MAR60 scenario in billion cubic meters (BCM). Each stress period is 6 months in length. The total simulation period is 60 years (2007-2067).

Figure 8. Net water budget components for all scenarios

Figure 9. Individual water budget components for all scenarios over the simulation time.

Figure 10. Head differences (in m) for all scenarios over the simulation time. Positive values indicate a rise in water table over the 60-year simulation period while negative values indicate a decline.