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Transitions from irrigated to dryland agriculture in the Ogallala Aquifer: Land use suitability and regional economic impacts

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A R T I C L E I N F O

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A B S T R A C T

Many agricultural communities depend on groundwater irrigation as a supplemental or primary water source. However, groundwater resources are finite, and depletion can make continued irrigation inviable. When modeling the economic impacts of future aquifer decline, studies often assume that irrigated cropland will transition uniformly to dryland crop production. In reality, irrigation has allowed crops to be grown across a wider range of soil and climate conditions than can support dryland crop production. Here, we test the agronomic and economic importance of this assumption by mapping the spatiotemporal distribution of anticipated future irrigation losses across the Ogallala or High Plains Aquifer (USA) at annual, 30 m resolution. We then develop a land use suitability model to determine whether these lands would transition to dryland agriculture or pasture use. We find that 22,000 km^2 (24 %) of currently irrigated lands in the High Plains Aquifer may be unable to support irrigated agriculture by 2100, and 13 % of these areas are not suitable for dryland crop production due primarily to low quality soils. To quantify the farm-scale and regional-scale economic importance of land use suitability, we selected six case study counties across the aquifer and modeled farm and community-scale economic outcomes (gross revenue and value added, respectively) with and without consideration of land use suitability. We find that not accounting for land use suitability leads to an overestimate of economic benefits in transitioned land by 12–45 %, with variability across counties primarily driven by the distribution of soil capability, dryland crop mix, and local economic factors. Notably, this implies that the economic impacts of land transitions are not directly proportional to area lost but rather mediated by underlying variability in these three factors. Our analyses highlight the importance of considering local biophysical constraints in planning for future land use trajectories. Community and regional land use planning needs to incorporate the possibility that irrigated cropland may transition to non-irrigated pasture production rather than dryland crop production, which can have substantial biophysical and economic impacts.

1. Introduction

Groundwater resources are facing multiple pressures including increasing food demand, expanding urban areas, and changing climates. Agricultural production and communities are disproportionately impacted by increased competition for groundwater, which represents 42 % of the water used for agriculture globally and supports 60 % of U.S. irrigated agricultural production (Döll et al., 2012; Siebert et al., 2010). While improvements in water use efficiency and conservation can reduce agriculture's groundwater consumption, some groundwater resources are non-renewable on any meaningful timeline for human use due to biophysical constraints that limit recharge. Regions with declining groundwater sources, such as large portions of the Ogallala Aquifer region, will need viable paths for transitioning away from groundwater dependence that cause the least disruption to residents and the local economy.

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The Ogallala Aquifer is the largest unit of the hydrologically connected High Plains Aquifer system (hereafter referred to as the HPA), the largest freshwater aquifer in the world that underlies 450,660 km^2 across eight states (Thelin and Heimes, 1987). It provides the main source of agricultural and public water supplies that has sustained economic development in the region for more than 80 years. In the early 20th century, conversion of native grasslands to annual, non-irrigated crop production and prolonged drought led to the Dust Bowl of the 1930s (McLeman et al., 2014). The adoption of irrigation and soil conservation methods transformed agriculture and expanded the region's economy while reducing soil erosion.

However, the HPA is an exhaustible resource. While some regions, particularly in the Northern High Plains, can achieve sustainable groundwater pumping levels while still supporting irrigated agriculture through conservation efforts, regions with low recharge rates and high evaporative demands, common in the Southern High Plains and the western portions of the Central and Northern High Plains regions, are likely to see declining groundwater levels even with the best conservation practices (McGuire, 2017; Scanlon et al., 2012). Generally, irrigated crop production is assumed to be impractical once aquifer saturated thickness drops below 9 m (~30 ft) due to low well yield (Rawling and Rinehart, 2018; Schloss et al., 2002). Under status quo management and climate, the proportion of the aquifer below this threshold is predicted to grow from 25 % to 40 % between 2012 and 2100 (Haacker et al., 2016).

There are several forces that could affect this timeline to depletion, including climate change and producer adaptation. Historical analysis of land-use variations in the U.S. Great Plains shows that many land-use systems continuously adapt to climate and biophysical changes in response to socioeconomic drivers, land-use legacies, and regional land-use traditions (Drummond et al., 2012). It is likely that producers will adapt by shifting crop choice and irrigation practices to slow the rate of decline and extend aquifer life (Haacker et al., 2019; Manning et al., 2017; Schuck et al., 2005) through both self-organization (Butler et al., 2018; Deines et al., 2019a) and top-down restrictions (Hrozencik et al., 2017). Climate change, in contrast, will likely shorten the timeline due to warming temperatures, increased crop evaporative losses, and increased frequency and duration of prolonged droughts that will increase the need for groundwater resources to support crop production (Cotterman et al., 2018). While the time to depletion is challenging to predict precisely, the spatial distribution of regions where depletion will first occur is more certain due to observable variation in saturated thickness and historic depletion trajectories. Thus, it is more a question of when than whether communities will need to develop strategies to sustain their livelihoods without reliance on groundwater for irrigated crop production. For these communities, an integrated assessment is needed to evaluate potential future land management options and the associated regional economic impacts.

When modeling future groundwater use and aquifer decline, studies often assume that irrigated cropland will transition to dryland crop production (e.g., Amosson et al., 2009; Cotterman et al., 2018; Dobrowolski and Engle, 2016; Wheeler et al., 2008). However, irrigation has allowed crops to be grown across a wider range of soil and climate conditions than can support dryland crop production. Recent evidence suggests that an appreciable proportion of irrigated cropland may transition to pasture, rather than dryland crop production, due to biophysical limiting factors such as soil quality (Golden and Guerrero, 2017; Golden and Johnson, 2013). However, the extent to which these factors affect areas projected to be depleted, and the associated economic impacts of these limits, remains unknown due to a lack of spatially explicit data on the areas in question.

Here, we evaluated the degree to which projected economic outcomes associated with transitioning out of irrigated agriculture change when land use suitability is incorporated into the accounting process. To quantify the economic impacts of irrigation curtailment, we first identified currently irrigated land that would no longer support irrigation through 2100 due to groundwater depletion, including the estimated year in which irrigation would become nonviable. We then developed a land use suitability model to predict the appropriate non-irrigated land use for these areas based on the soil properties of current pasture and rainfed cropland. Using a case-study approach, we estimated the potential economic impact of these different land use practices on producer profitability and regional economics for a high-priority county in each of six states with appreciable irrigated area sourced from the aquifer. In the 21st century, reduced well outputs coupled with prolonged drought events have already led to dust storms in the Southern High Plains region reminiscent of the Dust Bowl (Gaskill, 2012). Our analysis can inform strategic planning to sustain rural communities and avoid a second Dust Bowl in this ecologically vulnerable region.

2. Methods

2.1. Quantifying annual losses in irrigated land through 2100

We used recently available gridded datasets of groundwater depletion and irrigated areas to estimate future irrigation losses by year at 30 m resolution across the aquifer. Haacker et al. (2016) projected the year of groundwater depletion for the full HPA by linearly extrapolating annual decline rates between 1993–2012 through year 2300 at 250 m resolution. Haacker et al. (2016) defined the year of functional depletion for each grid cell as the year in which the extrapolated saturated thickness fell below a viability threshold of 9 m. This represented a minimum saturated thickness required for high capacity wells based on discussion with state agencies and irrigators in Kansas (Hecox et al., 2002; Schloss et al., 2002). These depletion projections indicate the area unable to support irrigation may increase from 25 % to 40 % between 2012 and 2100 (Fig. 1a; Haacker et al., 2016). This linear extrapolation approach represents a "business-as-usual" scenario and does not account for dynamic management responses to water decline, behavior changes induced by increased pumping costs or reduced well yields, new technologies that may increase water use efficiency, or changes in climate. Additionally, the 9 m saturated thickness threshold may not be a uniform predictor of groundwater irrigation feasibility given heterogeneity in other aquifer properties (e.g., specific yield). To check its reasonableness, we summarized the distribution of saturated thicknesses in 2016 underlying actively irrigated lands between 2015–2017 (see below). We found that although the vast majority of irrigated area occurs above this threshold, approximately 8 %, 14 %, and 24 % of irrigated area in the Northern, Central, and Southern High Plains, respectively, occurs below this 9 m threshold (Fig. S1). While the exact date of depletion will likely differ from the Haacker et al. (2016) predictions, our analysis still focuses on the portions of the HPA which are likely to be depleted first given the current saturated thickness and rates of decline, and therefore provides an estimate of the potential magnitude of the biophysical and economic impacts of transitions to pasture rather than rainfed cropland.

We identified currently irrigated land using recently published maps of annual irrigation over the HPA derived from Landsat satellite data at 30 m resolution (the AIM-HPA dataset, Deines et al., 2019b). This dataset spans 1984–2017, has an accuracy of 91.4 % compared to over 17,000 ground truth point locations, and shows strong agreement with county statistics from the 5-yr US Agricultural Census (r^2 = 0.86). Here, we used these maps to generate a new layer of currently active irrigation, defined as any area classified as irrigated between 2015–2017 (Fig. 1b). We used this 3-year span to capture fields which may have been in a fallow rotation or only partially irrigated in any single year.

We then used the depletion map to identify irrigated pixels likely to transition out of irrigated production by 2100 due to groundwater depletion and assigned these pixels a curtailment year based on the year in which saturated thickness was projected to drop below 9 m. We

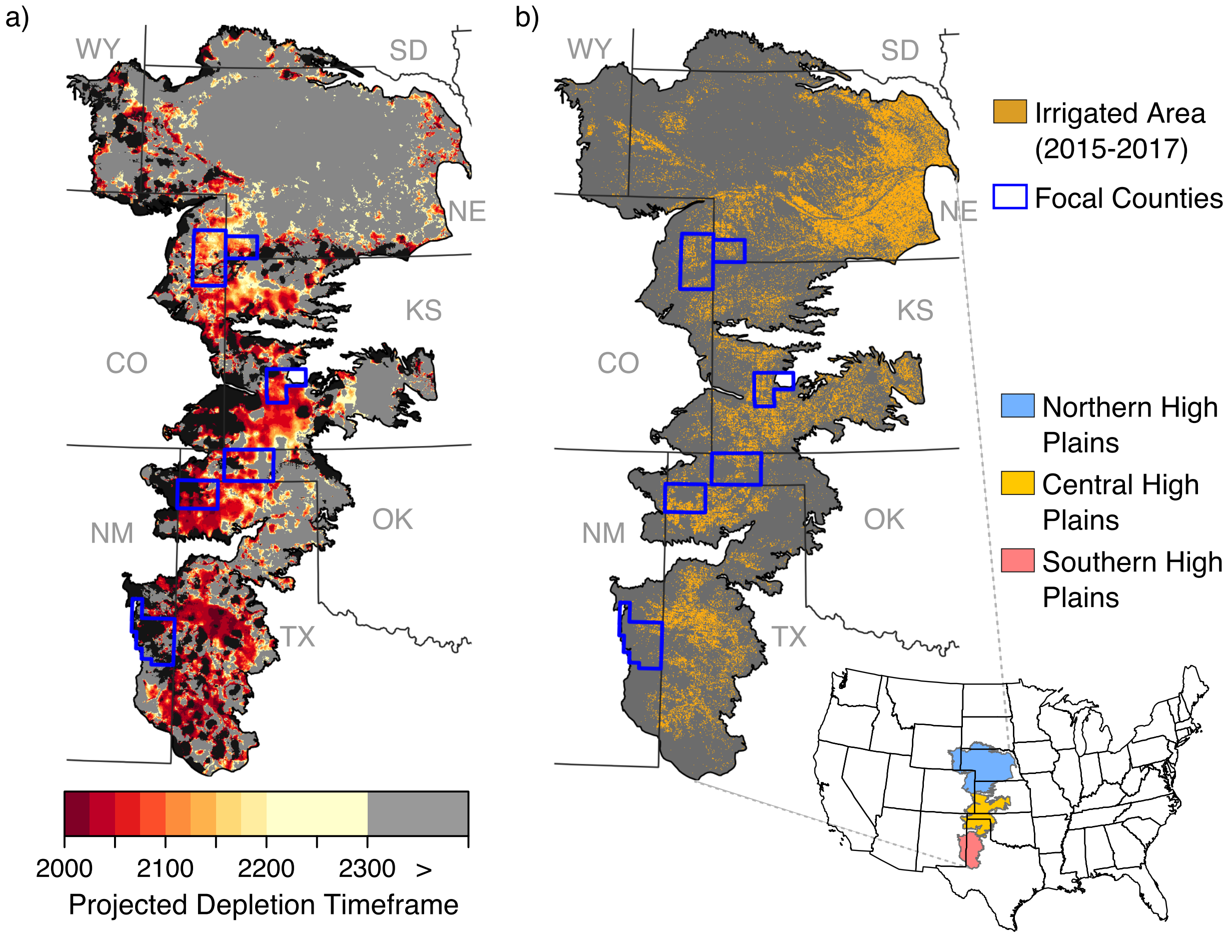

Fig. 1. Aquifer depletion and current irrigation extent in the High Plains Aquifer study area. a) Projection year of effective aquifer depletion where groundwater pumping likely becomes uneconomical (saturated thickness < 9 m). Black regions represent areas historically below this threshold. Data from Haacker et al., 2016; figure modified from Deines et al., 2019b. b) Actively irrigated areas across the aquifer. Active irrigation is defined as all 30 m pixels classified as irrigated between 2015–2017 from Deines et al., 2019b. The six counties highlighted in this analysis are outlined in blue. Inset: shaded regions denote the three major aquifer subregions. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

assumed that all irrigation overlying depletion zones is sourced from groundwater, not surface water. To test the validity of this assumption, we compared the percent of irrigated area projected to be lost by 2100 to the percentage of irrigation which is sourced from groundwater (Dieter et al., 2018) by county. We found that all counties have a higher proportion of groundwater irrigation than the proportion of land expected to be depleted by 2100 (Fig. S2), supporting our assumption that transitioned lands were sourced from groundwater.

The resulting map of projected year of irrigation curtailment is unique in that it explicitly locates the year and location of irrigation curtailment, allowing us to assess non-irrigated land use alternatives based on location-specific soil properties and to estimate economic impacts by year through 2100.

2.2. Land use suitability analysis

To determine whether depleted areas would transition into pasture or dryland agriculture, we developed a land use suitability model to predict the appropriate non-irrigated land use based on the soil properties of current pasture and dryland crops. First, we used our map of current 2015–2017 irrigation to isolate non-irrigated cropland and pasture from the USDA Cropland Data Layer (CDL) land use maps (USDA National Agricultural Statistics Service, 2018). To do so, we used the 2017 CDL cultivated layer (Boryan et al., 2012), which identifies all cultivated land from 2013 to 2017, to comprehensively locate cultivated and non-cultivated land areas. We then used the 2017 CDL to retain only pasture non-cultivated land cover, removing forests, wetlands, water, and urban areas. Any land enrolled in the Conservation Reserve Program in 2015 (as identified by a spatial data layer obtained from the USDA National Resources Conservation Service through a Freedom of Information Act request) was removed to avoid confounding the analysis with land suitable for cropland but currently under conservation status and possibly classified as grassland in the CDL.

From this resulting map of non-irrigated cropland and pasture, we randomly generated 40,000 sampling points across the aquifer. At each point, we extracted the latitude, longitude, and field slope calculated from a digital elevation model (U.S. Geological Survey, 2017), and non-irrigated land capability class and subclass values from the gSSURGO soil database (soil capability for brevity) (USDA Natural Resources Conservation Service, 2017). The soil capability scale ranges from 1 (best for agriculture) to 8 (worst for agriculture). A soil capability rating of 5 is defined as soils which “have little or no hazard of erosion but have other limitations…that limit their use mainly to pasture, range, forestland, or wildlife food and cover”. Soil classes are also subdivided into four subclasses based on climate, erosion, root zone soil limitations, and excess water. Based on preliminary analyses using the 2017 CDL and our irrigation layer, 93 % of current dryland agriculture in the study area occurs where the soil capability < 5, but 31 % of pastureland also occurs below this threshold (Figs. S3 and S4).

We then randomly split the sampled points into training and validation subsets (~50 % to each group) and trained a random forest classifier to create a binary prediction for dryland crops vs pasture based on these soil classes, field slope, and spatial location (latitude and longitude). Random forests are ensemble classifiers that generate predictions based on the consensus of multiple decision trees, each parameterized with a random subset of variables and training samples. We selected this classifier because they are demonstrably fast, accurate, robust to variable collinearity and overfitting, and straightforward to implement (Belgiu and Drăgu, 2016). The resulting overall classification accuracy evaluated on the validation data points was 82 %, with 85 % and 78 % of test points correctly classified for dryland crops and pasture, respectively. The full accuracy table is presented in Table S1. Soil capability class, slope, and latitude ranked highest among variable importance metrics measuring contributions to classification accuracy (Fig. S5).

We then applied this land use suitability model to the full aquifer at 30 m resolution to obtain suitable land uses for areas predicted to transition from irrigated agriculture. Finally, we tallied the total area of transitions from irrigation to each of dryland cropping and pasture classes by county for 157 counties fully covered by the AIM-HPA irrigation map product, which was produced with a 15 km buffer around the aquifer.

2.3. County scale economic analysis

To demonstrate the importance of accounting for land use suitability when planning for transitions away from irrigated cropland, we selected one county from each of six Ogallala states with appreciable aquifer-fed irrigation for the economic analyses (Fig. 1, Table 1). Counties within each state were ranked by the percentage of current irrigated area projected to transition to pasture. We then selected the county with the highest percent transitioning to pasture, with the exception of Texas, where the 2nd ranked county (Dallam County) was selected due to the availability of economic data. Therefore, our case study examples were developed to demonstrate the largest impacts within each state and provide an upper bound for the error associated with the common assumption that all irrigated land will transition to non-irrigated crop production when groundwater supplies are depleted.

Economic and land use suitability models were linked through time to estimate temporal land use changes and corresponding economic impacts in response to changes in groundwater availability. Two scenarios were evaluated, referred to as the 'Simple Scenario' and the 'Land Use Suitability Scenario'. The Simple Scenario assumed that 100 % of land transitioning out of irrigation will shift to dryland crop production regardless of land use suitability. This assumption is consistent with the dominant approach to date (e.g., Amosson et al., 2009; Cotterman et al., 2018; Dobrowolski and Engle, 2016; Wheeler et al., 2008). The Land Use Suitability Scenario uses the land use suitability model described in Section 2.2 to partition transitioning areas in to either dryland crop production or dryland pasture production based on underlying soil and physiographic properties. Thus, the differences between the two scenarios represent the economic impact typically overlooked by studies that do not take land use suitability into account and assume all land will transition to dryland crop production. By focusing on the difference in scenario outcomes, our approach buffers against the effects of our simplifying assumptions regarding groundwater hydrology and change in irrigated crop revenues over time, since our results are based on the difference in scenarios (both of which use the same underlying assumptions). Thus, this research only considers the economics associated with the land area transitioning to non-irrigated production.

We consider two economic metrics for emerging dryland crop and pasture areas due to aquifer depletion: the gross revenue and the community economic impact ("value added"; see below). In order to generate the economic metrics, we assumed that the future proportions of dryland crop types would remain similar to the present-day crop mix. To quantify this, we found the average area for each crop type by county between 2013–2017 based on the areas of annual CDL crop layers not classified as irrigated in the AIM-HPA irrigation maps. Table 1 provides the average percentage of major dryland crops by county, as well as the percent of total dryland acreage these major crops cover. While there are other non-irrigated crops grown in these counties, the major crops account for between 95.2 % and 98.3 % of non-irrigated cropland. We then estimated weighted average gross revenues per hectare based on county specific crop production budgets. As an example, the Kansas State University Agricultural Experiment Station and Cooperative Extension Service annually develops Farm Management Guide crop budgets. Since there can be large annual variations in commodity prices and input costs, we used a 5-year average from 2013–2017.

We estimated the gross revenue for pasture based on a previous analysis for Finney County, Kansas (Asem-Hiablie et al., 2015). In this region, pasture is primarily used for cow-calf operations which raise animals for later sale to concentrated feeding operations. Specifically, we assume that each cow-calf unit requires 15.6 ha, that the calf weaning weight is 250 kg, and a 7-year average future price of $3.44 per kg. These data result in an expected gross revenue for pastureland of $54.29 per hectare. The gross revenues for the remaining counties were adjusted based on a ratio of average annual rainfall relative to Finney County, so that precipitation is used as a proxy for non-irrigated pasture productivity. For example, the gross revenue in Dallam County was calculated by multiplying the gross revenue in Finney County by the ratio of average annual rainfall in Dallam County compared to Finney County. It was assumed that these gross revenues would be obtained the first year of conversion.

To evaluate the community economic impacts beyond direct farm revenue of each scenario, we used these gross returns as input to a regional economic impact model, IMpact analysis for PLANning (IMPLAN) (IMPLAN Group, LLC, 2013). We used 'Value added' as the metric of regional economic impact (BBC Research et al., 1996; Thorvaldson and Pritchett, 2006). Value added is calculated by multiplying dryland crop and pasture revenues by a land use-specific regional impact factor (Table 1). Value added consists of four components: 1) employment compensation (wage, salary, and benefits paid by employers), 2) proprietor income (payments received by self-employed individuals as income), 3) other property income (payments to

Table 1
Attributes for the six case study counties. Non-irrigated crop mixes, weighted adjusted gross revenues (revenues), and the weighted average total impact on value added (regional impact factor) for crop and pasture lands in the six case study counties.

County	Non-Irrigated Crop Mixes (%)						Crop Revenue ($/ha)	Pasture Revenue ($/ha)	Regional impact factor (Crops)	Regional impact Factor (Pasture)
	Corn	Cotton	Fallow	Hay	Sorghum	Wheat				
Yuma, CO	14.5	0	39	0	2.4	44.1	$626.52	$51.44	0.58	0.58
Finney, KS	6.7	0	34.2	0	22	37.2	$572.95	$54.29	0.57	0.54
Dundy, NE	38.1	0	27.6	0	5.2	29.1	$797.86	$51.44	0.55	0.58
Roosevelt, NM	2.1	0	21.8	17	22	37	$195.29	$53.31	0.64	0.55
Texas, OK	1.6	0	25.3	0	17.3	55.8	$624.14	$54.29	0.56	0.5
Dallam, TX	6.5	5.2	19.6	0	6.9	61.7	$660.79	$57.49	0.58	0.53

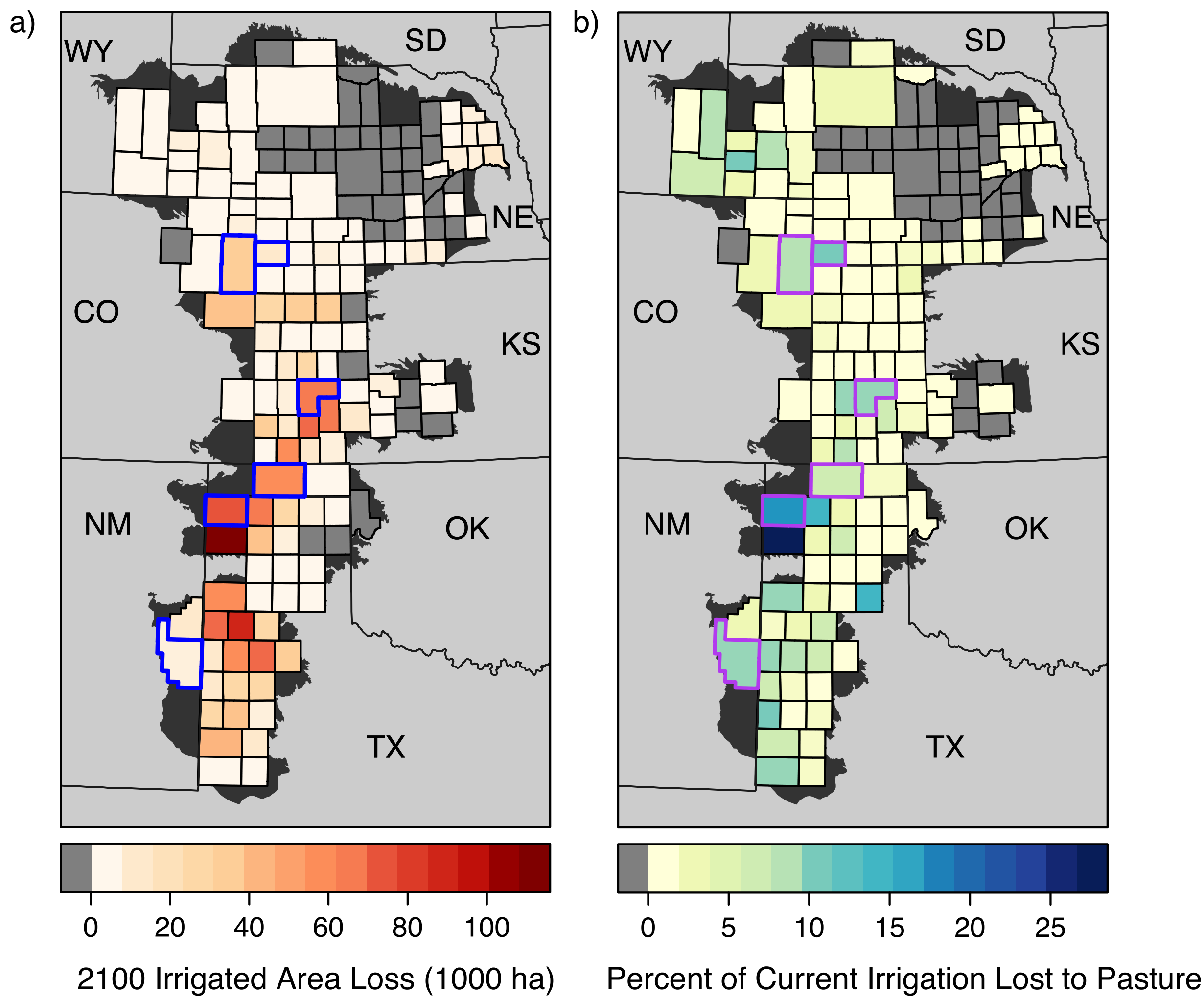

Fig. 2. Projected irrigation declines and suitable dryland land use by count. a) Irrigated area per county projected to be lost due to aquifer depletion by 2100. b) The percentage of current irrigated area projected to lost and suitable for pasture (not suitable for dryland agriculture). Counties selected for the economic analysis are outlined.

individuals in the form of rent), and 4) indirect business taxes (all taxes with the exception of income tax). We adapted work by Golden and Guerrero (2017) that estimated multipliers for non-irrigated crops in southwest Kansas and used IMPLAN to generate pasture land multipliers based on the beef cattle ranching and farming sector (Table 1). Finally, we compared gross revenues and community economic impacts for the Simple Scenario and Land Use Suitability Scenario for each of the six counties through 2100.

3. Results and discussion

3.1. Land use transitions across the High Plains Aquifer

3.1.1. Irrigation losses

Based on maps of projected groundwater depletion and current irrigated area, we estimated that 22,000 km^2 (24 %) of currently irrigated lands in the HPA will be unable to support irrigated agriculture by 2100 (Fig. 1). Fig. 2a shows irrigated area losses aggregated to the county level across the aquifer. The majority of these losses occur in the Central and Southern High Plains, which may lose 9500 and 7500 km^2, respectively. This is equivalent to 40 % of currently irrigated area in the Central High Plains, and 54 % for the Southern High Plains; in contrast, the Northern High Plains is projected to lose only 10 % of currently irrigated area. The more humid north-central and eastern portions of the aquifer will likely experience far less depletion-induced losses due to higher annual rainfall and natural recharge (Scanlon et al., 2012), although it's worth noting that irrigated area continues to expand in these regions (Deines et al., 2019b) and may alter future rates of extraction.

We also found different patterns of irrigation loss through time among the HPA regions. In the Southern High Plains, 80 % of losses occur prior to 2060, midway through our study time period. In contrast, the Central High Plains sees 53 % of losses by this time. In the Northern High Plains, where natural recharge is highest, 56 % of losses occur between 2060–2100, in the latter half of the study period. Our method implicitly also assumes that irrigated crop producers transition to rainfed agriculture only as a result of the lack of available groundwater. However, a profit maximizing producer may make the transition when the net profit of rainfed production exceeds the net profit of irrigated crop production. As a result, this analysis may underestimate the rate of transition, particularly as pumping costs increase with water table decline (Foster et al., 2017).

Our estimates agree fairly well with existing projections of lost irrigated area in parts of the aquifer. Scanlon et al. (2012) estimated that 35 % of the Southern High Plains would lose irrigation within 30 years; we estimate a 32 % loss for 2042. Cotterman et al. (2018) project that 60 % of area irrigated for corn and 50 % of area irrigated for wheat will be lost in the Central High Plains by 2099. Our estimate for all irrigated area in this region is lower (40 %). Although we consider all irrigated crop types, which could account for some discrepancy, it's likely our use of a higher resolution irrigation layer (30 m) compared to the lower 250 m resolution layer they use provides a more refined estimate of current irrigated areas.

3.1.2. Non-irrigated land use suitability

We were able to predict suitable dryland land use with 82 % accuracy using the random forest model trained on soil capability classes, field slope, and location (Table S1). From this land use suitability model, we estimated that 87 % of lost irrigated area through 2100 could support dryland crop production, while the remaining 13 % is better suited to pasture use. This percentage is relatively consistent across aquifer regions (Figs. 1, 2b), with 12 % (860 km^2) of lost area suitable for pasture in the Southern High Plains and 14 % in both the Central High Plains (1400 km^2) and Northern High Plains (680 km^2). These areas are also distributed fairly uniformly across depletion timeframes, with similar percentages suitable for pasture for areas lost before and after 2060 (14 % and 13 %, respectively).

The land capability classification system that informed this analysis was primarily developed as a response to the Dust Bowl of the 1930s to prioritize conservation efforts on soils with high erosion potential (Helms, 1992). Thus, our results suggest that a substantial proportion (13 %) of lands facing groundwater depletion across the entire aquifer region may again be priority areas for targeted conservation programs to avoid wind erosion events. Forethought will be particularly important to support the transition. Highly erodible lands can be stabilized by the establishment of perennial vegetation due to their extensive root systems that can improve soil structure, water infiltration, and increase soil organic matter levels (Durán Zuazo and Rodríguez Pleguezuelo, 2008). However, establishing slower-growing perennial pastures can be difficult on erodible lands in the absence of irrigation water (Porensky et al., 2014). These soils tend to have low water holding capacity and erosion events can readily kill vulnerable seedlings. Hence, the quality of the soil will likely dictate the future land use options and these differing options may require proactive transitions to establish perennial vegetation prior to the loss of irrigation.

3.2. County case studies

3.2.1. County land use transitions

While the land area more suitable for pasture under non-irrigated management is evenly distributed through depletion timeframes and within larger HPA regions, some counties will be impacted more than others (Fig. 2b). Here, we selected one county from each of six states overlying the HPA to examine land use transitions in detail and estimate the economic impacts of failing to account for land use suitability when planning for irrigation curtailment. Fig. 3 provides the annual changes in depleted irrigated area suitable for dryland cropping and pasture uses for each case study county. We estimate that Dallam County, TX, Finney County, KS, and Texas County, OK, will lose 65 %, 57 %, and 55 % of their currently irrigated cropland through 2100 due to groundwater depletion, with smaller losses of 30 % in Dundy County, NE, and Roosevelt County, NM, and 27 % in Yuma County, CO (Fig. 3). In terms of magnitude of area lost, Dundy and Roosevelt counties will lose only $130-150$ km^2, while Texas and Finney counties will both lose over 500 km^2 and Dallam County is projected to lose over 760 km^2.

Based on the land use suitability model, the percent of current irrigation projected to become pasture (rather than dryland crops) for these counties ranged from 6.3 % in Texas County, OK, to 19 % in Dallam County, TX (Fig. 2b). Looking specifically at areas facing groundwater depletion, we estimated that at least one-third of lost irrigated area through 2100 is more suitable for pasture than crop production systems in Dundy County, NE (42 %, 66 km^2), Yuma County, CO (34 %, 120 km^2), and Roosevelt County, NM (33 %, 46 km^2). Smaller percentages were more suitable for pasturelands in Dallam County, TX (28 %, 220 km^2), Finney County, KS (18 %, 110 km^2), and Texas County, OK (11 %, 62 km^2). These future anticipated reversions to non-irrigated pasture may partially counteract recent cropland expansion, which was preferentially concentrated on former grassland and less suitable soils (Lark et al., 2015).

3.2.2. The influence of land use suitability on economic impacts

We estimated that gross revenues for dryland crop production are

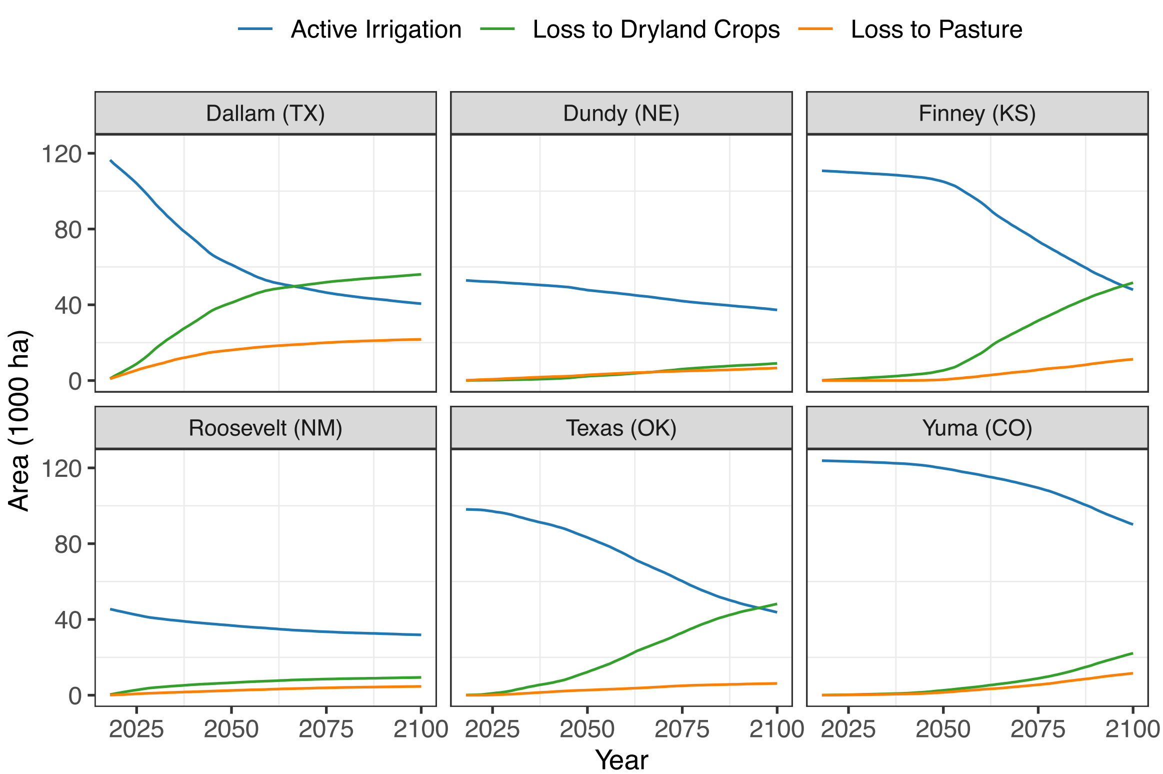

Fig. 3. Land use suitability projections for area transitioning out of irrigation in the focal counties.

Table 2
Cumulative gross revenue from 2019 to 2100 for the six case study counties. Gross revenues for the Simple Scenario (all transitioning land becomes dryland crops) and the Soil Suitability Scenario (transitioning land can be either dryland crops or pasture depending on soil).

State	County	Simple Scenaro	Soil Suitability Scenario	Difference ($)	Difference (%)
Colorado	Yuma	$547,564,295	$362,547,099	$185,017,197	33.8
Kansas	Finney	$1,086,610,526	$935,337,299	$151,273,227	13.9
Nebraska	Dundy	$488,820,290	$268,002,433	$220,817,857	45.2
New Mexico	Roosevelt	$154,946,109	$121,659,738	$33,286,371	21.5
Oklahoma	Texas	$1,279,628,015	$1,123,041,191	$156,586,823	12.2
Texas	Dallam	$3,031,785,080	$2,243,459,122	$788,325,958	26.0

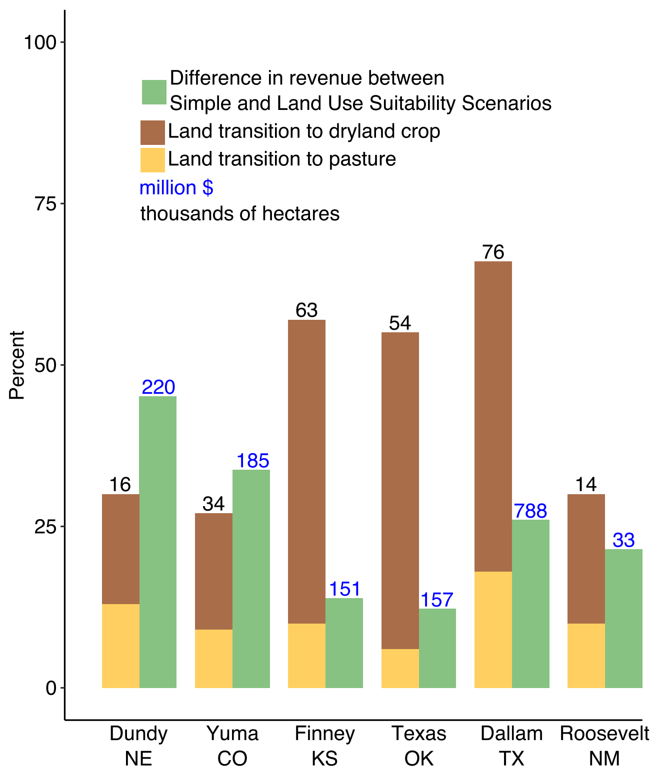

Fig. 4. Summary of projected land use transitions and the economic implications of including land use suitability for the six case study counties. For each focal county, stacked bars show the percent of land transitioning to dryland crop (brown) vs. pasture (yellow) when land use suitability is taken into account during modeling (Land Use Suitability Scenario), as opposed to assuming that all land transitions to dryland crop (Simple Scenario). Green bars represent the relative difference in cumulative gross revenue between the two scenarios for each county by 2100. Blue numbers indicate monetary difference (in million $), and black numbers indicate the total area no longer able to support irrigation by 2100 in each county (thousand ha). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

10.8 times greater than for pasture on average across the 6 case study counties, illustrating why previous analyses have assumed that transitioned land will be converted to dryland crops rather than pasture. The revenue difference for individual counties ranging from 3.7–15.5. This spread arises from the considerable variability in dryland crop revenues among counties, ranging from $195/ha in Roosevelt County, NM, to $798/ha in Dundy County, NE (Table 1). This is likely due to rainfall and temperature gradients across the aquifer that influence dryland crop mix as well as local economic factors. Pasture revenues were more consistent across counties (Table 1), in part due to our assumption of a consistent animal density across counties.

By comparing the Simple Scenario and the Land Use Suitability Scenario (Section 2.3), we were able to quantify the economic implications of ignoring soil limits on future land uses in depleted areas. We found that ignoring land use suitability considerations in economic models of future agricultural revenues would overestimate cumulative gross revenue from transitioning land by 23.3 % across all six counties by 2100, a difference of $1.54 billion for these six counties alone. Across individual counties, overestimations ranged from 12 to 45% (Table 2, Fig. 4). These estimates are likely conservative because they predict revenue generation in the first year following a transition to pasture, but it can take several years before newly established perennial pastures in a dryland environment can support grazing and achieve peak productivity (Cook et al., 2012).

Because our estimates of the regional impact factors for dryland crops and pastureland spanned a limited range across our focal counties (0.5–0.64; Table 1), the community economic impacts as measured by value added (see Section 2.3) were similar in range to the gross revenues as a percent difference between scenarios (Table 3). Averaged across the six counties, we found a cumulative difference of 23.5 % through 2100 between scenarios, or an absolute difference of $887 million. Thus, the economic impact of land transitions will ripple through rural economies with implications for sustaining livelihoods beyond the farm. Due to the large differences in gross revenues and smaller differences in regional impact factors between dryland crop and pasture lands, the gross revenue differences will be the primary driver of community economic impacts. Consequently, taking land use suitability into consideration suggests that communities with more land transitioning to pastures will face a more difficult economic transition away from irrigation.

In all counties, annual differences between scenarios increased with time, though with different temporal signatures which are reflective of the interplay between irrigation depletion and land use suitability (Fig. 5). For instance, in Yuma and Finney counties, the Simple and Land Use Suitability Scenarios are identical for the next several decades before diverging in the 2030s (Yuma County) and 2050s (Finney

Table 3
Cumulative community economic impacts (value added) from 2019 to 2100 for the six case study counties. Values provided for the Simple Scenario (all transitioning land becomes dryland crops) and the Land Use Suitability Scenario (transitioning land can be either dryland crops or pasture depending on soil).

State	County	Simple Scenario	Land Use Suitability Scenario	Difference ($)	Difference (%)
Colorado	Yuma	$317,587,291	$210,277,317	$107,309,974	33.8
Kansas	Finney	$619,368,000	$532,667,200	$86,700,800	14.0
Nebraska	Dundy	$268,851,159	$147,857,833	$120,933,327	45.0
New Mexico	Roosevelt	$99,165,510	$76,737,528	$22,427,982	22.6
Oklahoma	Texas	$716,591,688	$628,007,929	$88,583,759	12.4
Texas	Dallam	$1,758,435,347	$1,297,450,427	$460,984,919	26.2

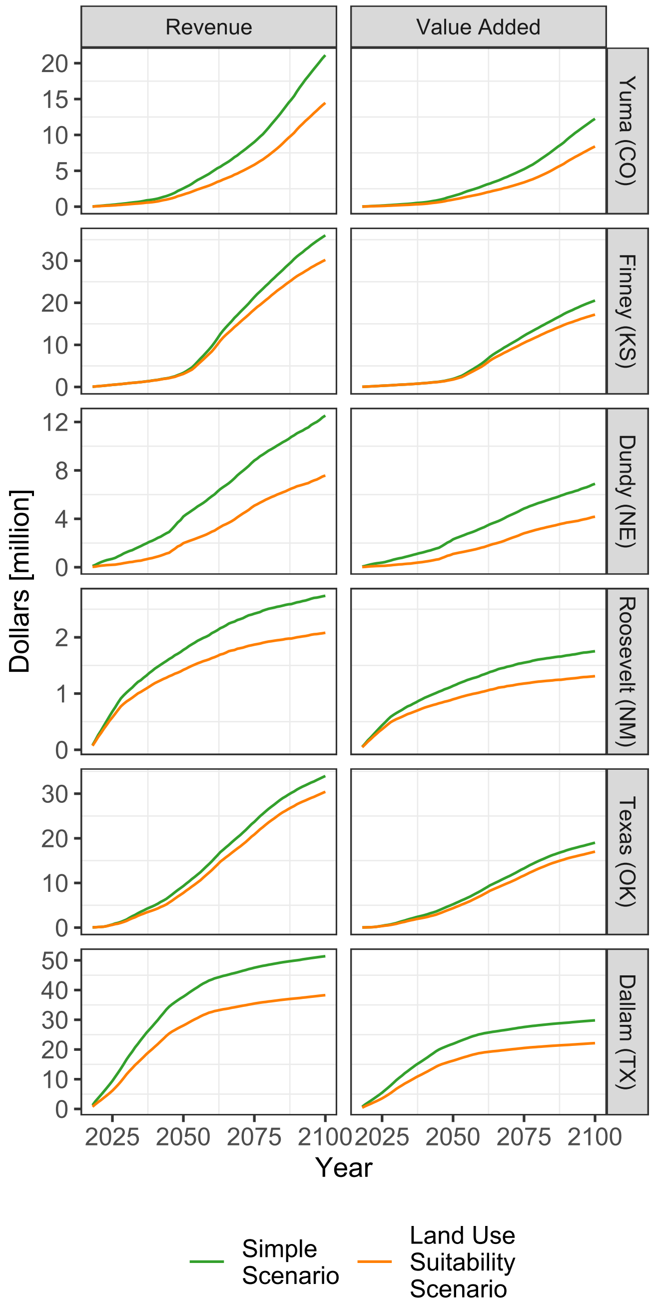

Fig. 5. Annual revenue (left) and value added (right) estimated under the Simple Scenario (all transitioning land becomes dryland crops) and the Land Use Suitability Scenario (transitioning land can be either dryland crops or pasture depending on soil) from 2019 to 2100 for each of the 6 focal counties.

County). This indicates that near-term depletion-driven transitions will primarily be on higher-quality soils capable of supporting dryland crops, while long-term depletion-driven transitions will be concentrated on lower-quality soils (see also Fig. 3). In contrast, the two scenarios diverge almost immediately in Dundy and Dallam counties (Fig. 5), indicating near-term loss of irrigation in lower-quality soils. In all counties, however, the difference between the Simple Scenario and Land Use Suitability Scenario grows larger through time, reflecting the increasing aggregate area converted to pasture over time. Combined, these results suggest that land use conversions will impact future generations more than the current generation. Similarly, it indicates that the error caused by failing to account for land use suitability (particularly soil quality) in economic planning models will have significant spatial and temporal variability, though errors generally will increase over time.

It is also worth noting that lost irrigated area is not directly proportional to the economic impact (e.g., Fig. 4). For example, Dundy County, NE, would lose only 15,600 ha of irrigated area, but due to the high percentage of land unsuitable for dryland crops (42 %) and high ratio of crop to pasture revenue, where cropland has 15.5x higher gross revenue than pasture, these losses lead to $220 million difference in gross revenue between scenarios. Roosevelt County, NM, on the other hand, would lose a similar 13,600 ha of irrigated area, but its lower percentage of pasture-suitable land (33 %) and lower crop:pasture revenue (3.7) result in only a $33 million difference between scenarios. This emphasizes the variable impact land use suitability can have across space and the importance of local-scale modeling to understand the economic implications of aquifer depletion.

4. Conclusions

Groundwater depletion threatens the viability of irrigated agriculture across the High Plains Aquifer, particularly in the southern, central, and western regions. Local economies need information on the timing, scale, and plausible replacement land uses in order to plan for diminishing irrigation capabilities. Here, we developed a high-resolution map time series projecting annual irrigation losses across the HPA through 2100 and find that 24 % of current irrigated acreage may undergo a forced transition to dryland crops or pasture by 2100 due to groundwater depletion. By mapping the locations of these losses, we were able to estimate likely non-irrigated land use for these transitioning irrigated areas based on underlying soil and physiographic properties. Overall, our land use suitability model indicated that 13 % of these future depleted areas would be unable to support dryland crop production. These limitations are not typically recognized in studies to date; instead, all irrigated lands are generally assumed to transition to dryland cropping. Using six county-scale case studies, we found that soil-driven land use suitability limitations which would require conversion to pasture rather than dryland crops can reduce future revenue by an order of magnitude. Overall, failing to account for land use suitability can underestimate economic impacts of aquifer depletion by 12–45 %, depending on county attributes. Moreover, these economic impacts are not directly proportional to the area of irrigation losses, largely reflecting differences in the dryland crop types able to be grown in different settings.

We made several simplifying assumptions to demonstrate the importance of land use suitability. The future is dynamic and unknown; changes in irrigation technology, dominant crop mixes, farmer behavior, and climate could all affect our depletion timeframes as well as county economic parameters. Climate change, in particular, is likely to impact the study region and alter historical water extraction rates and cropping patterns. For example, climate change has and will continue to shift the geographic range suitable for different dryland crops (Cho and McCarl, 2017) and is expected to alter crop yields (Cotterman et al., 2018). In addition, future climate scenario analyses suggest that some areas may be unable to support even pastureland, possibly shifting to even less productive shrublands (Sohl et al., 2019). Agricultural systems are coupled human-natural systems (Liu et al., 2007) characterized by feedbacks, surprises, and complex global connections through food supply chains and international trade (Dalin et al., 2017). For example, land use can provide positive or negative feedbacks to groundwater levels, since studies have found that groundwater recharge drops substantially between irrigated and dryland cropping systems, and that grasslands or pasture have lower recharge than either cropping system (Scanlon et al., 2012; Riley et al., 2019). Future work should seek to capture these system dynamics to better understand and manage the full range of impacts from depletion-motivated transitions away from irrigated agriculture. With this study, we used a scenario approach to highlight a key factor that warrants inclusion in future analyses.

Land use transitions have occurred repeatedly throughout human history, but often without proactive management to ensure the best possible outcomes for human well-being and the environment. The development of integrated models that can inform decision support tools and evaluate possible policy mechanisms in collaboration with community stakeholders are critical for managing land use transitions in water-limited regions (Dobrowolski and Engle, 2016; Maczko et al., 2016). Without forethought and strategic planning, a second Dust Bowl is not outside the realm of possibilities. Therefore, future research integrating soil quality and land use suitability into agricultural, hydrological, and economic models is essential to understand all aspects of the transition process and help guide the High Plains Aquifer areas into a sustainable social, environmental, and economic future.

Declaration of Competing Interest

None.

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Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at doi:https://doi.org/10.1016/j.agwat.2020.106061.

Data processing and analysis code along with the derived data necessary to reproduce analyses and figures is available at https://doi.org/10.5281/zenodo.3661369, with the exception of the IMPLAN economic software. Summarized results from the economic analyses needed to reproduce manuscript figures are provided.

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