RESEARCH ARTICLE

Modeling and evaluating marginal pumping fees in groundwater commons: do varying scarcity levels matter?

Godwin Kwabla Ekpe[1]

Abstract
Price-based irrigation water-conservation policies are often designed as fixed per unit fees. In groundwater commons, however, this approach presupposes that irrigators assign the same value to each unit of water withdrawn, irrespective of the scarcity levels they individually face. This ignores spatial interdependencies in groundwater commons. In this paper, I examine the effect this possible tax structure misspecification has in measuring the performance of such Pigouvian taxes. I model the price of irrigation water as a non-constant marginal cost function dependent on the constant per unit fee and a variable cost-metric measure of scarcity, namely depth-to-water. Using a difference-in-difference econometric framework with irrigation data from San Luis Valley, results show that irrigators' response to the constant marginal fee significantly depends on the scarcity levels individual irrigators face. More importantly, the results suggest that models that overlook the spatial element of scarcity would overestimate irrigators' response to such pumping fee—which can misguide policy decisions.

Keywords Conservation · Groundwater commons · Irrigation · Scarcity · Spatio-temporal interdependencies · Water tax

JEL Classification Q21 · Q25

1 Introduction

Increasing levels of drought, population growth and consumption have put immense pressure on groundwater resources in many parts of the world, including the United States (Ficklin et al. 2015; Williams et al. 2015; Gleick 2010; Castle et al. 2014;

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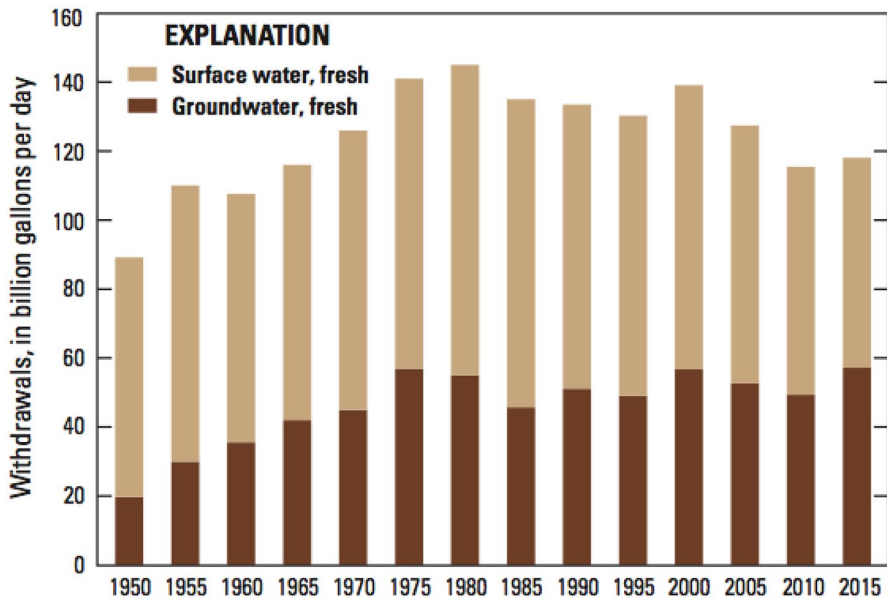1 Department of Economics, Northern Illinois University, DeKalb, IL, USA

Fig. 1 Irrigation water use in the US by source and State, 2015. *Source*: USGS Circular 1441

Smith et al. 2017; Ward 2014). Like any other common-pool resource, we can expect unsustainable levels of exploitation to plague the groundwater commons, especially where property rights are not fully developed (Libecap 2011; Ostrom 1990; Dales 1968; Rouhi Rad et al. 2021). As millions of gallons of groundwater are withdrawn from wells and aquifers on a daily basis in the US for irrigation (see Fig. 1 for the 2015 records), the rates of withdrawals outstrip many aquifers' rates of recharge (Scanlon et al. 2006; Smith et al. 2017). This increasing reliance on groundwater has put the sustainable extraction of the resource in jeopardy.

To ensure water conservation, researchers and policy makers advocate for price incentives like a groundwater tax. Usually where authorities or water agencies use pricing as conservation policy, such policies are often designed by setting the price of water as a fixed per unit (marginal) fee/tax. In this paper, I provide evidence that the constant marginal cost pricing/tax may be inappropriate in the case of irrigated agriculture in groundwater commons where irrigators share a common aquifer and do not withdraw from the same well. I show that such a constant per unit fee/tax structure may be misspecified as it ignores spatio-temporal aspects of pumping and underplays the impact of past and present withdrawals on the cost of current and future withdrawals.

In regions faced with high water-stress situations, the incidence of spatial externality can be important in analyzing groundwater extraction by irrigators (Provencher and Burt 1993; Pfeiffer and Lin 2012; Negri 1989; Huang et al. 2013). This is essentially because in the groundwater commons where seepage exists, depth-to-water is the medium through which the incidence of spatial externalities may be felt (Wang and Segarra 2011). A constant marginal price policy assumes that every unit of water pumped is uniformly valued by irrigators in monetary terms

regardless of where their wells are located and the depth from which they withdraw water. Yet, the existence of stock or pumping cost externality in groundwater commons implies that the lift or extraction cost is expected to vary both with time and across wells as the depth-to-water changes due to the interplay of the continuous extraction and recharge rates, where probable. Specifically, in groundwater commons, water scarcity affects irrigators in a non-constant manner which can interact with the per unit tax in a way that affects how irrigators respond to such a pricing policy. As such, scarcity cannot be overlooked when adjusting the cost of the input.

Linked to the structure of such constant per unit pricing policies is the approach to estimate their impact. Given the role of depth-to-water in groundwater commons as described above, this paper shows that an important way to get a better picture of the impact of such a policy is to isolate and study the effect depth-to-water has on irrigators' responses—even where non-spatial models are used. This paper therefore contributes to the literature by isolating and emphasizing the role of depth-to-water, not merely as an input in determining the cost of lift but more importantly as a cost-metric measure of relative water scarcity. I model the price of water as a non-constant marginal cost with the fixed per unit tax and a variable component. The variable component of this cost structure relies on the varying depth-to-water levels in wells that serve irrigators as a cost-metric measure of water scarcity. By introducing this non-constant marginal cost into a multi-output irrigated agriculture production function, I show both theoretically and empirically that groundwater pumping taxes/fees designed as constant per unit fees may inadequately capture the true price of water for individual irrigators. This inadequacy arises from such charges not internalizing the relevance of the spatio-temporal variability in scarcity levels faced by irrigators. Consequently, estimating the impact of such taxes without accounting for this variability may result in estimates that are biased, incomplete, or insufficient in capturing the full scope of the effect of the tax on water use in groundwater commons.

To empirically examine the proposition that depth-to-water levels would have important interactions with such a tax and, thus, affect its performance, I utilize a data set on irrigated agriculture from the groundwater commons of the San Luis Valley (SLV) of Colorado, similar to that used by Smith et al. (2017). Using standard fixed effects specifications within a difference-in-difference framework, I find that the effect of the tax on water use significantly depends on depth-to-water levels. Additionally, the introduction of depth-to-water as a proxy for water-stress severity (water scarcity) significantly alters the magnitude of the estimates in comparison to models that do not incorporate it. Specifically, models that do not incorporate such a measure significantly overestimate the impact of the tax, which may misguide policy decisions.

The rest of this paper is structured as follows. Section 2 describes some of the background of irrigation water pricing, including reasons for this study's specification of irrigation water price as a non-constant marginal cost function dependent on the constant per unit fee and a variable cost-metric measure of scarcity, namely depth-to-water. In Sect. 3, I demonstrate how introducing this non-constant marginal cost into a multi-output irrigated agriculture production function à la Moore and Negri (1992) and Moore et al. (1994) reveals a possible bias inherent in models

that only consider the fixed per unit price of irrigation water. Section 4 describes the data used in the study, while Sect. 5 presents the identification and empirical estimation framework. Results are presented in Sect. 6, with robustness checks; Sect. 7 concludes.

2 Background

The prospect of pricing as a policy tool for conserving water has been discussed within academic circles since at least the 1960s (Scheierling et al. 2006). While Price-based incentives could internalize the social cost of withdrawals, their efficiency depends on the elasticity of water demand. A higher elasticity translates to more substantial responses to pricing.

However, water pricing policies have not historically succeeded in inducing large reductions in water use. Although elasticity estimates vary, they tend to be low (Koundouri 2004; Scheierling et al. 2006). Scheierling et al. (2006) found a mean price elasticity of 0.48 with a standard deviation of 0.53 from a meta-analysis of 24 studies involving 73 irrigation water price elasticity estimates.

Cultural or societal norms can affect how users respond to water prices (Smith 2018). Irrigators can directly adjust through short-run water demand at the intensive margins (that is, amount of water applied per acre of land), and they can also indirectly adjust through long-run water demand at the extensive margins which can involve altering the size or number of parcels irrigated and land reallocation decisions (Moore et al. 1994). The intensive margins response is noted to be mostly decreasing in water price but because the extensive margins adjustments can either be negative or positive, their magnitudes determine whether overall response will be elastic or inelastic.

Data aggregation over different hydro-geographical regions can obscure pricing policy impacts, often leading to lower elasticity estimates (for example, Moore et al. 1994). In contrast, studies using relatively more disaggregated data focused on a specific geographic area tend to show relatively more price elastic estimates (for example, Schoengold et al. 2006; Smith et al. 2017).

Three major approaches that have been used to study irrigation water price elasticity include mathematical programming, field experiments, and econometric estimation (Yang et al. 2003; Scheierling et al. 2006). While large variances in demand estimates are common among programming approaches, field experiments that employ agronomic concepts often produce low elasticity estimates (Scheierling et al. 2006; Ogg and Gollehon 1989). Econometric studies generally suggest inelastic demand (for example, Ogg and Gollehon 1989; Moore et al. 1994) but also show estimates can be elastic at higher water prices (for example, Frank and Beattie 1979; Nieswiadomy 1985; Schoengold et al. 2006; Smith et al. 2017).

It is intuitive that irrigators would be more responsive when the magnitude of the price change is large as shown, for example, in Schoengold et al. (2006) and Smith et al. (2017) where the estimates are − 0.787 and − 0.77, respectively. However, they seem to be outliers even though they still fall below unit elasticity. Moreover, Schoengold et al. (2006) only considered surface water users. Indeed, there are cases

specifically concerning groundwater taxes/pumping fees, where there was total failure of such policies to induce conservation (for example, Yang et al. 2003; Schuerhoff et al. 2013). This prompts questions about the role of the inherent competitive nature of groundwater extraction due to the incidence of spatial externality. It also raises questions as to whether irrigation water pricing policies adequately reflect the real price of groundwater, especially in the groundwater commons.

Although a full exploration of the questions regarding competition in groundwater irrigation is not the focus of this study, the fact of the existence of externalities in groundwater irrigation as established in the literature provides a basis to interrogate the pricing of groundwater.[1] The focus of this study is, therefore, with regards to an important observation in the literature regarding irrigation water pricing policy design and how the impact of such policy is estimated. Across studies and water conservation management bodies that focus on water pricing as a conservation tool, water is considered a variable input with its price fixed per unit. It is worth highlighting the fact that such a constant per unit pricing policy is with respect to the price of the good (i.e., water) and not any other cost associated with using it.

Indeed, many types of water conservation policies have been designed and used in different parts of the world. But where pricing the actual good is involved, the policies have mostly entailed a constant per unit fee in one form or the other. Tsur and Dinar (1997) chronicled examples from across the world, including California, India, Jordan, Spain, Morrocco, Türkiye, and Chile. In all instances, the pricing policy involved one or combinations of volumetric pricing, tiered and two-part pricing, output and input pricing, and area pricing. In the Netherlands, for example, groundwater fees have been euro cents per cubic meter of water used (Schuerhoff et al. 2013)—an example of constant per unit volumetric pricing policy. In China, groundwater itself used to be free and the only cost for irrigation comes from fuel and electricity costs of lift (Yang et al. 2003). A tiered pricing is also reported in Jordan (Venot and Molle 2008).

Yet, none of these pricing mechanisms has the sort of variable component that relates to the spatio-temporal variability in the levels of scarcity faced by individual irrigators. Implied in such a pricing policy is the assumption that every unit of water pumped is of the same value to all irrigators regardless of well location, depth-to-water and, thus, their scarcity levels they face. Unfortunately, such fixed per unit cost pricing/tax may be inappropriate in the context of groundwater commons irrigation where irrigators share a common aquifer. The possible existence of "strategic externality" as described by Negri (1989) or its equivalent "stock externality" and "risk externality" described by Provencher and Burt (1993) imply that depth-to-water is the medium through which any such externality is felt, and as such marginal units of water pumped would not be valued by irrigators the same. If marginal units of water do not have the same and constant value across irrigators because they face different scarcity (depth-to-water) levels, then

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[1] See, for example, Theis (1938), Negri (1989), Provencher and Burt (1993), and Wang and Segarra (2011) for discussions on externalities in groundwater commons.

their response to an externally imposed or exogenous constant fee could be very much influenced by these spatio-temporal varying scarcity levels.

Examples of empirical/econometric evaluation of actual pumping fees in groundwater commons as a policy are rare due to unavailability of observational data (Lago et al. 2015; Smith et al. 2017). Much of the empirical literature on evaluating irrigator responses to water price is not really an evaluation of a pricing policy implemented in real life. What prices that are used by researchers in most cases are either imputed (Mieno and Brozović 2017) or simply energy cost of extraction or irrigation. For example, Moore et al. (1994) uses an engineering equation to derive the price of groundwater pumping as energy cost, which is a function of fuel price, fuel efficiency, depth-to-water, and pumping pressure. Hendricks and Peterson (2012) defined a per unit cost of pumping following Rogers and Alam (2006) in similar manner.

Indeed, there are empirical studies (for example, Mieno and Brozović 2017; Hrozencik et al. 2022) and hydro-economic modeling studies (for example, Guilfoos et al. 2016; Hrozencik et al. 2017; Mulligan et al. 2014) that, although not assuming constant marginal pumping costs, also consider energy cost of pumping as the price of groundwater. While the price derived across these studies involves the depth-to-water as an input, they differ significantly from the current study. First, these past studies did not entail an evaluation of any price-based conservation policy where a per unit fee is imposed on water withdrawal. They essentially studied how farmers respond to total pumping cost as price of water. This paper estimates the impact of such a constant per unit fee imposed on irrigators in San Luis Valley, Colorado. Second and more importantly, this paper, specifying the price of water as a non-constant marginal cost—with depth-to-water treated as cost-metric variable component and the constant per unit fee as fixed component—isolates and studies the effect this cost-metric scarcity variable has on irrigators' response to the imposed fee. Because the previous studies combined the depth-to-water with fuel price and other variables to arrive at dollar price per acre-foot, they could not isolate how depth-to-water affects farmer decisions regarding water use, and since there is no separate per unit dollar fee imposed, they could not study how the effect of such a fee could be impacted by depth-to-water. It is also worth emphasizing that, in the instant case of San Luis Valley, irrigators also bear individual energy cost of extraction but that does not form part of the pricing policy being implemented. The "price" of groundwater withdrawal as per the policy is neither a function of the levels of individual water use nor the depth-to-water (the scarcity) levels that confront irrigators individually.

Furthermore, it is important to note that in the above-mentioned studies, the source of variability in water "price" is attributed to variation in energy cost of pumping. Energy prices may vary between counties and not within the same county (Huang et al. 2013). But in relatively close geographic region as in the groundwater commons setting of San Luis Valley of Colorado, it is reasonable to expect the variation in marginal pumping cost to be mostly from a source other than electricity prices.
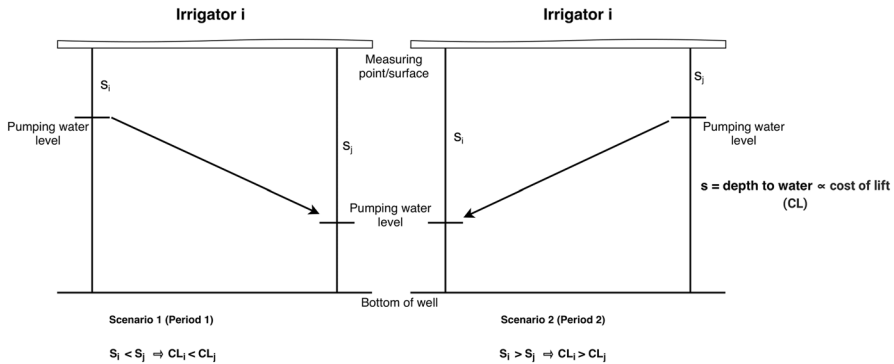
Fig. 2 The spatio-temporal variation of depth-to-water between irrigators *i* and *j*. *Source*: Adapted from Pfeiffer and Lin (2012)

3 Theoretical model

To provide a theoretical foundation for the empirical model, I present a model of multi-output irrigated agriculture production in which marginal cost of water depends on water scarcity. It is important to note that this study is within the context of irrigation in groundwater commons in arid regions. Within this setting, I proxy water scarcity with depth-to-water. Though water scarcity could be measured or indicated by other factors/variables, including drought, precipitation levels and water table recovery rate, the dataset used for this study does not have well-specific measures for these variables. Moreover, depth-to-water does not only reflect the availability of water in the groundwater system but also provides a direct indication of lift cost at each individual well and thereby serving as a variable that is likely to influence water withdrawal decisions of irrigators as shown in previous studies (Pfeiffer and Lin 2012; Hendricks and Peterson 2012; Huang et al. 2013). Accordingly, in this model, it is assumed that individual irrigators consider depth-to-water as a signal of the level of water scarcity they are confronted with at a given time when making decisions. The cost of water for irrigator *i* is affected directly by the depth-to-water in the well from which he/she pumps water; a larger depth-to-water at irrigator *i*'s well causes a higher lift cost to irrigator *i*, while a smaller depth-to-water indicates a smaller lift cost to irrigator *i*.

Under scarce water conditions and a lowered water table (i.e., pumping water level), irrigator *i* might respond by pumping less water than normally. Figure 2 demonstrates how the variation in depth-to-water across wells and across time may affect the cost of lift at two neighboring wells. If authorities impose a fixed per unit pumping fee for conservation purposes, the impact of such a fee would partly depend on the producer's existing level of costs, which largely hinges on the cost of lift. Given that this depth-to-water-based cost is not likely to be constant in spatio-temporal terms, the effect of this important interaction may be overlooked, if this cost is not factored into the total per unit cost of irrigation water when designing and implementing water conservation fees/taxes for groundwater commons.

3.1 A multi-output irrigated agriculture production model with a non-constant marginal cost of water

I adapt the model of multi-crop agricultural production from Moore and Negri (1992) and Moore et al. (1994) which extrapolates from the theory of the multi-output competitive firm. In this model, producers make long-run decisions regarding crop-choice, land allocation, and crop supply as well as short-run decisions on irrigation water use. Groundwater irrigation is considered the main variable input of interest with the marginal cost of groundwater extraction as the price of water. In this current study, the marginal cost of water is defined as a function of a water scarcity measure *(s)* and the constant monetary price of per unit water pumped *(b)* which has conventionally been used as a conservation policy tool. I write this per unit cost as $B = B(s,b)$, where s is measured as *depth-to-water*.[2] As stated earlier, there are other water scarcity indicators, including drought, however depth-to-water—or the distance from the top of a well to the groundwater level below—most appropriately mimics cost of lift across wells. Furthermore, in a shared aquifer context, there is bound to be spatio-temporal interactions among irrigators and this interaction is reflected through depth-to-water levels in wells from which irrigators draw water (Theis 1938; Wang and Segarra 2011). The two scenarios depicted in Fig. 2 are a highly simplified demonstration of the nature and dynamics of groundwater irrigation and the importance of depth-to-water in understanding spatial externalities in groundwater commons. Both scenarios show that the larger the depth-to-water, groundwater is scarcer and the higher cost of lift. In scenario 1, individual i has a shallower depth to the water (i.e., greater water stock) than j. Where aquifer material allows for seepage, water would flow from i to j. Scenario 2 is the reverse scenario. In a shared aquifer setting, an irrigator cannot store their stock of water in their wells to use at a later date. Thus, in each scenario, the only way irrigator i (in scenario 1) or irrigator j (in scenario 2) can lay claim to water and extract it at lower marginal cost is to increase pumping (Pfeiffer and Lin 2012). Another way to look at this is, for instance, if we assume that scenario 1 is what pertains in period 1 and irrigator i with a smaller depth-to-water does not increase pumping, water will flow to j and by period 2, i would be extracting at a higher marginal cost. In sum, when the depth-to-water increases, the cost of lifting the water to the surface is expected to go up for a given well, all else equal. As such the per unit cost B is increasing in both arguments, i.e. $B_s(s, b) > 0$ and $B_b(s, b) > 0$.

Farmers[3] are assumed to be aware that they operate in a water-stressed region and are concerned about the possible impact of increasing water cost (associated with water scarcity) on their operations. In this light, I assume that a farmer may choose to optimize water use by investing in a water-saving technology. Assume water-saving technology investment by farmer i is z, where z is a function of the cost of the amount of effective water (crops beneficial water use) used relative to the cost of the

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[2] It is worth noting that the index for individual irrigator is dropped for simplicity since irrigators' optimization problem is identical.

[3] The terms farmer, irrigator and producer will be used interchangeably in this paper.

actual amount of water pumped before and after the adoption of the water-saving technology. Let a_0 represent the relative cost before the adoption of the water-saving technology and a_1 be the relative cost after the adoption of the water-saving technology. Thus $z = z(a)$, where $a = a_0 - a_1$.

Farmers are risk-neutral producers who maximize profits subject to land constraints. Constrained by land, the irrigators choose to maximize the value of their land by optimally cultivating the most profitable crops. Without loss of generality, it is assumed that, a producer chooses among $L+1$ number of growable crops, ordered in terms of increasing water use intensity. Thus the choice of crop-land (acres of crop l) is a choice from the set $l \in \{0, \dots, L\}$, where for tractability, fallowing $l = 0$ is considered a crop type which requires zero water use. This crop-land decision is based on factors such as cost of water B, prices of other non-water inputs $\mathbf{r}$, crop prices $\mathbf{p}$, land constraints N_f, and other variables $\mathbf{x}$ relating to climate, weather and soil conditions. To match the definition of crops above, crop price is, thus, a vector $\mathbf{p} = (p_0, p_1, \dots, p_L)$, with $p_0 = 0$. In addition, water-saving technology $z(a)$ is also a major consideration particularly because the producer operates in a water-stressed environment and has concerns over the cost implication of water scarcity to his operations. Moreover, crop-land choices or reallocation after a given initial period is affected by the technology in use during the initial period as it constitutes a source of switching cost. In effect, the choice of crop-land n_l is a function that can be stated as $n_l\big(\mathbf{p}, \mathbf{r}, B(s, b), N_f, z(a); \mathbf{x}\big)$.

The crop-land allocation functions can be obtained by maximizing profit subject to the land constraint. Write the enterprise-wide profit maximization problem as:

$$
\mathbf{\Pi}\big(\mathbf{p}, \mathbf{r}, B(s, b), N_f; z(a), \mathbf{x}\big) = \max_{\{n_0 \dots n_L\}} \sum_{l=0}^{L} \boldsymbol{\pi}_l\big(p_l, \mathbf{r}, B(s, b), n_l; z(a), \mathbf{x}\big)
$$

$$
\text{s.t.} \sum_{l=0}^{L} n_l = N_f. \tag{1}
$$

where $\boldsymbol{\pi}_l(.)$ is the individual crop-land level profit function, and N_f is the total allocatable fixed land.[4] The standard competitive profit maximization assumptions are invoked such that producers behave as price takers and $\boldsymbol{\pi}_l(.)$, as shown by previous researchers (for example, Lau et al. (1976)), are considered continuous, twice differentiable, convex and closed in output and input prices (p_l, $\mathbf{r}$, and B) in the non-negative orthant and homogeneous of degree one in output and input prices. Additionally, these individual crop-land level profit functions $\boldsymbol{\pi}_l(.)$ are strictly decreasing in $\mathbf{r}$ and B, and non-decreasing in n_l and p_l.

This implies our problem set-up is well defined and fulfills the Kuhn–Tucker conditions for necessity and sufficiency. Therefore, the producer will solve the following Lagrangian:

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[4] The subscript f is used to explicitly indicate that the total allocatable fixed land includes total land fallowed for the growing season, which is not modeled in Moore and Negri (1992) and Moore et al. (1994).

$$\mathcal{L}(n_l, \lambda) = \sum_{l=0}^{L} \boldsymbol{\pi}_l\Big(p_l, \mathbf{r}, B(s,b), n_l; z(a), \mathbf{x}\Big) + \lambda\Big(N_f - \sum_{l=0}^{L} n_l\Big). \tag{2}$$

An interior solution solves the following first order conditions.

$$\frac{\partial \mathcal{L}}{\partial n_l} \quad = \quad \frac{\partial \boldsymbol{\pi}_l\big(p_l, \mathbf{r}, B(s,b), n_l; z(a), \mathbf{x}\big)}{\partial n_l} \quad = \quad \lambda. \tag{3}$$

$$\frac{\partial \mathcal{L}}{\partial \lambda} \quad = \quad N_f \quad = \quad \sum_{l=0}^{L} n_l. \tag{4}$$

The irrigator allocates land among crops such that the marginal profit is equalized across cultivated crop-land, with λ being the shadow cost of crop-land. Furthermore, any interior solution will have a binding land constraint. Following Lau et al. (1976) and Moore and Negri (1992), I assume the crop-land level profits take a normalized quadratic form. Accordingly, let $n_l^*\big(\mathbf{p}, \mathbf{r}, B(s,b), N_f; z(a), \mathbf{x}\big)$ denote multi-output optimal crop-land allocation. We restate the multi-crop profit function in (1) as:

$$\mathbf{\Pi}\Big(\mathbf{p}, \mathbf{r}, B(s,b), N_f; z(a), \mathbf{x}\Big) = \sum_{l=0}^{L} \boldsymbol{\pi}_l\Big[p_l, \mathbf{r}, B(s,b),$$
$$n_l^*\big(\mathbf{p}, \mathbf{r}, B(s,b), N_f; z(a), \mathbf{x}\big); z(a), \mathbf{x}\Big]. \tag{5}$$

Since (5) is the optimal profit based on the optimal crop-land choice, we can derive the crop supply function using Hotelling's lemma:

$$y_l\big(p_l, \mathbf{r}, B(s,b), n_l^*; z(a), \mathbf{x}\big) = \frac{\partial \boldsymbol{\pi}_l\big(p_l, \mathbf{r}, B(s,b), n_l^*; z(a), \mathbf{x}\big)}{\partial p_l},$$
$$\forall l = 0, \dots, L. \tag{6}$$

Similarly, since groundwater is a variable input, we can derive its conditional factor demand as:

$$w_l\big(p_l, \mathbf{r}, B(s,b), n_l^*; z(a), \mathbf{x}\big) = -\frac{\partial \boldsymbol{\pi}_l\big(p_l, \mathbf{r}, B(s,b), n_l^*; z(a), \mathbf{x}\big)}{\partial B(s,b)},$$
$$\forall l = 0, \dots, L. \tag{7}$$

To assist the smooth characterization of the crop-level profit functions, we will assume input non-jointness (Shumway et al. 1988; Moore and Negri 1992).[5] The

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[5] This assumption applies to the agricultural production in the San Luis Valley in Colorado, the study region of this paper. In the San Luis Valley, irrigators generally grow a specific crop on a parcel of land for a growing season. In other words, cultivated parcels of land are in effect crop-specific parcels; there is generally no mixed cropping on a given parcel in a given farming season. We can, therefore, argue that inputs are exclusively assigned to crop-specific cultivation activities and this in turn ensures that we are able to attribute each crop's output to their unique input assignment.

fixed land which is allocatable among the various crop types, provides the sole source of input jointness. Thus, the multi-output profit function naturally disaggregates into the crop-specific profits. As such the enterprise-wide water demand can also be disaggregated into crop-land level water demand (Moore et al. 1994).

3.2 Examining irrigation decisions arising from change in total marginal cost of water

To understand the impact of water scarcity on agricultural choices by irrigators, I investigate the comparative statics on the optimal enterprise-wide (firm) groundwater demand function. Write this function as the sum of crop-land level water demand as follows:

$$TW = \sum_{l=0}^{L} w_l\Big(p_l, \mathbf{r}, B(s,b), n_l^*\big(\mathbf{p}, \mathbf{r}, B(s,b), N_f; z(a), \mathbf{x}\big); z(a), \mathbf{x}\Big). \tag{8}$$

To analyze the mechanisms by which producers react to changes in cost of groundwater, I take total derivative of (8) with respect to the cost of groundwater (B). This produces:

$$\frac{\mathrm{d}TW}{\mathrm{d}B} = \sum_{l=0}^{L} \left(\frac{\partial w_l(\cdot)}{\partial B(\cdot)} + \frac{\partial w_l(\cdot)}{\partial n_l^*(\cdot)} \cdot \frac{\partial n_l^*(\cdot)}{\partial B(\cdot)} \right). \tag{9}$$

Equation (9) shows that a producer's response to an increase in groundwater cost has two components. The first term on the right hand side of (9) reflects a change in the amount of water used in irrigating each crop-land already in cultivation. This is referred to in the literature as the response at the *intensive margin*. The second term, referred to as the *extensive margin*, shows a further response through cropland reallocation.

The intensive margin shows how irrigators react in the short run to a change in cost of water. For example, given an increase in cost of water per unit pumped, an irrigator may reduce the amount of water applied to a specific crop-land by reducing irrigation frequency. Long-run adjustments include adjustments to the types and acreages of planted crops. An increase in water cost could induce a reduction of acreage for water-intensive crops, a shift to less water-intensive crops, or even a decision to fallow more parcels of land. In similar breath, depending on prevailing conditions, another profit-maximizing option could be to adopt a water-intensive crop that is extremely profitable. This is especially the case where irrigators are able to switch to precision irrigation technologies with higher profitability relative to traditional irrigation methods (Schoengold et al. 2006). Thus, two points are worth noting about the extensive margin. First, these adjustments involve switching costs (e.g., investing in new irrigation technology), require longer time to implement, and therefore cannot be made immediately following a rise in water cost. Second, the contribution of the extensive margin to change in

water use can be negative or positive since the effect water cost has on crop-land choice, as shown above, can be negative or positive.

Next, we decompose the nature of the response to examine the impact of an explicit change in the pumping fee, b. Previous work in this area models total water demand in Eq. (8) with $B(s, b)$ replaced by only b—a constant marginal cost (See, for example, Moore et al. 1994). Differentiation with respect to the fee b then yields a function identical to that of Eq. (9), but with b in place of B. However, when we additionally consider water scarcity through $B(s, b)$, then the change with respect to the fee by itself is:

$$\frac{\mathrm{d}TW}{\mathrm{d}b} = \sum_{l=0}^{L} \left(\frac{\partial w_l(\cdot)}{\partial B(s,b)} \cdot \frac{\partial B(s,b)}{\partial b} + \frac{\partial w_l(\cdot)}{\partial n_l^*(\cdot)} \cdot \frac{\partial n_l^*(\cdot)}{\partial B(s,b)} \cdot \frac{\partial B(s,b)}{\partial b} \right)$$

(10)

$$= \sum_{l=0}^{L} \left(\frac{\partial w_l(\cdot)}{\partial B(s,b)} + \frac{\partial w_l(\cdot)}{\partial n_l^*(\cdot)} \cdot \frac{\partial n_l^*(\cdot)}{\partial B(s,b)} \right) \left(\frac{\partial B(s,b)}{\partial b} \right).$$

This equation essentially scales Eq. (9) by the exclusive effect of a change in the pumping fee on the cost of groundwater. Earlier, we assume that $\frac{\partial B(s,b)}{\partial b}$ or $B_b(s, b) > 0$. If in absolute terms, the change in total marginal cost of water is unit proportional to the change in fee (i.e. $B_b(s, b) = 1$), then (10) equals (9), implying evaluating the impact of a change in the pumping fee on water use is equivalent to doing same with change in total marginal cost of groundwater. However, for absolute values of $B_b(s, b) > 1$, the change in total marginal cost due to a change in b scales upward the magnitude or the absolute value of the change in water use when considering the entire/total change in marginal cost of water $B(s,b)$ as in (9)—that is, when factoring s in the analysis. Conversely, if $B_b(s, b)$ is such that $0 < |B_b(s, b)| < 1$, we have the opposite effect—that is, the change in total marginal cost due to a change in b scales downward the magnitude or the absolute value of the change in water use when considering the entire/total change in marginal cost of water $B(s, b)$.

From these results, it can be argued that estimates of the impact of price-based groundwater conservation policy that only consider the constant component of the total marginal cost of groundwater might be over/under estimating the impact depending on the nature of site-specific water availability (the spatio-temporal variation in the variable component of the total marginal cost). The total response to an increase in the constant marginal cost works indirectly through the effect such a change has on the total marginal cost, which also encompasses the changes in water scarcity measures and how they impact water withdrawal behavior among irrigators.[6]

Finally, with respect to the direction of the total effect of the policy variable b, note that even though the intensive margin is expected to explicitly fall with increase

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[6] The relationship is clearer by recognizing that from the total derivative of $B(s, b)$, we have $\frac{\mathrm{d}B(\cdot)}{\mathrm{d}b} = \frac{\partial B(\cdot)}{\partial s} \cdot \frac{\mathrm{d}s}{\mathrm{d}b} + \frac{\partial B(\cdot)}{\partial b}$.

in water cost, the same cannot be said of the extensive margin. Observe that $\frac{\partial n_l^*(\cdot)}{\partial B(s,b)}$ can be negative or positive, depending on irrigators' expectation of profit levels in relation to specific crop and irrigation technology choices. This makes (9) and (10) analytically indeterminate and, as such, the total response of crop-land level water demand to change in water cost is considered an empirical issue.

4 Study region and data

4.1 Pumping fee implementation in San Luis Valley, Colorado

The data studied in this paper come from a groundwater conservation program for some irrigators in the San Luis Valley (SLV) of Colorado. Due to severe aridity coupled with sustained high levels of groundwater withdrawals, water levels in wells have dropped significantly, raising sustainability concerns. This led to the creation of six sub-districts of the Rio Grande Water Conservation District (RGWCD) within the Colorado portion of the Rio Grande basin (also known as Division 3) of the SLV.[7] The ultimate goal is to ensure that irrigation water conservation policies are put in place by a collective action among the irrigators in each subdistrict. Among the six subdistricts, Special Groundwater Subdistrict No.1 (shown in Fig. 3) was first to be legally recognized in 2006 and subsequently implemented a policy which entailed a fee of \$45 per acre-foot of water pumped at the start of the 2011 irrigation season. The pumping fee was further increased to \$75 per acre-foot of water pumped in 2012.

As of the end of the 2013 farming season, which serves as the cutoff point for this study, none of the other five districts have implemented any such policy. This generates a rare quasi-experiment in which the Sub-district no.1 serves as a treated group, while the other five sub-districts serve as a control group and thus provides the opportunity to employ a difference-in-difference econometric framework, to analyze the effect of the pricing policy.

It is important to note that the division of the subdistricts was completed taking cognizance of the spatial interconnectedness among wells in the same sub-district. Wells in a particular subdistrict were determined to be hydrologically independent of wells in other sub-districts (Smith et al. 2017). As such, in analyzing a causal impact of the policy on farmers' irrigation behavior in a difference-in-difference framework, the unconfoundedness assumption holds because a farmer's membership of a sub-district is not due to self-selection.

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[7] A seventh sub-district was also formed, known as the Trinchera Groundwater Management Subdistrict, which is managed under the Trinchera Water Conservancy District (TWCD). The data for this study covers only RGWCD and not the TWCD.
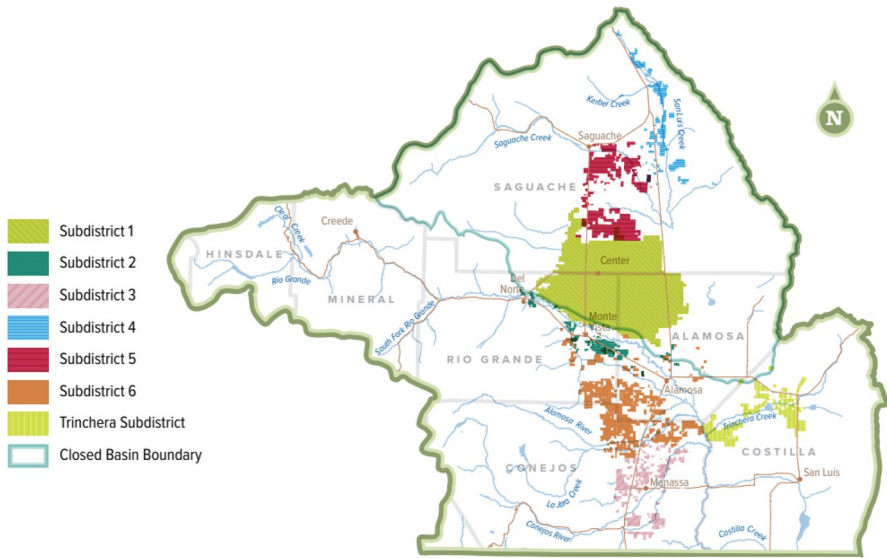
Fig. 3 Groundwater management subdistricts of the RGWCD and TWCD. *Source*: Rio Grande Basin Implementation Plan, January 2022

4.2 Data aggregation

The raw data used in this study primarily come from the Colorado's Decision Support Systems (CDSS), while their aggregation and further update were inspired by Smith et al. (2017). The primary variables are from annual diversion records for wells and ditches maintained by the CDSS in their HydroBase platform as well as irrigated land acreages provided in geospatial databases covering the years 1936, 1998, 2002, and 2005 through 2013.[8] Well-specific attributes including well depth, elevation, and decreed flow rates are also contained in the HydroBase data. I rely mainly on a sample of the data covering 2009–2013 as this matches with the wells data which has complete recordings especially for acre-foot pumped, elevation, and groundwater height from which the depth-to-water variable is derived.

Before delving into the summary statistics of the data utilized in this study, it is important to draw attention to certain relevant concerns regarding the data and its aggregation process. This will serve as a means to facilitate comprehension of the summary statistics.

Upon aligning the well data with the irrigated parcels, it becomes apparent that certain parcels are irrigated using water from multiple wells within a given season. Furthermore, there are instances where a single well irrigates two or more parcels during a season. Given the absence of records indicating the precise amount of water

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[8] Available online: https://www.colorado.gov/pacific/cdss/division-3-rio-grande.

applied to each parcel, an assumption is made for parcels irrigated by the same well, wherein an equal division of water is considered. As for parcels irrigated by multiple wells, the aggregation of these evenly divided amounts (shares) is then regarded as the quantity of water employed for irrigation purposes.

In addition to groundwater volume, another variable of potential importance in the decision-making process of groundwater irrigation is the availability of surface water. These data are extracted from the geospatial database pertaining to irrigated land. The database provides a connection between identifiers for surface water sources and the parcels they serve. However, similar to the situation with wells, direct records specifying the amount of water from each ditch used for irrigating individual parcels are unavailable. Consequently, the same approach of equal sharing, employed in the analysis of well-to-parcel relationships, is likewise applied in this scenario.

In aggregating the data into analyzable units, one method would be to consider parcel (crop-land) level analysis since each parcel grows a single crop per season. However, when using lag of depth-to-water as an instrumental variable for depth-to-water, it is infeasible to use this type of aggregation despite the options it provides for intra-crop analysis. I therefore adopt an aggregation method by Smith et al. (2017) that involves linking of all parcels irrigated by the same well to form initial units and then linking those units that have parcels in common to produce one final unit. This process is repeated across time to produce a panel of time-consistent fixed units that encompass a set of wells and all the parcels they irrigate.

4.3 Descriptive statistics

In Table 1, I present a summary of the outcome and the control variables for the 2009–2013 period for the sample of units that are served by wells for which complete records exist for deriving the lag of the *depth-to-water* variable and the main dependent variable, *Acre-feet pumped. Acre-feet pumped* is the total groundwater withdrawal in acre-feet by irrigators in the San Luis Valley for a growing season. Irrigators in the sample on average withdrew 263.46 acre-feet and irrigated 170.28 acres. Also, the amount of water used per acre *(Acre-feet pumped/acre)* on average is 1.65 acre-feet/acre.

Out of the 170 acres irrigated on average, 166 of them were by sprinkler irrigation technology as opposed to other technologies like flood irrigation. This suggests that most parcels have adopted what is considered a more efficient water-saving technology. Among the control variables, *Depth-to-water (ft.)*, *Well decreed flow rate (in Cubic Feet per Seconds, CFS)*, *Well depth (ft.)*, and *Surface water (AF)*, show mean values less than the standard deviation.[9] This

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[9] The well decreed flow rate is the maximum allowable flow rate at which water can be pumped from a well.

Table 1 Descriptive statistics

	Observations	Mean	Standard Dev.
Panel A: major variables			
Acre-feet pumped	8,355	263.46	229.24
Acre-feet/acre pumped	8,355	1.65	1.41
Acreage irrigated	8,355	170.28	94.41
Sprinkler acreage	8,355	166.06	97.64
Alfalfa acreage	2,467	131.24	73.52
Grass pasture acreage	849	78.32	32.64
Potato acreage	2,832	163.88	92.45
Small grains acreage	3,030	149.22	87.74
Wheat and other crops acreage	695	133.21	42.91
Fallow acreage	3,307	30.59	63.86
Depth-to-water (ft.)	8,355	27.93	31.91
Well decreed flow rate (CFS)[†]	8,355	5.15	5.29
Well elevation (ft.)	8,355	7,642.71	73.77
Groundwater height (ft.)	8,355	7,614.79	63.59
Well depth (ft.)	8,355	186.78	231.45
Wells	8,355	2.18	1.44
Surface water (AF)	8,355	90,332.06	124,142.92
Surface water/acre (AF)	8,355	158.71	307.87
No ditch service	8,355	0.17	0.37

	Observations	No. of units	
Panel B: Counts of units by treatment type			
Treated District Membership			
Pre-intervention	2,327	1,197	
Post-intervention	3,607	1,221	
Control Districts Membership			
Pre-intervention	869	511	
Post-intervention	1,552	574	

Data summarized only for unit-years that have Depth-to-water variable record to calculate lags. ft, feet; CFS, cubic feet per second; AF, acre-foot. [†]The well decreed flow rate is the maximum flow that can be pumped from a well

suggests significant variation about the mean and requires particular attention as we investigate their impact on water withdrawal.

The variation in depth-to-water noticed among units in the data sample across space and time is of particular importance. Since cost of lift is affected by depth-to-water, it is imperative to study how irrigators respond to conservation policies such as constant per unit pumping fees given the variation in depth-to-water at the wells that serve their parcels.

5 Empirical identification and estimation

This section empirically tests the proposition that differences in water scarcity change how total water demand responds to the introduction of a pumping fee. Specifically, I run two tests. First, I test the use of depth-to-water as a cost-metric water scarcity measure and its importance in the estimation of a pumping fee's effect. Leaving out water scarcity measures such as depth-to-water in the design and evaluation of the impact of a marginal pumping fee may result in estimates that are significantly different in comparison to models that do incorporate such measure, which may lead to misguided policy choices. Second, I test the proposition that the constant marginal fee will differentially affect irrigators' water use based on the depth-to-water levels from which they pump. This is achieved by interacting the pumping fee policy and depth-to-water level variables and observing the significance of their joint effect.

5.1 Estimation using levels of depth-to-water—mean fixed effects

Using the regular fixed-effects model in the spirit of Smith et al. (2017), and based on the water demand function in Eq. (8), I estimate a set of five regression models for each of the three outcome variables (*acre-feet pumped*, *acre-feet pumped per acre* and *acres irrigated*). I compare the models to examine how the effect of the pumping fee on water use differs across the specifications with the incorporation of depth-to-water into the analysis. This helps in establishing how a cost-metric scarcity measure like depth-to-water interacts with the pumping fee policy variable and whether this interaction significantly affects the performance of the policy in curbing water use. The first is a baseline model that does not factor in depth-to-water, the second and third models incorporate the depth-to-water variable and with its instrument respectively. The fourth model incorporates depth-to-water and its interaction with the *Tax_Treated* (policy) variable, and the fifth is the instrumental variable (IV) version of the fourth model. To examine the relevance of depth-to-water in estimating the impact of the policy on irrigation groundwater demand, I adopt the following parametric specification:

$$W_{idt} = \beta_0 + \beta_1 s_{it} + \beta_2 D_{idt} + \beta_3 s_{it} \cdot D_{idt} + \beta_4' x_{dt} + c_i + \pi_t + \epsilon_{idt}. \tag{11}$$

The response variable W_{idt} is the total acre-feet (AF) of groundwater withdrawal per irrigation season t by irrigating unit I that is also served by surface water ditch d. The policy variable D is a binary indicator that takes the value one for membership of the treatment group (i.e., Subdistrict No.1) of a unit observed post-intervention (2011–2013).[10] This indicator is also interacted with the water-scarcity variable s (i.e., depth-to-water) in an attempt to reflect the fact that due to spatio-temporal connections, some of the impact the pumping fee has on total groundwater withdrawals is transmitted through the adjustments irrigators make as a result of the changes

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[10] The binary indicator, D is renamed *Tax_Treated* in the results tables to be more informative.

in total marginal cost of groundwater extraction, emanating from the interaction between the pumping fee b and water-stress measure s. The full impact of the pumping fee policy is captured by $\beta_2 + \beta_3 s$, where β_2 alone will be the effect of the policy where s is either zero or the same across unit-years and thus not seriously taken into consideration when withdrawing water. The coefficient β_3 will then be the rate at which the marginal effect of the policy increases per unit increase in s. c_i captures unit-fixed effects, including decreed well permit flows, well depth and an indicator for whether the unit is served by a ditch. π_t represents year-fixed effects accounting for variations in factors that affect all irrigators in the same manner including output and input prices, except water.

An important factor that affects the amount of groundwater withdrawal is the availability of surface water x_{dt}. Surface water could be seen as perfect alternative to groundwater, raising a potential simultaneity bias. It is for this reason that the San Luis Valley (SLV) makes an appropriate area of study. The SLV lacks readily available surface water; furthermore, what surface water is available is appropriated with a "use-it-or-lose-it" feature attached. This feature implies one can only use groundwater after exhausting their surface water supplies, otherwise you lose your right to it. For these reasons, surface water does not constitute a substitute to groundwater. Furthermore, the major source of what surface water is available is snowpack or precipitation, which are not determined in any way by the amount of groundwater demand. As such surface water is duly deemed exogenous.

The model in Eq. (11) does, however, have a major source of endogeneity. The water scarcity measure (s_{it}) and the outcome variable of groundwater use (W_{idt}) simultaneously determine each other. Withdrawals decrease the water table and hence the height of groundwater, which increases depth-to-water. As depth-to-water increases, it raises cost of extraction and this in turns affects the demand (groundwater withdrawal). To overcome this problem, I adopt a two-stage least squares Instrumental Variables (IV) regression.

It is quite difficult to find an outright exogenous variable as IV for depth-to-water. In the absence of such a variable, I consider the one period lag of depth-to-water (s_{it-1}) as its instrument. The choice of this variable is informed by the fact that in terms of exogeneity, the previous period's depth-to-water cannot be affected by irrigators' withdrawal in the current period. Meanwhile depth-to-water at time t is expected to correlate with depth-to-water at time $t-1$. Thus, I expect s_{it-1} to be uncorrelated with the random term ϵ_{idt}, clustered at the unit level. Additionally, to estimate effects at the intensive and extensive margins, the model in (11) is estimated by replacing total withdrawal W_{idt} with acre-feet pumped/acre and acres irrigated, respectively.

Finally, it is worth noting that, nested in Eq. (11) is the baseline model which excludes the water scarcity measure—i.e., where (s_{it}) equals zero.

6 Results

In this section, I present the results from the various regression models considered. For each outcome variable, I compare results from the inclusion of depth-to-water and its interaction with the policy variable in the model, first without accounting for

endogeneity and then with one that accounts for endogeneity (using the first lag of depth-to-water as an instrument).

6.1 Results of overall groundwater withdrawal—using levels of depth-to-water

The baseline model estimates are presented in Column (1) of Table 2, while results of the impact of the pumping fee policy on groundwater withdrawal from the regressions using levels of depth-to-water are presented in Columns (2) through (5). The results for the intensive (acre-feet/acre pumped) and extensive (acres irrigated) margins for this specification are reported in Tables 4 and 5.

Just as found by Smith et al. (2017), the difference-in-difference estimator (the coefficient on *Tax_Treated*, the policy variable) in Table 2 is negative and highly significant across all model specifications, demonstrating that irrigation units in the treated subdistrict reduced their average level of withdrawal following the pumping fee intervention. Columns (2) through (5) of Table 2 add depth-to-water and show that not only is depth-to-water statistically significant in explaining variations in groundwater withdrawals with the inception of the pumping fee, but its inclusion results in reduction in the magnitude of the estimated impact of the policy on groundwater withdrawals.

Furthermore, since the Instrumental Variable (IV) specification provides for better identification of the relevance of the depth-to-water variable and its interactions with the policy variable, the sharp decrease in the magnitude of the policy's effect vis-à-vis the increase in the magnitude of depth-to-water's effect in the IV models is instructive. First-stage regression result for the IV specifications presented in Table 3 shows the instrument is relevant and valid. By shifting to the sign on the coefficient estimate of depth-to-water, we may be able to understand what could be driving the reduction in magnitude of the estimated impact of the policy with the introduction of depth-to-water into the model.

Columns (2)–(5) of Table 2 demonstrate that depth-to-water actually has a positive effect on total water withdrawals in the sample. This would appear contrary to the expectation that increases in depth-to-water would be associated with reduced pumping. This indicates some effect not yet understood in the theoretical model. In fact, irrigation from groundwater commons may be of a strategic nature. If irrigators attribute a drop in pumping water level to increased pumping by their neighbors, irrigators may pump more in hope of extending the cone of depression around their well so as to direct the flow of water into their well at a faster rate. For instance, in a recent work specifically on San Luis Valley, Smith (2018) found that pumping by neighboring irrigators within a radius of quarter mile leads to about 0.56 feet drop in groundwater level per year.

Regardless, the evidence shows that depth-to-water is such a strong factor influencing pumping decisions such that its exclusion from the model may introduce an omitted variable bias in the estimated impact of the policy. Additionally, ignoring the endogeneity of depth-to-water also over-estimates the impact of the policy as

Table 2 Results for total withdrawals: mean effects with levels of depth-to-water

Variables	(1) Baseline model–no depth-to-water	(2) Depth-to-water model	(3) Depth-to-water model-IV	(4) Interacted depth-to-water model	(5) Interacted depth-to-water model-IV
	AF pumped	AF pumped	AF pumped	AF pumped	AF pumped
Tax_Treated	− 98.2503***	− 95.4653***	− 72.9990***	− 87.8141***	− 54.9187***
	(9.0320)	(9.0191)	(13.0305)	(9.9258)	(17.5493)
Depth-to-water		1.0624***	9.6330***	1.1819***	9.4124***
		(0.3625)	(3.2637)	(0.3845)	(3.1747)
Tax_Treated × Depth-to-water				− 0.3043**	− 0.7736*
				(0.1519)	(0.4060)
Surface water	0.0011***	0.0011***	0.0010**	0.0011***	0.0010**
	(0.0004)	(0.0004)	(0.0004)	(0.0004)	(0.0004)
Surface water × Decree	− 0.0004	− 0.0004	− 0.0003	− 0.0004	− 0.0003
	(0.0004)	(0.0004)	(0.0004)	(0.0004)	(0.0004)
Observations	8,355	8,355	8,355	8,355	8,355
R-squared	0.7493	0.7500	0.0166†	0.7501	0.0328†
Number of clusters	1746	1746	1746	1746	1746
Unit FE	Yes	Yes	Yes	Yes	Yes
Year FE	Yes	Yes	Yes	Yes	Yes
Ditch FE	Yes	Yes	Yes	Yes	Yes

Fixed-effect estimates for Eq. (11) are presented in Column 4 and Column 5 shows its IV version using one lag values of depth-to-water as its instruments. Columns 2 and 3 are similar to Column 4 and Column 5, except that they do not have the interaction between the policy and depth-to-water variable. Column 1 is the baseline model that does not use depth-to-water. Surface water × Decree is interaction for surface ditches with special recharge decrees. Robust standard errors clustered by unit, are in parentheses. $^{*}p < 0.1$; $^{**}p < 0.05$; $^{***}p < 0.01$. †These are centered R-squared

Table 3 First-stage results of the IV estimations for total withdrawals: mean effects with levels of depth-to-water

Variables	(1)	(2)	
	Depth-to-water model-IV	Interacted depth-to-water model-IV	
	Depth-to-water	Depth-to-water	Tax_ Treated × Depth-to-water
Lag_Depth-to-water	− 0.117017***	− 0.124319***	− 0.085242***
	(0.026178)	(0.026618)	(0.011575)
Tax_Treated × Lag_Depth-to-water		0.059777*	0.970994***
		(0.030784)	(0.013028)
Tax_Treated	− 2.483696***	− 3.793932***	2.795310***
	(0.552511)	(0.831046)	(0.292881)
Surface water	0.000015**	0.000015**	0.000003**
	(0.000007)	(0.000007)	(0.000001)
Surface water × Decree	− 0.000018**	− 0.000018**	− 0.000004**
	(0.000007)	(0.000007)	(0.000002)
Observations	8,355	8,355	8,355
R-squared	0.972617	0.972785	0.967793
Number of clusters	1746	1746	1746
Unit FE	Yes	Yes	Yes
Year FE	Yes	Yes	Yes
Ditch FE	Yes	Yes	Yes

First-stage regression results for the IV fixed-effect estimates for Eq. (11) presented in Table 2, Column (3) and Column (5). Surface water × Decree is interaction for surface ditches with special recharge decrees. Robust standard errors clustered by unit, are in parentheses. $^*p < 0.1$; $^{**}p < 0.05$; $^{***}p < 0.01$

seen in the comparison between Columns (2) and (3) and between Columns (4) and (5).

The reduced pumping also suggests that the imposed fee actually curtails the influence of depth-to-water on pumping. Indeed, running the models without the policy variable shows statistically significant and increased positive estimates for depth-to-water.[11] To thoroughly examine the nature of the interaction between the fee policy and depth-to-water and how such interaction impacts ultimately on pumping levels, I turn to specifications (4) and (5) which include an interaction between the fee policy (*Tax_Treated*) and *Depth-to-water* variables.

Before proceeding further, it is important to note that it would be inappropriate to make direct comparison between the models in Columns (2) and (4), and between models in Columns (3) and (5). This is because the interaction models can only be equivalent to the non-interacted models for zero depth-to-water value—an uninteresting case with no instantiations in this study. In these interaction models, the coefficient on *Tax_Treated × Depth-to-water*, captures the relative rate of change in the marginal effect of the policy on pumping per unit (one foot) increase

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[11] These results are not presented in the paper but can be produced upon request.

Table 4 Results for intensive margin adjustment: mean effects with levels of depth-to-water

Variables	(1) Baseline model–no depth-to-water	(2) Depth-to-water model	(3) Depth-to-water model-IV	(4) Interacted depth-to-water model	(5) Interacted depth-to-water model-IV
	AF pumped/acre	AF pumped/acre	AF pumped/acre	AF pumped/acre	AF pumped/acre
Tax_Treated	− 0.593280***	− 0.584400***	− 0.432688***	− 0.533279***	− 0.333217***
	(0.063957)	(0.063476)	(0.096621)	(0.066242)	(0.111810)
Depth_to_water		0.003388	0.061263***	0.004186*	0.060050***
		(0.002240)	(0.021584)	(0.002468)	(0.021169)
Tax_Treated × Depth_to_water				− 0.002033*	− 0.004256*
				(0.001225)	(0.002339)
Surface water	0.000006***	0.000006***	0.000005**	0.000006***	0.000005**
	(0.000002)	(0.000002)	(0.000002)	(0.000002)	(0.000002)
Surface water × Decree	− 0.000002	− 0.000002	− 0.000001	− 0.000002	− 0.000001
	(0.000002)	(0.000002)	(0.000002)	(0.000002)	(0.000002)
Observations	8,355	8,355	8,355	8,355	8,355
R-squared	0.705446	0.705574	0.035939†	0.705656	0.02463†
Number of clusters	1746	1746	1746	1746	1746
Unit FE	Yes	Yes	Yes	Yes	Yes
Year FE	Yes	Yes	Yes	Yes	Yes
Ditch FE	Yes	Yes	Yes	Yes	Yes

Fixed-effect estimates for Eq. (11) using AF pumped per acre as the dependent variable is presented in Column 4 and Column 5 shows its IV version using one lag values of depth-to-water as its instruments. Columns 2 and 3 are similar to Column 4 and Column 5, except that they do not have the interaction between the policy and depth-to-water variable. Column 1 is the baseline model that does not use depth-to-water. Robust standard errors clustered by unit, are in parentheses. $*p < 0.1$; $**p < 0.05$; $***p < 0.01$. †These are centered R-squared

in depth-to-water. The fact that this coefficient on *Tax_Treated* × *Depth-to-water* is non-zero speaks to some incidence of spatial (across wells) and temporal (across time) variations in depth-to-water levels. Additionally, the fact that this coefficient is negative implies the policy yields larger reductions in pumping at higher depth-to-water levels. However, the nature of the specification of these interaction models is such that there is a separate effect of the policy for every value of depth-to-water. For example, using the coefficients from Column (5), a well with a depth-to-water of 20 feet would reduce pumping in response to the fee by $-54.9 - 0.77 \times 20$. Now, this does not make the estimates to readily lend themselves to a useful interpretation without specifying some interesting values of depth-to-water. Often the practice in such situations is to specify values such as the mean, or the lower and upper quartiles in the sample (Wooldridge 2010).

With this caveat in mind, a comparison of the fixed-effect (FE) and the instrumental variable (IV) FE versions of the interacted model reflects how pronounced the impact of depth-to-water can be, as well as how much the impact of the policy depends on depth-to-water (as reflected by the coefficient on *Tax_Treated* × *Depth-to-water*). In the properly identified model, the IV model in Column (5), irrigators increase pumping by about 9 *acre-feet* in response to a *one foot* increase in depth-to-water—about 8 *acre-feet* more compared to the model in Column 4 which does not account for endogeneity. The interaction effect, the coefficient on *Tax_Treated* × Depth-to-water, more than doubled from -0.3 *acre-foot* to -0.77 *acre-foot*. These results indicate that the FE model without the IV underestimates the impact of depth-to-water as well as the part of the effect of the pumping fee policy that is due to depth-to-water. It is instructive to note that, regardless of whichever values we assign to depth-to-water from the sample, it is the case that the pumping fee policy and depth-to-water interact in a way that curtails their respective impact on pumping independent of each other. But the impact of the policy in reducing pumping is felt more at higher depth-to-water levels. Additionally, the effect of the depth-to-water is strong enough when evaluated at values such as the 25th percentile (13.7 *feet*), median (21.5 *feet*), mean (27.9 *feet*) and even 75th percentile (32.26 *feet*) to warrant the significant upward bias in the estimated impact of the policy on pumping (as observed across Table 2) should it be excluded from the model.

6.2 Adjustment at the intensive and extensive margins—using levels of depth-to-water

The result of the regression examining the intensive margin adjustments using depth-to-water levels is presented in Table 4 while that for the extensive margins is presented in Table 5. As stated earlier, the intensive margin examines the short-run adjustment by irrigators to the fee policy while the extensive margin reflects long-term responses in terms of acres of crop-land irrigated. In the intensive margin regression, *acre-feet/acre* replaces *acre-feet pumped* as the outcome variable in Eq. (11). The extensive margin regression uses acres cultivated as the dependent variable in Eq. (11).

With respect to the short-run adjustments, the results in Table 4 follow exactly the observations made in the over-all water use analysis in Table 2. Across all models, irrigators respond to the fee by using less water per acre irrigated. The baseline model

Table 5 Results for extensive margin adjustment: mean effects with levels of depth-to-water

Variables	(1) Baseline model—no depth-to-water	(2) Depth-to-water model	(3) Depth-to-water model–IV	(4) Interacted Depth-to-water model	(5) Interacted Depth-to-water model-IV
	acres	acres	acres	acres	acres
Tax_Treated	− 14.667649***	− 13.438788***	− 14.458999***	− 18.843325***	− 18.630289***
	(1.939179)	(1.923105)	(3.137484)	(2.455174)	(4.292751)
Depth-to-water		0.468792***	0.079597	0.384366***	0.130493
		(0.081667)	(0.615299)	(0.079266)	(0.606653)
Tax_Treated × Depth-to-water				0.214914***	0.178488**
				(0.049411)	(0.079735)
Surface water	0.000147**	0.000141**	0.000146**	0.000142**	0.000145**
	(0.000060)	(0.000062)	(0.000073)	(0.000062)	(0.000073)
Surface water × Decree	− 0.000102*	− 0.000095	− 0.000101	− 0.000094	− 0.000098
	(0.000061)	(0.000062)	(0.000073)	(0.000062)	(0.000073)
Observations	8,355	8,355	8,355	8,355	8,355
R-squared	0.891767	0.892463	0.106070†	0.892725	0.110260†
Number of clusters	1746	1746	1746	1746	1746
Unit FE	Yes	Yes	Yes	Yes	Yes
Year FE	Yes	Yes	Yes	Yes	Yes
Ditch FE	Yes	Yes	Yes	Yes	Yes

Fixed-effect estimates for Eq. (11) using acres irrigated as the dependent variable is presented in Column 4 and Column 5 shows its IV version using one lag values of depth-to-water as its instruments. Columns 2 and 3 are similar to Column 4 and Column 5, except that they do not have the interaction between the policy and depth-to-water variable. Column 1 is the baseline model that does not use depth-to-water. Robust standard errors clustered by unit, are in parentheses. $*p < 0.1$; $**p < 0.05$; $***p < 0.01$ †These are centered R-squared

overestimates the amount of water used per acre on average compared to models that incorporate the scarcity measure. Depth-to-water is statistically significant in the interacted models with similar sign as in the over-all water use response analysis.

On the extensive margin adjustment, not much difference is seen comparing the baseline model with the relevant depth-to-water models. There is, however, a little evidence of the baseline underestimating the effects of the pumping fee on acres irrigated when considering interacted models (Table 5).

6.3 Robustness check

At the beginning of the 2011 growing season when the pumping fee policy was implemented, the fee was pegged at \$45 per acre-foot of water withdrawn. This, however, was increased to \$75 per acre-foot starting in 2012. Thus, the policy can be said to be staggered. I redefine the policy variable accordingly to examine whether there is significant change in the results as found using the binary indicator variable D (i.e., *Tax_Treated*). Specifically, D is redefined as a

Table 6 Results for staggered policy effects on over-all water use

Variables	(1)	(2)	(3)
	Depth-to-water Model	Interacted Depth-to-water Model	Interacted Depth-to-water Model-IV
	AF pumped	AF pumped	AF pumped
1.Tax_Treated1	−74.8072***	−59.3363***	−32.2729*
	(10.9525)	(12.0720)	(16.7088)
2.Tax_Treated1	−105.3192***	−103.8005***	−70.6004***
	(9.6864)	(10.7911)	(18.5143)
Depth-to-water	1.0655***	1.1079***	9.1335***
	(0.3618)	(0.3867)	(3.0925)
1.Tax_Treated1 × Depth-to-water		−0.6522***	−0.9289***
		(0.1905)	(0.3479)
2.Tax_Treated1 × Depth-to-water		−0.0700	−0.5547**
		(0.1635)	(0.2789)
Surface water	0.0011***	0.0011***	0.0010**
	(0.0004)	(0.0004)	(0.0004)
Surface water × Decree	−0.0004	−0.0004	−0.0003
	(0.0004)	(0.0004)	(0.0004)
Observations	8,355	8,355	8,355
R-squared	0.7505	0.7507	0.0413[†]
Number of units	1746	1746	1746
Unit FE	Yes	Yes	Yes
Year FE	Yes	Yes	Yes
Ditch FE	Yes	Yes	Yes

Fixed-effect estimates for Eq. (11) with staggered policy definition *Tax_Treated1*. Robust standard errors clustered by unit, are in parentheses. *$p < 0.1$; **$p < 0.05$; ***$p < 0.01$ †These are centered R-squared

three-level indicator variable such that $D = 1$ for membership of the treatment group (i.e., Subdistrict No.1) of a unit observed post-intervention in 2011 and $D = 2$ if observed in 2012–2013; $D = 0$ otherwise.

The statistical significance, direction (sign) and magnitude of the estimated effects are similar to the main regression. Additionally, the results show that water reduction levels increase with an increase in the constant per unit pumping fee (Table 6).

7 Conclusion

This paper examined the role of depth-to-water as a cost-metric measure of relative water scarcity in determining how irigators respond to irrigation water pricing in groundwater commons. Using data on irrigation in the San Luis Valley of Southern Colorado, I examined how a constant marginal pumping tax imposed on some irrigators differentially affected their pumping behavior based on the depth at which they pump water from their respective wells. This is only one area for which population growth and climate change have led to increasing reliance of irrigated agriculture on groundwater resources. As conservation and sustainability concerns grow globally, researchers and policy makers have renewed the debate on the use of price mechanisms as a conservation tool. More often than not, where considered, such pricing is designed and modeled as constant per unit fees. This paper shows that, in the context of groundwater commons, such a tool may overlook some important spatio-temporal aspects of pumping among irrigators that affect their water scarcity levels and may be an important consideration in how they respond to pumping fees.

The effect of the constant marginal tax policy on withdrawals is significantly affected by depth-to-water levels such that estimates that do not account for this important cost-metric (depth-to-water) variable are likely to overstate or understate the impact of the policy which may misguide policy decisions.

The results show that depth-to-water has a positive effect on water use contrary to the expectation that higher depth-to-water would be associated with reduced pumping. A potential explanation for this is the fact that irrigation in groundwater commons may be of a strategic nature where the aquifer material allows for seepage—physical movement of water between wells. If irrigators attribute a drop in pumping water level to increased pumping by their neighbors, irrigators may pump more in hope of extending the cone of depression around their well so as to direct the flow of water into their well at a faster rate. Indeed, a couple of studies, namely Pfeiffer and Lin (2012) and Smith (2018), have found that pumping by neighboring irrigators within a radius of a quarter mile up to one mile leads to a drop in groundwater level. This means that depth-to-water for a well at a given time could be a function of water withdrawals from both own and neighboring wells. An important extension of this paper in the future will be to use a game-theoretic model to investigate the nature and extent of spillover effects generated by this spatio-temporal relationship.

Data availability Data may be made available upon reasonable request as it is publicly available from the Colorado's Decision Support Systems (CDSS).

Declarations

Conflict of interest The author has no conflict of interest or funding sources to report.

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