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Land Economics, Volume 97, Number 1, February 2021, pp. 1-16 (Article)

Published by University of Wisconsin Press

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ABSTRACT *An efficient market implies that potential gains from trade are fully captured. Achieving this requires a well-functioning market where prices reflect all available information. In the case of water rights markets, this implies that the permanent water rights transfer price reflects the sum of discounted returns to this asset (i.e., the lease price), the market interest rate, and a risk premium that reflects potential future water scarcity. The purpose of this study is to assess the efficiency of western U.S. water markets by using the asset pricing model to measure how well prices reflect long-run returns to permanent water rights.* (JEL Q21, Q25)

1. Introduction

The number and scale of environmental markets have increased over time with success in programs such as air pollution permit trading (Schmalensee and Stavins 2017) and individual transferable quota systems for fisheries (Costello, Gaines, and Lynham 2008). However, even well-functioning environmental markets often fail to achieve their maximum theoretical advantage over other allocation or regulation mechanisms (Teitenberg 1990; Keohane 2007). The achieved efficiency gains from markets for natural resource assets have been highly variable. Markets for fishing quotas appear to be quite successful in this regard (Grainger and Costello 2011); those for wetland and habitat preservation less so (National Research Council 2001; Parkhurst and Shogren 2003).

This article examines markets for scarce water, advocated by economists over many decades (Hartman and Seastone 1970; Vaux and Howitt 1984; Saliba and Bush 1987). Although informal water markets are common in some developing countries (Bjornlund and McKay 2002), formal intersectoral water markets have been slow to develop (Easter, Rosegrant, and Dinar 1999) and are generally immature (Carey, Sunding, and Zilberman 2002; Brewer et al. 2008), making empirical studies of actual water markets uncommon. Nonetheless, some research has demonstrated potential and realized net benefits from trading in California (Hagerty 2019; Bruno and Jessoe 2021), south Texas (Chang and Griffin 1992), southern Italy and Spain (Pujol, Raggi, and Viaggi 2006; Rey, Garrido, and Calatrava 2014), north-central Chile (Hearne and Easter 1997; Hearne and Donoso 2014), Morocco (Diao and Roe 2003), and Australia (Bjornlund and McKay 2002; Tisdell 2014; Wheeler, Bjornlund, and Loch 2014; Zuo et al. 2015; Grafton, Horne, and Wheeler 2016; Loch, Wheeler, and Settre 2018). Australia's water markets may be especially relevant to western U.S. water markets given that a number of regions in the western United States are following the Australian example in designing of water markets (e.g., Nevada's Diamond

Land Economics • February 2021 • 97 (1): 1–16
ISSN 0023-7639; E-ISSN 1543-8325

Appendix materials are freely available at http://le.uwpress.org and via the links in the electronic version of this article.

Valley and Humboldt Basin; Young 2015; Wheeler et al. 2017; Zeff et al. 2019).

U.S. water market efficiency (or lack thereof) has been examined in prior literature.[1] Brookshire et al. (2004) and Brewer et al. (2008) suggest that U.S. western water markets are becoming more efficient and that water is moving from lower-valued (agricultural) to higher-valued (urban and environmental) uses. But some current water rights transfers in the United States are informal, and even active markets may exhibit high transaction costs (Scott and Coustalin 1995; Huffaker 2005; Rosegrant, Ringler, and Zhu 2014; Wheeler, Bjornlund, and Loch 2014; Hagerty 2019).

Market efficiency requires that prices reflect available information about scarcity and value in use. Therefore, an important area of study—which has been largely missing from the water market literature—is the role of pricing mechanisms in water rights markets. The purpose of this study is to assess the efficiency of western U.S. water rights markets by using the asset pricing model to measure how well prices reflect long-run returns to permanent water rights. We exploit the variation in prices and quantities for water trades in the western United States between 1990 and 2010 to assess water markets' capacity to incorporate available information about long-run returns.

We apply the financial asset pricing model—similar to Newell, Sanchirico, and Kerr (2005) and Newell, Papps, and Sanchirico (2007) in their applications to New Zealand fishing quota markets—to U.S. water rights markets, econometrically estimating a water transfer price equation for nine western U.S. states for which the requisite data are available. The asset pricing model specifies the structural relationship between the permanent transfer price (i.e., the asset price) and the lease price (i.e., returns to the asset), the market interest rate, and a risk premium. Estimating an empirical specification of the asset pricing model (following Newell, Papps, and Sanchirico 2007) allows us to assess the extent to which permanent transfer prices are influenced by these factors, with a central focus on the lease price—the greater the influence of lease price on permanent transfer price controlling for the market interest rate and water scarcity (a measure of risk), the more efficient the market.

To our knowledge, the asset pricing model has not been applied to water markets.[2] Despite the small number of observations in our analysis, results suggest that water transfer prices are positively correlated with lease prices and negatively correlated with interest rates, as asset pricing theory would predict. These results are somewhat surprising. Although Newell, Papps, and Sanchirico (2007) find that prices in markets for fishing quota comport with the asset pricing model, these markets in New Zealand fisheries may be the most well-functioning created markets for natural assets (Grainger and Costello 2014). In water rights markets—where prices are less stable and more heterogeneous, transaction costs are high, and trading is thin—it would not be surprising to find the data to be inconsistent with the asset pricing model (Yoskowitz 1999; Edwards and Libecap 2015).

Recognizing that our data include heterogeneous water markets in nine states, we extend our analysis to examine whether water market prices convey economic information more efficiently in relatively better-functioning markets. We apply the asset pricing model to a small regional market (the Mojave Basin area in California), where water transfers and leases represent trades in more homogeneous goods, trading is more active and better monitored, and transaction costs are probably much lower than in the general case in our multistate analysis. The asset pricing results for the Mojave Basin trades are stronger than those in our nine-state model and even stronger when we focus on the most active regions of that

[1] Truly efficient markets would also fully address externalities and public goods, important considerations in the context of water markets (Olmstead 2010). Our analysis does not consider these potential market failures and instead focuses on the capacity of water market prices to transmit information about the private benefits and costs of water use. Finding that water markets have this capacity would be a necessary but not sufficient condition to support their use (through taxation or other means) to address market failures.

[2] Asset pricing models have been applied to natural resource markets, such as those for agricultural land (Alston 1986) and dairy quota (Wilson and Sumner 2004) as well as fishing quota markets.

market. These results provide reason for optimism about water markets, should barriers to efficient trading be reduced in the future.

2. Asset Pricing Theory and Empirical Models

One implication of rational pricing theory for water market transactions is that the present value of permanent water rights should equal the discounted value of all future expected earnings from annual water leases. With constant lease prices and a constant growth rate, the price of a permanent water right would be as in equation [1], where the interest rate (r) is equal to the expected annual rate of return from holding a water right.

$$p^{sale} = \frac{p^{lease}}{r}. \quad [1]$$

Newell, Papps, and Sanchirico (2007) apply the present value asset pricing model to examine the relationship between fishing quota asset and lease prices. Similarly, we use the Gordon growth model (Campbell, Lo, and MacKinlay 1997), as shown in equation [2], and modify it to conform to the water rights market context:

$$p_t = \frac{\pi_t}{r_t - g}. \quad [2]$$

In equation [2], p_t is the asset price, in our case the permanent water rights transfer price in period t; π is the future annual return from the asset—that is, the one-year lease price; r is the interest rate, and g is a constant, asset-specific growth rate.

Water rights markets should be affected by future expectations about the value of water, influenced by expected climate conditions and institutional settings. As suggested by Alston (1986), Cochrane (1992), and Newell, Papps, and Sanchirico (2007), we decompose r into a real market interest rate ($\tilde{r}$) and a risk premium (θ), which in our water market case accounts for future water supply uncertainty.[3] The variables used to represent these parameters in the theoretical model are described in Section 3. Equation [3] provides the final form of our asset pricing model:

$$p_t = \frac{\pi_t}{\tilde{r}_t + \theta - g}. \quad [3]$$

We empirically estimate the asset pricing model using data from nine western U.S. states where permanent transfers and one-year leases are both prevalent. However, even within a state, regional markets vary, for example, from large federal projects like the Colorado-Big Thompson project in Colorado or the Central Valley project in California, to bilateral transactions between two neighboring farmers. Thus, as a second test of the asset pricing model, we use a unique dataset from a single water market known to be relatively well developed and active, located in the Mojave River Basin, in San Bernardino County, California. We repeat our empirical tests from the nine-state model on the Mojave River Basin market.

As in Newell, Papps, and Sanchirico (2007), in our reduced-form empirical tests of the asset pricing model in both contexts, we regress water transaction prices on a set of explanatory variables as in equation [4],

$$lnp_{ijqt} = \gamma_0 + \gamma_1 ln\pi_{ijqt} + \gamma_2\tilde{r}_t + \gamma_3\theta_j + \gamma_4 g_j + \gamma_5\mu_t + \alpha_j + \varepsilon_{ijqt}, \quad [4]$$

where lnp is the log price of permanent water rights transfer i, in state j, in quarter q, and in year t; $ln\pi$ is the log price for a one-year lease; $\tilde{r}$ is the annual real U.S. market interest rate; θ is a risk premium; g is the growth rate; μ is linear time trend; α is a state fixed effect or random effect; and ε is the error term. In Section 3, we describe the variables used for $\tilde{r}$, θ, and g. In well-functioning markets, prices of permanent transfers and leases may be determined simultaneously, so we estimate alterna-

[3] Ideally, the risk premium would be specific to a particular water right and address institutional uncertainty and climate uncertainty. Another approach would be to add a multiplicative function that includes factors related to uncertainty about the future to the growth model (Newell, Papps, and Sanchirico 2007). However, because of the small number of observations in our empirical analysis, this approach is not possible.

tive models in which we instrument for own-state lease prices using the annual average lease price in all states except the state where a transaction takes place. From the asset pricing theory discussed above, we expect $\gamma_1 > 0$ and $\gamma_2 < 0$. The expected signs of γ_3 and γ_4 will depend on the variables used to proxy for θ and g. Generally, a riskier asset should have a lower price and one with a higher long-term growth rate in expected profits should have a higher price, all else equal.

3. Multistate Sample Data and Asset Pricing Model Estimation

Water Transaction Data

For the dependent variable in the multistate asset pricing analysis, we purchased water market transactions data (permanent water transfers and one-year leases) from Stratecon, Inc. The data come from monthly issues of an industry publication, *Water Strategist* (Smith and Vaughan 1990–1994, 1995–2001; Smith 2002–2010), which has been previously described and analyzed in the literature (Howe and Goemans 2003; Brookshire et al. 2004; Howitt and Hansen 2005; Brown 2006; Brewer et al. 2008; Basta and Colby 2010; Libecap 2010; Colby, Basta, and Adams 2011; Grafton et al. 2012; Hansen, Howitt, and Williams 2013, 2014; Goemans and Pritchet 2014; Olmstead, Fisher-Vanden, and Rimsaite 2016).[4] We omit observations associated with transactions that are outside the scope of our study, for example, those involving recycled wastewater effluent, water storage rights, and multiyear leases as well as those with missing or unreasonable prices (less than $1 per acre-foot), or unidentified buyers. In addition, although market transactions from 12 states appear in the *Water Strategist* data, we include only those nine states with at least some years where both one-year leases and permanent transfers occurred. A minimum number of years with both types of transactions occurring in a state is required to estimate equation [4] with state fixed effects.[5] The final sample comprises 2,158 transactions in nine states, with one water-supplying sector (agriculture) and two water-buying sectors (agriculture and urban) over the period 1990–2010. The *Water Strategist* data product ends in 2010, preventing us from extending the dataset beyond that point.

On average, there are 13 permanent transfers and six one-year leases per year in each state (Figure 1). The average annual number of transactions increased for both permanent transfers and leases until the early 2000s, when both dropped significantly, and both have declined slightly since then. On average, there have been more permanent trades than leases, except in 2001 and again in 2008, when the numbers were very close. Figure 1 also shows that the average annual quantity of water permanently transferred by state is much lower and less variable than the annual average quantity leased. Average state water lease and transfer prices have trended upward over time with significant year-to-year variation (Figure 1).

Summary statistics are reported in Table 1, which reveals significant heterogeneity across state markets. Over the period 1990–2010, most states averaged fewer than four permanent transfers and fewer than four leases per year. Colorado, in contrast, averaged more than 60 transfers annually, and California averaged more than 12 one-year leases annually.

Transfer and lease prices are converted to 2009 dollars using the consumer price index (CPI) and are expressed in dollars per acre-foot ($/AF).[6] As one would expect, the average annual water lease price is much lower than the permanent transfer price for both sectors in all states (Table 1). Price dispersion across states is large, especially in the case of perma-

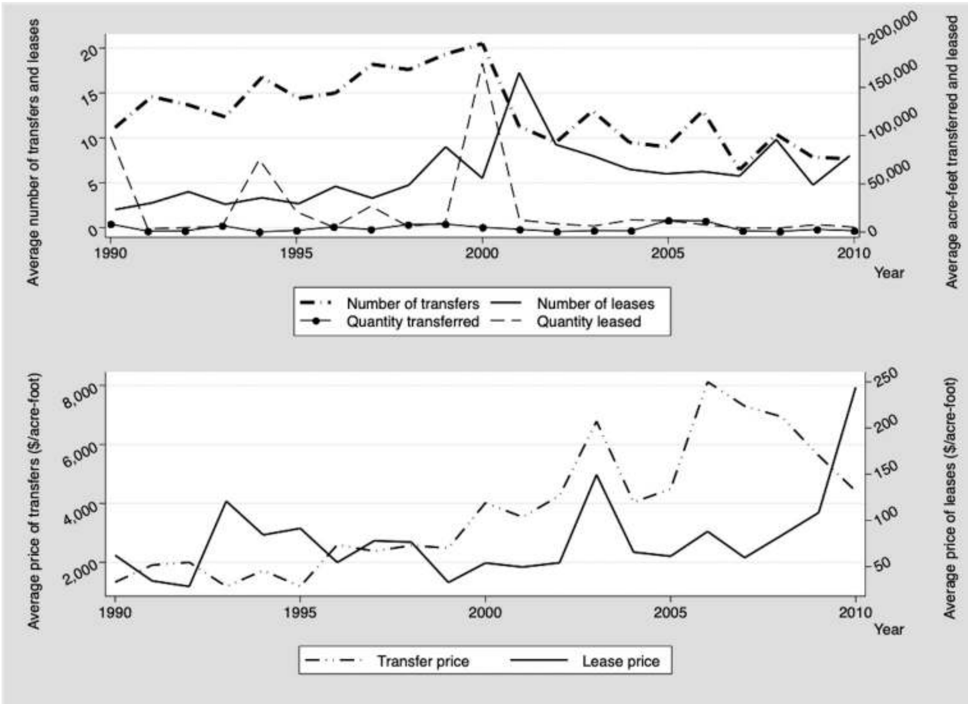
[4] Until 1995, water transactions were reported in a separate publication associated with the journal *Water Intelligence Monthly*. Transactions were reported quarterly from 1995 to 1998 and monthly from 1999 to 2010. Although prior researchers have made these data available publicly, we reconstruct the entire nine-state, 21-year panel to ensure that the summary data for each transaction (culled from descriptive text in .pdf files) are interpreted consistently.

[5] See Appendix section A1 for an additional discussion regarding omitted data. Most states have at least one year where they report zero permanent transfers or zero one-year leases.

[6] An acre-foot, a common unit of volume in U.S. western water trades, is the quantity of water that would flood an acre of land to 1 ft. in depth, about 326,000 gallons.

Figure 1

Average Number of Transactions and Quantity Traded (Top) and Average Prices of Transactions (Bottom) in the Nine-State Water Market Sample



nent transfers. This is not surprising because water cannot be transferred or leased across states, similar to prohibitions in fishing quota markets on trades across regions, species, or species-regions. The highest prices for water leased to the urban sector are found in Texas, but prices for water leased to agriculture are highest in California. Permanent water rights transfers in both sectors are most expensive in Colorado.

To estimate the asset pricing model in equation [4], we aggregate the nine-state water transaction data by state and quarter, so the dependent variable, p_{ijqt}, is the average state-quarter permanent transfer price, and π_{ijqt} is the average state-quarter one-year lease price. This reduces our sample size to 66 observations in the nine states over the period 1992–2009. Together, six out of nine states account for only 13 observations in the panel, whereas the remaining three states—California, Colorado, and Texas—contribute 13, 17, and 23 observations, respectively. Although this small dataset is far from ideal, to our knowledge, the *Water Strategist* is the only available data that summarize such a large and diverse set of U.S. western water trades that could be used to test the asset pricing model at this scale.

Other Data Used in the Multistate Asset Pricing Models

Summary statistics for the remaining variables used in the multistate asset pricing model are reported in Table 2.[7] For the real market interest rate ($\tilde{r}$), we use the three-month U.S. Treasury bill rate from the U.S. Federal Reserve (2019) website. Real interest rates are calculated by subtracting the inflation rate (measured by the CPI) from these nominal interest rates.

[7] Table 2 also reports summary statistics for the variables used in estimating the asset pricing model for the Mojave Basin market. These data are discussed further in Section 4.

Table 1

Annual Average Transactions, Quantities, and Prices by State and for the Full Nine-State Sample, 1990–2010

State	Number of Leases	Number of Transfers	Qty. Leased (AF)			Qty. Transferred (AF)			Lease Price ($/AF)			Transfer Price ($/AF)		
			to urb	to ag	total	to urb	to ag	total	urb	ag	total	urb	ag	total
AZ	2.3	2.6	259,253	237,671	222,668	3,585	825	3,380	103	59	68	1,347	835	1,182
CA	12.1	2.2	14,464	8,523	9,371	12,206	9,978	11,839	130	68	91	2,129	953	1,998
CO	3.4	64.0	7,903	5,304	8,042	148	31	115	100	29	78	10,456	7,184	10,142
ID	3.4	1.7	11,805	15,557	15,583	1,140	1,700	1,604	14	11	12	645	1,095	988
NM	1.0	2.2	n/a	22,455	22,455	164	774	232	n/a	46	46	4,177	2,701	4,013
NV	1.0	2.3	7,700	448	5,283	625	n/a	625	45	43	45	9,243	n/a	9,243
TX	9.1	3.7	1,885	7,413	3,852	10,025	7,217	9,196	224	31	162	1,655	1,288	1,618
UT	1.4	1.8	3,037	9,944	9,312	677	94	538	181	7	30	1,693	1,410	1,566
WA	1.4	1.4	1,320	5,352	5,005	15,209	1,370	15,336	80	34	39	819	266	842
Total	26.7	76.0	10,809	19,505	16,951	884	1,101	849	183	49	109	9,342	5,780	8,944

Note: All statistics are yearly averages weighted by number of transactions that occurred in each year during the period 1990–2010. Most states do not have observations for every year of this period. Number of leases and number of transfers indicate an annual average number of one-year leases and permanent transfers for each state, conditional on a lease/transfer (e.g., AZ had 23 permanent transfers, which occurred during nine years during 1990–2010; hence, the annual average number of transfers is 2.6). Quantity leased and transferred is an annual average transferred to urban, to agricultural, and to both (total) uses. Lease and transfer prices indicate annual average price per acre-foot paid for water to be used in urban, agricultural, and both (total) sectors. Total average (bottom row) is the mean of 21 yearly averages over the period 1990–2010 without state weights (e.g., total of 1,597 transfers across the nine states during 21 years is 76 transfers/year); the total average in the bottom row does not equal the sum of state averages due to the unbalanced panel (e.g., in Arizona, permanent transfers occurred in nine years during the 1990–2010 period, but only eight of those years showed transactions to urban uses, and only three of those years showed transactions to agricultural uses).

Table 2
Summary Statistics for the Asset Pricing Models

	9-State Model				Mojave Market Model			
	Mean	Std. Dev.	Min	Max	Mean	Std. Dev.	Min	Max
Transfer price ($/AF)	4,504.23	6,451.63	76.44	28,298.85	2,108.13	1,794.43	58.51	5,546.25
Lease price ($/AF)	95.25	108.57	1.47	559.48	122.63	123.01	1.00	422.58
Growth Rate								
Farm acres irrigated	0.10	0.35	–0.76	0.75				
Water consumption					0.50	0.26	0.13	0.95
Risk Premium								
Irrigation vulnerability index	0.57	7.53	–2.41	59.47	–20.43	8.63	–42.52	–7.14
Real interest rate	0.01	0.02	–0.02	0.03	–0.00	0.02	–0.03	0.03
Areas (J)	9 states: AZ, CA, CO, ID, NM, NV, TX, UT, WA				5 subareas: Alto, Baja, Centro, Este, Oeste			
Quarters (Q)	4				4			
Years (T)	18 (1992–2009)				24 (1995–2018)			
Observations (N)	66				89			

For the risk premium variable, θ, we use the Irrigation Vulnerability Index (IVI), constructed with the methodology from Liu et al. (2017). The index provides projections of future long-term water stress, which we aggregate by state, dividing the difference between water supply and water use by irrigation water use. In this index, water supply is the sum of surface water (including reservoir storage) and renewable groundwater sources. Water use is the sum of irrigation, domestic, industrial, and livestock water use.[8] Lower values of the IVI indicate higher levels of water stress (< 0.2 is considered stressed). Additional information about the index is provided in the Appendix section A2.

Because the majority of water withdrawals in the region are for irrigation, the growth rate variable (g) in equations [3] and [4] should capture expectations about the future returns to using water as an input to agriculture. We use the growth rate of farmland acres irrigated for this purpose. We know of no projections available for this variable, so we follow Newell, Papps and Sanchirico (2007), using historical data on farmland acres irrigated at the state level from the USDA Irrigation and Water Management Survey (U.S. Department of Agriculture 2019) over 1988–2018 to estimate an AR(1) model expressing growth in value as a function of the natural log of acres irrigated, a time trend, and a constant for each state.[9] This approach assumes that the drivers of future growth in irrigated land value will be consistent with past drivers. Given our approach, the estimate of g varies by state but not over time.

Results from the Multistate Asset Pricing Models

Table 3 reports the results from estimating equation [4] on our multistate sample. Six models are reported in Table 3: three with data from all nine states and, as a robustness check, three with only observations from the three states with the largest number of transactions in the collapsed panel (California, Colorado, and Texas). In these latter models, as shown in columns (4)–(6), the sample size shrinks from 66 to 53. In columns (1) and (4), we include state fixed effects (FEs) to control flexibly and comprehensively for unobservable, non-time-

[8] Water supply, water use, and irrigation water use are generated by the University of New Hampshire's Water Balance Model at the grid cell level and at daily time steps, provided here as state-level (and basin-level for the Mojave Basin models in Section 4) annual aggregates, which we average over 2013–2099 to obtain a time-invariant measure. In cases where water supply is lower than water use, groundwater mining is used to fulfill the water use requirement.

[9] This survey was formerly the Farm and Ranch Irrigation Survey. Data are reported every five years. The Durbin's h-test statistic of no autocorrelation in AR(1) models was not rejected in all cases except Colorado.

Table 3
Asset Pricing Model Results for the Nine-State Sample (Dependent Variable: Log Transfer Price)

	(1)	(2) (9 States)	(3)	(4)	(5) (3 States)	(6)
Log lease price	0.154*	0.172**	0.171***	0.146	0.146**	0.153**
	(0.070)	(0.076)	(0.064)	(0.071)	(0.072)	(0.063)
Growth rate: acres irrigated	—	1.154*	1.039*	—	–1.650***	–2.395***
		(0.631)	(0.617)		(0.061)	(0.141)
Risk premium: irrigation vulnerability index	—	0.046***	0.052***	—	1.476***	1.349***
		(0.013)	(0.012)		(0.069)	(0.092)
Real interest rate	–20.061**	–20.164***	–9.403	–22.767	–22.767***	–11.437
	(7.320)	(7.514)	(6.571)	(8.306)	(8.477)	(7.851)
Time trend	—	—	0.070***	—	—	0.068***
			(0.013)			(0.017)
State controls	FE	RE	RE	FE	RE	RE
N (obs.)	66	66	66	53	53	53
R^2	0.212	0.053	0.106	0.269	0.724	0.788

Note: All models include a constant. Robustness checks using a different proxy (major irrigated crop prices instead of acres-irrigated) for the growth rate variable are reported in Appendix Table A3.
*** significance at 1%, ** significance at 5%, *significance at 10%; values in parentheses are robust standard errors clustered by state.

varying state water market characteristics. The use of state FEs precludes identifying coefficients for the risk premium and growth rate variables, neither of which vary over time. Thus, we also estimate a random effects (RE) model, which assumes that the variation across water rights markets (states) is random and uncorrelated with the explanatory variables in the model. Note that a Hausman test supports the RE estimator, failing to reject that RE is consistent and efficient. Estimates are very similar across FE and RE models for the coefficients that can be identified in both.

The coefficients on the lease price are positive, and the coefficients on the real interest rate are negative in all six models in Table 3, consistent with the basic principles of asset pricing theory. The magnitude of the lease price coefficient is about one-fifth the size of the analogous coefficient estimate in the New Zealand fishing quota markets study (Newell, Papps, and Sanchirico 2007). In the New Zealand fishing quota case, the lease price coefficient suggests that a percent change in the lease price will result in a 76% to 86% change in the sale price. In the U.S. western water markets case, a percent change in the lease price will result in only a 14% to 17% change in the permanent transfer price, suggesting that the connection between lease price and permanent transfer price is weaker in the water markets case. This comparative result is not surprising, given that fishing quota markets in New Zealand may be among the world's most efficient created natural resource asset markets (Grainger and Costello 2011) and may thus represent a "best case" (thus far) for such markets in practice.[10]

The coefficients on the real interest rate, although always negative, are only significant in models without the linear time trend. Not surprisingly, adding the time trend also changes the magnitude of these coefficient estimates.

Recall that our proxy for the risk premium is the long-run IVI. An increase in the index, indicating reduced future risk of water stress and shortage, has a positive effect on the water transfer price in the four models in Table 3 where the coefficient can be identified (the RE models), consistent with the asset pricing model.

The growth variable coefficient in the multistate model is positive and statistically significant, suggesting that, consistent with asset pricing theory, growth in farmland irrigated acres increases water right transfer prices. Unexpectedly, the three-state sample results show a statistically significant

[10] An anonymous referee recommended that we also compare U.S. western water market performance to that of Australian water markets, using the asset pricing model. We discuss this comparison in the Appendix section A3.

but negative coefficient associated with the growth variable. This counterintuitive result is not consistent with asset pricing theory. However, robustness check results using a growth variable proxied by crop prices are very similar, showing a positive but statistically insignificant coefficient in the nine-state sample and a negative statistically significant coefficient in the three-state sample (Appendix Table A3).

One potential threat to identification in Table 3 is that lease prices may be endogenous. That is, although the asset pricing model specifies permanent water transfer prices as a function of one-year lease prices, lease prices could also be determined in part by transfer prices (or farmers may consider both options against the benefit of using water for crops). Thus, we also estimated two-stage least squares (2SLS) models using an instrument for the lease price: the average price specific to year and state, excluding local state observations. In all models, an endogeneity test fails to reject the hypothesis that the lease price variable is exogenous. Thus, we interpret the results in Table 3 as the main results and provide 2SLS results in Appendix Table A1.[11]

Taken together, the models estimated using the multistate data suggest that water market activity in the western United States is generally consistent with asset pricing theory. Given the fairly thin markets in this context, as well as the small dataset we constructed from the *Water Strategist* data, these results are encouraging. However, the difference between the efficiency with which information about water's long-run value is transmitted in short-run prices and that observed in New Zealand fishing quota markets and other natural resource applications also suggests that these markets have significant efficiency improvement potential.

4. Mojave Water Market

Although the results in Table 3 are encouraging, the small sample and the heterogeneity of the markets in our multistate analysis leave the analysis open to criticism that we cannot fully capture this heterogeneity when pooling state markets and with the available data. Thus, we examine the Mojave water rights market within the Mojave River Basin Area's jurisdiction in San Bernardino County, California, to see if more active trading in a more homogeneous market comports more closely with the asset pricing model. The Mojave market is a relatively well-defined groundwater market, where monitoring and verification of water production responsibilities are performed by a watermaster, which also acts as a clearinghouse for trades. We apply the asset pricing model to the Mojave market by reestimating equation [4] with the Mojave data, instead of the multistate data used in Section 3.

Mojave Market Data and Model

Summary statistics for the variables used in the asset pricing model estimation for the Mojave market are presented in Table 2. We obtained water transfer data for the Mojave water rights market from the annual watermaster's water transfer reports posted on the Mojave Water Agency's website (Mojave Water Agency 2019a, 2019b). The dataset consists of groundwater transfers in five subareas (Alto, Baja, Centro, Este, and Oeste).[12] The data comprise price and quantity information for 3,368 transactions (288 permanent transfers and 3,080 one-year leases) between 1995 and 2018.[13] Table 4 summarizes the Mojave market data. As expected, and similar to the multistate case, permanent transfer prices exceed lease prices. As shown in Figure 2, the positive correlation between the transfer and lease prices is stronger in the Mojave market than in the nine-state and three-state samples. That relationship be-

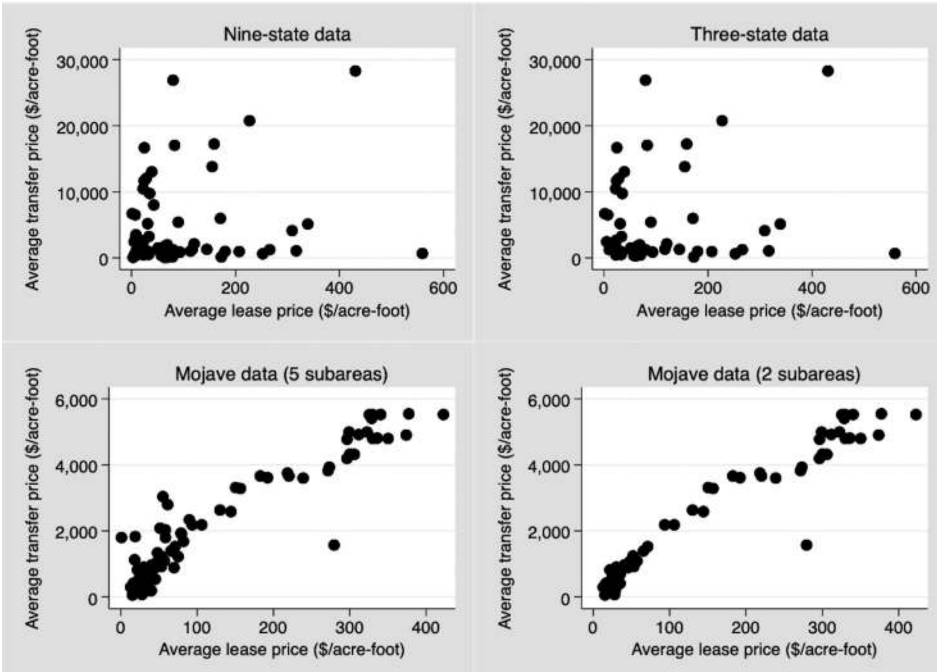
[11] The results of the four models are qualitatively similar to those in Table 3, with one important exception—the lease price coefficients are positive but statistically insignificant in all reported models. We hesitate to conclude much from this exercise, given that our tests suggest that the lease price is not, in fact, endogenous (suggesting that IV is unnecessary) and the challenges to 2SLS with such a small sample. For example, the test statistics for the IV estimators indicate that the models are only weakly identified, resulting in low first-stage F-test statistics.

[12] See https://www.mojavewater.org/files/mbamap_3wm931f0.pdf.

[13] Originally, the dataset included 4,086 observations. We dropped 718 observations with missing price information and when the reported price was 0.

Figure 2

Average (Quarterly) Lease and Transfer Prices in the Multistate Data and the Mojave Data



comes clearer when we consider only the two most active areas (Alto and Baja) in the sample. Price dispersion is noticeable across the subareas. Prices for permanent transfers and leases are highest in the Alto subarea, followed by Oeste and Centro (Table 4). Price differences could be linked to primary water uses in these areas. In the Mojave Basin area, water has five different uses: agricultural, municipal, golf course irrigation, industrial, and recreational. The Mojave dataset does not provide information about water uses at the transaction level. However, the watermaster's Annual Reports (2016) identify major water uses by subarea: Alto—urban, Baja—agricultural, Centro—agricultural and urban, Este—agricultural, Oeste—agricultural and urban. Thus, we can say that water rights prices are highest in the three areas using relatively more urban water, probably reflecting higher-valued uses. The Mojave dataset contains 89 observations (after converting transfer and lease prices to quarterly averages, as we did in the multistate case), from the five subareas between 1996 and 2018.

There are two differences between the variables in equation [4] for the multistate asset pricing model and for the Mojave asset pricing model. First, index j represents a subarea in the Mojave model, instead of a state. Second, the constant growth rate (g) in the Mojave model is proxied by the urban and agricultural water consumption growth rate, rather than the growth of acres irrigated. We use urban water consumption data for the Alto, Centro, and Oeste areas and agricultural water consumption data for Baja and Este, making g somewhat more asset-specific than we were able to do in the multistate context (where g varied only by state, because we could not identify the actual location of the transaction at a finer spatial scale). Water consumption data were obtained from the annual watermaster's reports posted on the Mojave Water Agency website, and they vary by subarea-year, beginning in 2000. Following Newell, Papps, and Sanchirico (2007), as we did in the multistate case, the growth rate variable is estimated using an AR(1) model (where we include the natural log of water consumption,

Table 4

Annual Average Transfers, Quantities, and Prices by Subarea and for the Total Mojave Sample

Subarea	Number of Leases	Number of Transfers	Qty. Leased (AF)	Qty. Transferred (AF)	Lease Price ($/AF)	Transfer Price ($/AF)
Alto	86.1	7.3	224	199	174	2,908
Baja	23.8	3.3	119	115	25	349
Centro	6.4	3.7	226	331	43	1,263
Este	9.0	1.7	47	199	36	494
Oeste	3.6	1.2	370	704	67	2,486
Total	128	12.0	197	219	137	1,710

Note: All statistics are yearly averages. The total five-subarea market statistics are yearly averages without subarea weights.

Table 5

Asset Pricing Model Results for the Mojave Sample (Dependent Variable: Log Transfer Price)

	(1)	(2) (5 Areas)	(3)	(4)	(5) (2 Areas)	(6)
Log lease price	0.413	0.584**	0.179	0.736*	0.736***	0.401
	(0.285)	(0.239)	(0.216)	(0.115)	(0.116)	(0.313)
Growth rate: water consumption	—	1.385***	1.750***	—	2.116***	2.674***
		(0.359)	(0.223)		(0.589)	(0.736)
Risk premium: irrigation vulnerability index	—	–0.026	–0.024	—	—	—
		(0.024)	(0.015)			
Real interest rate	–14.096	–9.859	–2.441	–3.724	–3.724	–2.290
	(8.480)	(8.504)	(3.725)	(7.131)	(7.185)	(3.436)
Time trend	—	—	0.043**	—	—	0.025
			(0.018)			(0.018)
Subarea controls	FE	RE	RE	FE	RE	RE
N (obs.)	89	89	89	69	69	69
R^2	0.480	0.717	0.799	0.756	0.903	0.914

Note: All models include a constant. The proxy for the growth rate variable is historical urban water consumption in the case of Alto, Centro, and Oeste and agricultural water consumption in the case of Baja and Este.

*** significance at 1%, ** significance at 5%, *significance at 10%; values in parentheses are robust standard errors, clustered by state.

year, and a constant) for each subarea.[14] The estimated value represents the likely future growth in demand for water, assuming that the historical drivers of growth will continue into the future.

Results for the Mojave Market

Results for the Mojave water rights market case are reported in Table 5. We estimate the same RE and FE models as we did before, starting with the full five-subarea Mojave sample in columns (1)–(3) of Table 5. The results mostly yield the expected coefficient signs: permanent transfer prices are positively correlated with lease prices and the growth rate and negatively correlated with real interest rates. The magnitude of the lease price coefficient is somewhat higher (0.18–0.58) than in the nine-state sample (0.14–0.17), suggesting a greater influence of lease prices on transfer prices. Unexpectedly, the risk premium coefficient is negative and statistically insignificant, which might be affected by the relatively smaller area that the Mojave Basin represents and, as a result, less variation. Also, urban water use is dominant in some subareas of the region, which may not be as well represented by the IVI.

We estimated the same model on a sample comprising only the two most active subareas in the Mojave region: Alto and Baja. The majority of the quarterly observations in the sam-

[14]The Durbin's h-test statistic of no autocorrelation was not rejected. The values for water consumption by subarea were not reported until 2000; thus, our proxy is estimated using data from 2000–2018, but the constant growth variable is applied to the entire dataset, 1996–2018.

ple (69 out of 89 area-quarters) are associated with these two areas. The results (shown in columns 4–6 of Table 5) imply that the water market consisting of the Alto and Baja subareas of the Mojave exhibits the greatest efficiency among all of the markets we examine, as evident from the large significant coefficient on the lease price (model [5]). In the Alto and Baja subareas of the Mojave market, a 1% change in the one-year lease price will result in a 74% change in the permanent water rights price. (The signs for the remaining estimated coefficients are also consistent with asset pricing theory.) As in the multistate asset pricing model, there is a potential concern about endogenous lease prices, but endogeneity tests suggest that the lease price is exogenous; for completeness, IV results are reported in Appendix Table A2, using the same instruments as in the multistate case (average lease prices by region-year, excluding local observations). Note that we cannot identify the risk premium (IVI) coefficient in the two-area models due to insufficient variation.

The results for the Mojave market, which is known to have fewer barriers to trade in comparison with the multistate markets, suggest that there is significant potential for efficiency improvements in other water markets if barriers to trade were addressed.

5. Conclusions

We examine the degree to which U.S. western water market prices in nine states act as asset pricing theory would predict. Findings suggest that water market transactions do generally comport with the asset pricing model; for example, permanent water transfer prices are positively correlated with one-year lease prices and negatively correlated with the real interest rate. However, the smaller coefficients associated with lease prices in the water markets compared with fishing quota markets suggest significant potential for market efficiency improvements in the water market case.

We find that water market efficiency is highest in one of the most active U.S. water rights markets located in the Mojave Basin area—markets that are known to have lower barriers to trade. The coefficients on water lease prices are higher in the Mojave markets than in other water markets and, in the case of the two most active areas of the market, the coefficient on lease prices is almost as high as those in New Zealand fishing quota markets. This difference in results suggests that there is significant potential for efficiency improvements in water rights markets in the western United States, which could lead to higher welfare gains from reallocating water.

Taken together, the results provide reason for optimism about water rights markets in the western United States. Comparing water rights transfers in the United States with other natural resource markets allows us to better understand the efficiency potential in the increasingly common markets for created natural resource assets. It is likely that a significant portion of the differences in the relationship between short-term and permanent transfers across different water rights markets (as well as other natural resource markets) can be attributed to the institutions that define and govern these markets. Poor governance across many water markets in the western United States is often evident in the lack of accountability, monitoring, and enforcement. Most surface water and some groundwater rights in the western United States are governed by the prior appropriation doctrine, where water rights are allocated based on seniority (“first in time, first in right”) leading to greater market frictions. This allocation rule tends to be less significant for short-term transfers, which also tend to be associated with lower transaction costs because of a simpler administrative process. Some regional water markets in the western United States are well-functioning. The market in the Mojave River Basin is one of them, which is supported by our findings. Transactions in this market are well recorded and managed, and rights are not allocated based on “first in time, first in right,” allowing for faster and less costly transfers.

The ability to quantify the efficiency of water rights transfers as a whole provides an opportunity for measuring progress in market development, learning from better-functioning markets, and, as a result, advancing policies to reduce barriers to water trading. More analysis of the price relationship between short- and long-term water transfers is necessary; however, more available data are needed to advance our empirical assessment. Our analysis of the western U.S. region was limited by the dataset ending in 2010. Since then, various water management efforts have been progressing that are likely to affect water accessibility and management to different water stakeholders in the near future. Examples include so-called smart market development (Young and Brozović 2016); an attempt to redefine the seniority rule based on the Australian example in Diamond Valley, Nevada (Young 2015; Wheeler et al. 2017; Zeff et al. 2019); the existence of local informal water transfers (Young and Brozović 2019); and an increase in groundwater protection efforts (e.g., Sustainable Groundwater Management Act in California; Babbitt et al. 2017). Further work examining the multilayered water rights structures, policies, and regulations that support efficient resource reallocation could make a valuable contribution to enhancing water markets' capacity to mitigate anticipated future reductions or increased variation in water supply in the U.S. West and elsewhere.

Acknowledgments

This work was supported by the U.S. Department of Energy, Office of Science, Biological and Environmental Research Program, Earth and Environmental Systems Modeling, MultiSector Dynamics, Contract No. DE-SC0016162.

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