REDEFINING WATER AND LAND MANAGEMENT STRATEGIES FOR THE EARLY 21st CENTURY

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ABSTRACT

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Water and land are key components to environmental and economic sustainability, and both their quality and availability serve as predictors for long-term socioeconomic development. Current water and land use patterns indicate that existing resource management strategies do not successfully capture the complex drivers that accelerate resource use; critical knowledge gaps exist between these management strategies and the behaviors of end-users. In other words, the strategies in place to conserve water and land resources have largely not proved effective at the regional scale. This dissertation identifies the areas where management strategies have missed their objectives, and converts these findings into practical steps for future management plans. In addition to the findings of each individual chapter, I summarize the redefinition of management strategies into four key components. (1) Management strategies must perform within a comprehensive framework or unaccounted for areas will be exploited. (2) Incentives promote actions away from unaccounted for areas, where restrictions push behaviors toward these areas. (3) Adequate values must be assigned to water and land resources in order to promote economically meaningful incentives. (4) Strategies must be designed around the objectives of maintaining the livelihoods of end-users.

To Sarah, my wife, my friend.

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Abstract

Successful water and land management strategies for the 21st century must integrate both the natural and social sciences to capture the processes and drivers that characterize and lead to resource availability, consumption, and use. In expenditure terms, the natural sciences describe the amount of resource available, and the social sciences describe the rate at which the resource is likely to be consumed. Both fields are necessary to achieve long-term resource sustainability, and both must be informed of the other to maintain a socially relevant, systems perspective when designing new management strategies. This chapter introduces current agricultural land and water management challenges in the context of the common social theories that describe resource use and consumption. Each remaining chapter investigates a knowledge gap introduced by common social theories with four primary topics: (1) water, (2) land, (3) technology, and (4) prediction. Results from this dissertation can be used to better inform management decision-making, in addition to the scientific implications derived from each study.

1. Introduction

The population of the United States is projected to reach 400 million by the year 2050, a 20% increase from the year 2015 (Colby and Ortman, 2015). As the U.S. population increases, so will the demand for food, fiber, and fuel. Meeting this demand relies heavily on agricultural production, which is linked to land and water availability. However, as population increases, land availability decreases and water stress increases (Chen, 2007; Vörösmarty et al., 2000). If land and water resource stability is to be achieved while matching the growing demand, then resource management strategies must be aware of how environmental systems react to additional stress. This includes the multiscale understanding of physical and social systems, as well as the interplay between them. Land and water management strategies for the early 21st century must evaluate how past management strategies have performed, determine notable gaps in those strategies, further investigate natural and social systems, and redefine the approach of developing new strategies to meet the demand posed by an increasing population.

A major challenge to resource management is that policies can quickly become outdated. In just the last decade, the United States experienced an economic recession (Grusky et al., 2011), a population shift towards urban areas (Seto et al., 2012), increased climate awareness (Bierbaum et al., 2013), and widespread technological development, all of which heavily influence the objectives of management strategies as well as the public perceptions and values that drive their intended purpose. Recent attempts to overcome outdated policies have emphasized multidisciplinary research to remain relevant by capturing integrated processes (Bouwer et al., 2000), but few strategies have effectively managed a multisystem framework. Management strategies can no longer afford to think traditionally about natural resource conservation. Instead, new strategies must look to integrate social science theory with the natural sciences to develop policies that are relevant at multiple timescales and can capture complex natural and social systems.

Natural and social system dynamics are particularly relevant in agricultural landscapes as the exploitation of natural resources is directly linked to end user decision making. For example, natural science methods can quantify water availability and use for irrigation (de Fraiture and Wichelns, 2010), while social science methods can explain why farmers choose to use the available water (Sanderson and Frey, 2015). In budget terms, natural sciences quantify inputs and outputs, while social sciences describe the rate and motivation at play between input and output. Total budget management, including the motivations and rates within a resource budget, is a critical objective for natural resource conservation. Integrated management that couples social and natural systems can anticipate end user dynamics to capture and remain relevant at extended timescales. By integrating these systems, management strategies become proactive towards future conditions rather than reactive to observed trends. This chapter identifies several key social theories related to water and land management as a motivational framework for the studies described in the remaining chapters.

2. Environmental Sociology Theory

2. 1. Treadmill of Production

The demand for water use across major aquifer regions (e.g., the High Plains Aquifer) can be described using the environmental sociology theoretical model known as the treadmill of production (ToP) (Schnaiberg, 1980). The ToP is primarily an economic exchange theory that links directly to natural resource extraction (Gould et al., 2004), and in the case of the High Plains, water extraction. ToP theory demonstrates that as crop production on major aquifer regions becomes more efficient by means of improved technologies (e.g., irrigation) then the consumption of crops will increase in global markets. As consumption increases, so does the demand for crop production, and subsequently, water extraction. This causes a cycle of increased demand where the intensification in water extraction is required to support the spinning production treadmill.

Slowing of the ToP through reductions in water use then poses risks to the two operations most dependent on its continued spinning: (1) the livelihood of local farmers and (2) the global economy built on food, fiber, and fuel. The central problem is that both have used the ToP as a means for economic stability. Local farmers use water extraction as a mechanism for efficient crop production and thus functional incomes, and global markets use efficient production as a mechanism for economic growth based on the cheaper produced goods. As the global economy expands, farmers must extract more water to maintain their living standards (Sanderson and Frey, 2015). Since both of these operations exist within a treadmill framework dependent on expansion, sustainable management is very difficult to achieve as the economic momentum is towards water supply decline (Sanderson and Frey, 2015).

2.2. Ecologically Unequal Exchange

Many major aquifer regions are net-exporting regions (e.g., water and biomass), while global consumption is net-importing (e.g., fiber and fuel). In theory, this exchange would be economically equal if the revenues earned in global markets were reflected in the profits returned to the local farmers distributing the exports. Equal exchange requires compensation for the commodity itself, including reimbursement for the resources required to produce the commodity (e.g., groundwater). In reality, farmers are compensated based on the market value of the final product, and the value associated with groundwater depletion is often not reflected in this market price. This discrepancy in economic exchange produces a scenario where both the resources and commodity product are exported from the region, and the resources used to produce the commodity receive minimal economic return (Sanderson and Frey, 2015).

The exchange of resources from undercompensated areas (e.g., farmers on major aquifer regions) to higher-income, core areas (i.e., global markets) can be described using ecological unequal exchange theory (Bunker, 1984; Rice, 2007). For example, the incomes of farmers in Kansas have remained stagnant despite massive groundwater extraction from the High Plains Aquifer (Sanderson and Frey, 2015). Given the structure of unequal exchange, farmers across the High Plains are required to degrade the exact resource that is sustaining their livelihood. When linked back to the production treadmill, ecologically unequal exchange exacerbates resource extraction as increased groundwater pumping is required to sustain the incomes of farmers across the High Plains.

2.3. Metabolic Rift

Metabolic rifts occur when there is an increasing gap between society and the natural resources available to sustain it. Thus, metabolic rift theory (Foster, 1999) highlights the discrepancies between places where the metabolism of resource consumption is greater than the replenishment of a resource. For example, a metabolic rift exists on the High Plains where farmers extract groundwater at rates unsustainable for future use. The metabolic rift on the High Plains is directly exacerbated by the production treadmill and the unequal exchange demanded by global markets (Sanderson and Frey, 2015). As groundwater depletion continues, both the livelihoods of farmers and the global markets dependent on major aquifer regions for crop production become further separated from the natural resources necessary to sustain them, and the rift becomes larger.

Water extraction has remained high on the High Plains, despite groundwater levels that have approached exhaustion (Haacker et al., 2015; Scanlon et al., 2012). A critical challenge to achieving sustainability across this region is that farmer revenues are instantly threatened if water use is restricted. In other words, the livelihoods of High Plains farmers are directly linked to the deepening of the metabolic rift. In this scenario, metabolic rift theory demonstrates that the economy and natural environment are in conflict (Sanderson and Frey, 2015). Particularly on the High Plains where agriculture is dominant, declines in groundwater availability directly compromise the regional way of life. Future water management must resolve this conflict or the rift will continue to grow.

2.4. Jevons Paradox

It is widely suggested that improved efficiency in irrigation technology will lead to a reduction in overall water use, but opposite has been found in major aquifer regions (Pfeiffer and Lin, 2014, Gomez et al., 2013). On the High Plains, water decline has continued at rates similar to past decades despite the introduction of efficient irrigation technologies (Haacker et al., 2015; Scanlon et al., 2012). While the purpose of efficient irrigation technology is to use less water, the result may create favorable economic conditions for farmers to irrigate more acreage at a similar cost to inefficient systems. As farmers irrigate more acreage, widespread water use exacerbates both groundwater decline and the metabolic rift. This relationship can be explained using Jevon's paradox (Alcott, 2005).

Jevon's paradox demonstrates that as the efficiency of production increases, so will the consumption of the resource. Jevon's paradox captures a multistep cycle: (1) as the efficiency of production increases, the cost of production decreases, (2) as the cost of production decreases, the demand for production increases, (3) as the demand for production increases, the overall consumption increases, and (4) as the overall consumption increases, the demand for efficient production increases, ultimately restarting the cycle. This paradox is confirmed by the steady increase in irrigated acreage across the High Plains in recent years (Pfeiffer and Lin, 2014), despite the increased knowledge of groundwater decline. Jevon's paradox simply demonstrates that efficient irrigation is not the sole solution to groundwater conservation. Instead, water management may prove more effective at conservation when targeting outside drivers.

2.5. Second Contradiction of Capitalism

The second contradiction of capitalism (O'Connor, 1991) demonstrates that a capitalist economy will exploit natural resources to the degree that production no longer becomes economically feasible (Stroshane, 1997). Across major aquifer regions, this describes groundwater extraction until it is too expensive to pump for a profit or is extracted beyond irrigable capabilities. Once extracted beyond use or too expensive for economic growth, groundwater stress will shift to other regions of the world to meet the demands derived from capitalistic enterprise. Similar patterns can be seen in other valuable resources such as oil and coal. HPA groundwater can be perceived as a nonrenewable resource when applied within the framework of the second contradiction of capitalism.

The challenge for water management is that groundwater recharge rates are highly variable across regions, though groundwater is often managed as renewable. For example, recharge across most of the northern portions of the High Plains Aquifer have generally matched extraction rates, demonstrating that groundwater can be treated as a renewable resource in these portions (Haacker et al. 2015). Yet, other portions of the aquifer have also been extracted at rates greater than recharge, demonstrating that the groundwater in these portions should be treated as a nonrenewable resource under current practices. If management schemes are established based on treating groundwater as renewable, then the nonrenewable portions of the aquifer would be mismanaged and extracted. Management strategies should consider the second contradiction of capitalism when creating schemes as a way to confirm that groundwater resources will be used until they are no longer economically feasible if allowed to do so. Future strategies will need to be aware that unprotected resources will be exposed to overconsumption due to the second contradiction of capitalism.

2.6. Farmer Behavior

Future management strategies should also consider the behavior of stakeholders directly involved with water use. For example, farmers have a very strong sense of place (Cantrill, 1998), and this links directly to place attachment, place identity, and place dependence (Mullendore et al., 2015). For the High Plains Aquifer, farmer livelihoods are heavily place dependent as groundwater pumping allows for high-production agriculture in an otherwise arid landscape. Given the long history of extensive groundwater pumping, the identity of the High Plains has been shaped by high-yield crop development. If farmers have a strong sense of identity towards water use, then they will be more reluctant towards policy aimed at water use reductions. However, farmers are also willing to adapt new conservation practices based on trust (Mase et al., 2015). If organizations or affiliations that farmers trust are promoting water conservation, then the behavior is more likely to be adopted by local farmers. Conservation strategy adoption is more likely with small agricultural landowners, though the largest barrier to adoption is perceived cost (Perry-Hill and Prokopy, 2014). Farmers rooted in water use dependence will need well-developed trust and risk assurance prior to management strategy change.

2.7. Stakeholder Involvement

Water management at the local and regional scales should involve stakeholders who are directly impacted by new conservation strategies (Davidson et al., 2015). Stakeholders should be involved in the creation of new planning and management strategies to holistically capture the values and desires of all users impacted by the new policies (Carter et al., 2005). In addition, management strategies must also consider the institutional needs of the new schemes including localized staff to manage the system, the financial resources necessary to carry out the strategies, and the availability of the tools necessary to enforce new strategies (Carter et al., 2005). It is also important to realize that local stakeholders can be sources of information and not just knowledge recipients (Armitage et al., 2015). Local farmers may in fact have valuable insights overlooked by management plans designed without contribution by the local stakeholders. Particularly across major aquifer regions, localized knowledge can prove beneficial for a heterogeneous landscape. Engaging with local stakeholders increases the likelihood of holistic strategy development.

2.8. Scales of Management and Exchange

In an effort to improve the overall management of groundwater, most High Plains states have created local management zones where water is controlled at the local level (Mossmann, 1996; Fipps and Pope, 1998). The problem with localized management is that water use decisions are made at the local scale, but decision-making is influenced by markets reacting to the global scale. Moreover, the scale of management continues to decrease while the scale of global demand continues to increase (Sanderson and Frey, 2015). In other words, the world market is becoming more global, but management plans are becoming more local. In this scenario, local management zones are asked to manage a resource that is heavily influenced by drivers much larger than what can be captured at the local level. When coupled with the ToP, unequal exchange, and metabolic rift theory, management zones at the local level are tasked with the objective of sustainably managing a resource when the livelihoods of the local farmers are economically dependent on depleting it (Sanderson and Frey, 2015). Future management strategies, even at the local level, will need to capture global scale drivers to successfully conserve water at the local scale.

2.9. Research Implications for the Social Sciences

Based on the discussed social theory, two major research questions still need to be answered: (1) what is the value of water and (2) to what degree do farmers believe water conservation is necessary to sustain future livelihoods? One of the largest implications of unequal exchange theory is that water is being exported out of major aquifer regions at a cost much lower than the cost of aquifer depletion. Researchers have suggested that market prices include the value of the water necessary to produce the commodity (Sanderson and Frey, 2014), but the quantification of water value is still incomplete. The term “virtual water” has been applied to the export of water through produced goods (Hoekstra and Hung 2002), but placing a global price on water is complex and has not been established. Using the ToP, unequal exchange, and metabolic rift theory, it is easy to see why farmers continue to extract groundwater despite the threats this poses to their livelihoods. The main question that needs answering is then to what degree farmers are okay with exacerbating groundwater loss. If farmers truly believed groundwater extraction was against their best interest, then they would no longer extract groundwater and accept the economic implications aligned with a transition to dryland farming. To some degree, farmers still find a greater incentive to use water than to save water. Understanding this turning point between using and saving may prove critical as groundwater levels continue to decline. By answering this question through future social science research, social theory can be coupled with natural science to mutually progress both fields and improve management schemes designed for holistic frameworks.

3. Dissertation Structure

Social theory demonstrates that water management is more complex than just the balancing of a regional water budget. Instead, social drivers heavily influence the rates and motivations of water exchange within the regional budget. Capturing the rates and motivations of exchange is critical for the development of effective management strategies. However, social theory cannot generate effective management strategies alone. Natural science is required to define the extents of resource availability and describe the framework for which social theory is at play. More work is needed in the natural sciences to define the historical and regional consumption trends and limits to resource availability. When combined, this type of approach can also be used to quantify the economic risks associated with resource use, which serves as a driving force to new strategy adoption. Overall, the objective is to conserve resources while maintaining livelihoods, and this objective requires the integration of the natural sciences into social theory.

This dissertation investigates four management topics, each derived from notable gaps in social theory: (1) water, (2) land, (3) technology, and (4) budget (Chapters 2-5, respectively). The dissertation outline is displayed in Figure 1, where each major topic includes a series of sub-topics highlighting the main research questions in each study.

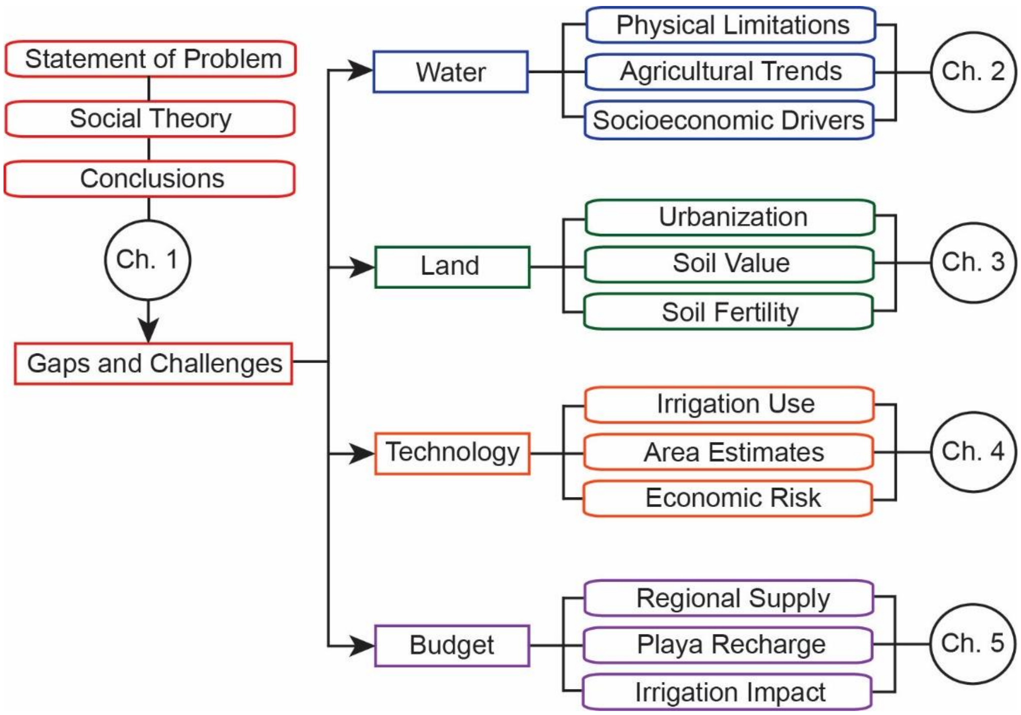

Figure 1. Conceptual diagram of the dissertation outline. Each topical category and subsequent chapter is derived from a land or water management challenge first described by social theory, and each chapter is subdivided into focus areas discussed within each study.

In Chapter 2, I investigate the main drivers associated with water use across the High Plains Aquifer as means to improve the conceptual framework for future water management strategies. In Chapter 3, I investigate the integration of a soil-based urbanization development strategy as a way to reserve valuable farmland to meet future resource demands. In Chapter 4, I investigate historical agricultural production data to isolate irrigated and dryland production as a way to evaluate the economic risk and trends linked to irrigation applications. Lastly, in Chapter 5, I investigate water flux across the Southern High Plains Aquifer region as a way to predict recharge and total water supply available for agricultural use. Collectively, this work aims to redefine water and land management strategies for the early 21st century with the objective of conserving resources while satisfying societal demands.

4. Comprehensive Dissertation Conclusions

Future water management strategies require a global perspective placed within a localized end-user framework. The challenge to successful water and land management is that the users of these resources are often trapped in a degrading cycle, where resource consumption is the only way to maintain livelihoods, while resource consumption is the sole driver leading to livelihood collapse. Future management strategies can benefit from the integration of social and natural sciences to understand the interplay between resource use and the decision-making that exacerbates resource degradation. Within this coupled approach, social theory can identify the motivations behind resource use decision-making, while natural science research can define the framework in which social behaviors are allowed to perform. Quantifying social behavior is a step in the right direction for establishing baselines for successful management strategies, but future work will require the integration of both the natural and social sciences to achieve resource sustainability and address resource demands. Water and land management strategies fall under the broad umbrella of "coupled human and natural systems", indicating that management strategies will need to fully understand the complexity of the drivers leading to resource use, consumption, and extraction in both systems.

In addition to the merit and conclusions discussed in each remaining chapter, this dissertation can be summarized into four main conclusions:

(1) It is paramount that future management strategies are comprehensive in their objectives. For example, water and land use decision-making is highly correlated to the economic incentives associated with resource use. As a result, any mismanaged or unaccounted for area will be exploited to capitalize on these incentives. For example, the biofuel mandate led to an increase in corn production in areas not suitable for corn (Chapter 2), and when coupled with improved irrigation technologies, led to an increase in water use in areas not traditionally irrigated (Chapter 4). A comprehensive approach will require holistic and integrated research.

(2) Incentives are more effective than restrictions to manage water use. Future strategies would benefit by restructuring water management based on economic opportunities instead of overuse restrictions. The impact of conservation policies and litigations are not reflected in overall consumption trends, but responses to increased revenue opportunities are very prevalent (Chapters 2 and 4). Using economic value as a basis for management strategies also proved effective by placing a higher penalty on the consumption of valuable resources (Chapter 3).

(3) If economic opportunities are to be integrated into management strategies, then adequate values must be assigned to water and land resources. The valuation of land and water will require a complete inventory of the resources available in a region (Chapter 5), as well as the change in supply due to complex drivers such as policies (Chapter 3), improved technologies (Chapters 2 and 4), climate controls (Chapter 2), and land use practices (Chapter 5).

(4) All the while, future management strategies need to consider the main motivation for implementation – preserving resources and livelihoods for people. With this in mind, it is important for management strategies to strongly consider the interests of the on-the-ground

decision makers (e.g., farmers), as they are the ones most economically affected by new strategies and are the ones who will implement the change. Future strategies will be far more successful if local end-users believe in the strategy and are passionate about resource conservation.

CHAPTER 2

Complex water management in modern agriculture: Trends in the water-energy-food nexus over the High Plains Aquifer

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Abstract

In modern agriculture, the interplay between complex physical, agricultural, and socioeconomic water use drivers must be fully understood to successfully manage water supplies on extended timescales. This is particularly evident across large portions of the High Plains Aquifer where groundwater levels have declined at unsustainable rates despite improvements in both the efficiency of water use and water productivity in agricultural practices. Improved technology and land use practices have not mitigated groundwater level declines, thus water management strategies must adapt accordingly or risk further resource loss. In this study, we analyze the water-energy-food nexus over the High Plains Aquifer as a framework to isolate the major drivers that have shaped the history, and will direct the future, of water use in modern agriculture. Based on this analysis, we conclude that future water management strategies can benefit from: (1) prioritizing farmer profit to encourage decision-making that aligns with strategic objectives, (2) management of water as both an input into the water-energy-food nexus and a key incentive for farmers, (3) adaptive frameworks that allow for short-term objectives within long-term goals, (4) innovative strategies that fit within restrictive political frameworks, (5) reduced production risks to aid farmer decision-making, and (6) increasing the political desire to conserve valuable water resources. This research sets the foundation to address water management as a function of complex decision-making trends linked to the water-energy-food nexus. Water management strategy recommendations are made based on the objective of balancing farmer profit and conserving water resources to ensure future agricultural production.

Keywords: High Plains Aquifer, water management, irrigation, agriculture, economics, policy

1. Introduction

Crop production across the High Plains Aquifer region (High Plains) in the central United States has an annual market value greater than $20 billion—approximately 10 percent of the entire U.S. crop value (NASS-USDA, 2012). Irrigation is essential to much of this crop production. Irrigated agriculture across the High Plains accounts for 30 percent of all irrigated acreage in the U.S. (Dennehy et al., 2002), and 97 percent of High Plains irrigation water is extracted from the High Plains Aquifer (HPA; Maupin and Barber, 2005). Due to extensive irrigation, groundwater levels across large sections of the HPA have been declining for decades, particularly in the southern section where the aquifer is thin and irrigation demand is high (Haacker et al., 2015; McGuire, 2009; Scanlon et al., 2012). Future decades are forecast to bring more widespread groundwater declines, effectively depleting broad regions of the HPA if current practices continue (Haacker et al., 2015). Major reductions in water availability would result in enormous consequences for food and energy production.

At the core of agricultural water management challenges is the water-energy-food nexus. Acting within this nexus across the HPA are the individuals and institutions that adapt to address the realities of groundwater depletion. These include creating and adopting new technologies, developing and planting different cultivars, shifting cropping patterns, implementing new policies, expanding monitoring, and pushing toward more efficient use of limited resources. These strategies have been designed around the objectives of increasing crop yields, decreasing production costs, improving or maintaining soil fertility, and reducing environmental impacts (Edwards, 1989; Stuart et al., 2015). They can be generalized into two broad focus areas: (1) water conservation to both use less water and be more efficient in application, and (2) water productivity to maximize the return on water use. Water conservation research has focused on strategies such as deficit irrigation (Fereres et al., 2007; Geerts and Raes, 2009), irrigation technologies (Colaizzi et al., 2009; Howell, 2001), rainfed agriculture (Rockström et al., 2010; Rosegrant et al., 2002), and land management practices (Bossio et al., 2008; 2010). Water productivity research has focused on improved seed genetics (Hu and Xiong, 2014; Passioura, 2004), variable rate irrigation (Basso et al., 2013; Evans et al., 2013), and intraseason water management through irrigation scheduling and soil moisture monitoring (Aguilar et al., 2015), vegetation indices (Basso et al., 2004), and tillage practices (Derpsch et al., 2010). Despite this increased emphasis toward groundwater conservation among researchers, and new technologies and strategies that can greatly improve water productivity, groundwater supplies across the HPA continue to decline at unsustainable rates (Haacker et al., 2015; Scanlon et al., 2012).

Historically, water management strategies have targeted water use drivers within three major domains: (1) physical (e.g., climate, geology), (2) agricultural (e.g., crop type, tillage practices), and (3) socioeconomic (e.g., groundwater doctrines, market values) (Pimental et al., 1997). However, water use drivers in modern agriculture are too complicated to be regulated individually within these separate domains. For example, changes in precipitation patterns have direct implications on irrigation scheduling and applications (Lorite et al., 2015), improved technologies allow for innovative and heterogeneous farming practices (Steven and Clark, 2013; Zhang and Kovacs, 2012), and crop prices respond to changes in global market demands (Rosegrant, 2008). Furthermore, drivers within these domains each influence short- and long-term water use decisions in ways that have not been addressed in static water management strategies (e.g., climate variability, government incentives, and annual crop insurance plans). Moreover, water use drivers across these domains are inherently linked, making it impossible to implement temporally relevant water management strategies in one domain without impacting another.

There are clear gaps in current water management strategies across the High Plains, as evidenced by the increase in both crop production and water use despite the reality of groundwater depletion (NASS-USDA). Nowhere is agricultural water management more prevalent than in the water-energy-food nexus of the HPA, making the region ideal to learn how complex management domains influence water use and decision-making. This study provides a comprehensive overview of the major drivers of water use across the HPA through a novel synthesis of data and an in-depth review of the relevant literature. We examine drivers in the physical, agricultural, and socioeconomic domains in contrast to the historical approach. Furthermore, within each domain, we analyze water use trends and examine how these drivers interact to influence water use decisions. We then synthesize across domains to present a framework for maintaining long-term aquifer supplies through improved agricultural water management strategies across the water-energy-food nexus.

2. Methods

This study synthesizes extensive agricultural databases along with the relevant water management literature across the HPA. When used, specific processing techniques are discussed within corresponding sections. Sections 3, 4, and 5 compile individual water use drivers or driver categories into major domains, where each subsection represents a major driver set or focus area. Subsections are selected according to the most significant topics for water supply or water use across the region, as a complete synthesis of these drivers is necessary to formulate water management suggestions and highlight areas where water resources are exploited. All drivers at every spatial and temporal scale may not be included, as our subsection lists are representative of and relevant to large scale management schemes. We derive our conclusions based on the trends found within and across each domain, and we make management suggestions based on the goals of maintaining farmer profit and achieving long-term aquifer sustainability.

3. The Physical Domain

The physical domain defines the limits of the water-energy-food nexus. For example, food production requires both energy and water. If water is limited, so will be the ability to increase crop yields. Thus, balancing components within the nexus to find the combination where production is highest and resource expenditures are lowest over time is critical for sustainable agriculture. A required step to reach this ideal nexus status is to assess total water availability and supply through time. Here, we analyze the major physical drivers that impact water availability and supply, and we highlight the trends that have the most influence on long-term sustainability goals.

3.1. Geology, Soil, and Land Cover

The HPA (450,000 km^2; Qi, 2010) is located in the west-central United States and spans portions of eight states: South Dakota, Wyoming, Nebraska, Colorado, Kansas, Oklahoma, New Mexico, and Texas (Figure 2A). Given its size, the HPA is often divided into three geographical areas, each with unique physical characteristics: the Northern High Plains (NHP; 249,509 km^2, Central High Plains (CHP; 127,168 km^2), and Southern High Plains (SHP; 75,921 km^2). At 3,750 km^3 of total water volume in 2012 (Haacker et al., 2015), slightly larger than the volume of Lake Huron, the HPA remains one of the largest known freshwater aquifers in the world. The total volume of water estimated within the NHP is ~2,940 km^3, the CHP is ~635 km^3, and the SHP is ~171 km^3. However, groundwater is being recharged at rates far below annual withdrawals in the south and central portions of the aquifer.

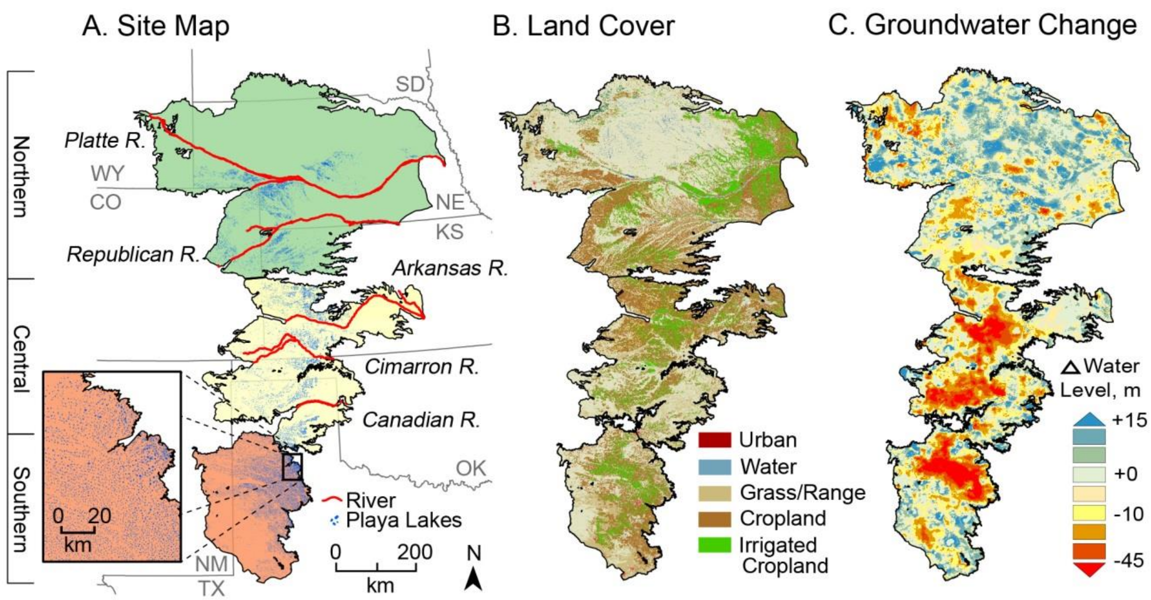


Figure 2. The High Plains and aquifer decline. A) Site map of the HPA and its three main regions. B) Land cover across the HPA region, dominated by range and cropland (NLCD, 2011). C) Interpolated groundwater level declines compared to predevelopment levels (modified from Haacker et al., 2015).

The High Plains have a semi-arid, temperate climate, with surface elevations that follow a west-east gradient from ~2,400-m in the west to ~350-m in the east (Dennehy et al., 2002); local relief is generally very low. Soil characteristics follow a general gradient of high permeability in the NHP (Dennehy et al., 2002; Gutentag et al., 1984) to low permeability in the SHP (Dennehy et al., 2002; Reeves Jr., 1970). Native land cover includes short and medium grass prairies, though large sections of modern land cover have transitioned to cropland (Figure 2B) with the major crop choices of corn, sorghum, winter wheat, soybeans, alfalfa, and cotton (Dennehy et al., 2002). Crop selections follow a general gradient of water-intensive crops in the north (e.g., corn, soybeans) to less water-intensive crops in the south (e.g., cotton, winter wheat).

The other major land use type across the region is livestock rangeland (primarily cattle; Dennehy et al., 2002). Collectively between cropland and rangeland, 94% of the High Plains is considered agricultural land (Figure 2B).

3.2. Hydrology and Hydrogeology

Several hydraulically-connected permeable units collectively form the HPA complex (Gutentag et al., 1984; Knowles et al., 1984); the largest of which is the Ogallala Formation, or Ogallala Aquifer, a name often used interchangeably with HPA. The Ogallala Aquifer underlies nearly 77 percent of the HPA area, with most of the remaining area composed of the Brule, Arikaree, Great Bend Prairie, and Equus Beds aquifers. Hydraulic conductivity and specific yield across the HPA vary from 1 to 105 m/day and 3 to 35 percent, respectively (Gutentag et al., 1984), resulting in highly variable groundwater yields across the aquifer. Saturated thickness ranges from 0 to 300-m but has drastically declined since predevelopment; average saturated thickness is approximately 60-m. Depth to water is generally from a few to 150-m, and average depth to water in 2012 was 30-m for the NHP, 44-m for the CHP, and 41-m for the SHP.

While groundwater supply in the NHP has been fairly stable since predevelopment, the CHP and SHP have experienced extensive groundwater depletion due to extensive groundwater pumping (McGuire, 2009). Peak groundwater level declines have reached more than 45-m in portions of the CHP and SHP (Figure 2C), while average declines by state for portions of the HPA are: 14-m in Texas, 9-m in Kansas, 6-m in Oklahoma, 5-m in New Mexico and Colorado, and (Haacker et al., 2015). Average groundwater declines in the NHP have been less than 0.5-m in both Nebraska and Wyoming (McGuire, 2009; Scanlon et al., 2012; Haacker et al. 2015), although areas of extensive groundwater withdrawals are common. Collectively, nearly 410 km^3

of water has been depleted from the HPA since predevelopment (Haacker et al., 2015), which is approximately the volume of Lake Erie.

Total annual surface water flow entering the HPA region is ~2.5 km^3 per year (Dennehy et al., 2002), though extensive groundwater depletion has resulted in a net loss in annual streamflow and surface water volume (Nativ, 1992; Scanlon et al., 2012). While major river systems flow from west to east across the NHP and CHP, the SHP has few streams, and none flow consistently. Instead, surface water in the SHP is largely drained and stored in thousands of localized playa lakes that are most concentrated along the eastern margins of the region. These broad, shallow lakes can span up to 1-km in diameter (Osterkamp and Wood, 1987) and drain an estimated 90% of the SHP region (Nativ, 1992). Playa lakes exist across the entire High Plains (~61,000 lakes; Gurdak and Roe, 2010) but are much more prevalent in the SHP (~30,000 lakes; Osterkamp and Wood, 1987; Figure 2A).

Natural recharge in the NHP and CHP occurs primarily as precipitation percolation through permeable soils and leakage from surface water bodies (Weeks et al., 1988; Dennehy et al., 2002). Localized recharge in the SHP region largely occurs as percolation beneath playa lakes where water passes through dissolved or fractured caliche (Osterkamp and Wood, 1987; Scanlon and Goldsmith, 1997; Wood and Osterkamp, 1987). Areal groundwater recharge across the High Plains decreases following a gradient from north to south. Secondary recharge across some portions of the HPA also occurs as irrigation return flow where some of the excess applied water is returned to the aquifer (McMahon et al., 2006; Scanlon et al., 2005; Whittemore et al., 2015).

3.3. Regional Climate

The High Plains are located in a wet-dry climate transition zone (Koster et al., 2004) where soil moisture plays a critical role in modulating the energy and mass transport that impact the regional water cycle (Berg et al., 2014). This is particularly relevant in areas of high irrigation where modified soil moisture significantly impacts the regional hydroclimate through adjusted land-atmosphere interactions (Harding and Snyder 2012a; 2012b; Jódar et al., 2010; Lo and Famiglietti, 2013; Moore and Rojstaczer 2001; 2002; Pei et al., 2016; Qian et al. 2013). One major effect of increased soil moisture is on the Great Plains low-level jet (GPLLJ; Walters et al., 2014; Weaver and Nigam, 2011). The GPLLJ brings moisture into the region from the Gulf of Mexico and provides the main external moisture source for precipitation over the High Plains and central United States (Cook et al., 2008; Higgins et al., 1997; Pei et al., 2014; Tuttle and Davis, 2006; Weaver, 2007). At shorter timescales (event-scale), fluctuations in the GPLLJ prompt nighttime rainfall maxima during warmer seasons, where greater moisture convergence results in heavier precipitation (Carbone and Tuttle 2008; Pu and Dickinson 2014; Zhong et al., 1996).

Climate models project a decrease in warm-season precipitation (Cook et al., 2008; Maloney et al., 2014) and an increase in regional temperatures for the High Plains by the end of this century (Cook et al., 2008; IPCC, 2007). Historically, the High Plains receives ~50-cm of average annual precipitation (Crosbie et al., 2013), with a gradient from ~40-cm along the western border to ~70-cm along the eastern edge (Gutentag et al., 1984). Precipitation is projected to increase for the NHP and decrease for the SHP, and regional temperatures are expected to increase by 2 to 5°C (Crosbie et al., 2013; IPCC, 2007). Increased temperatures would likely favor increased evapotranspiration (Green et al., 2011), and a decrease in precipitation and increase in temperature would both likely exacerbate groundwater supply declines under current water use scenarios (Crosbie et al., 2013).

Extreme drought events have also become more frequent over the past 45 years (NLDAS-2). The average annual HPA precipitation fell below 305-mm five times since 1998, whereas this occurred just once from 1979-1998 (Figure 3). While reductions in annual precipitation are most extreme in the SHP, similar trends have been seen in the NHP and CHP. In particular, SHP precipitation fell below 100-mm during 2012-2013 regional droughts, and for the first time on record, precipitation simultaneously fell below 300-mm for both the CHP and NHP regions during the same drought period.

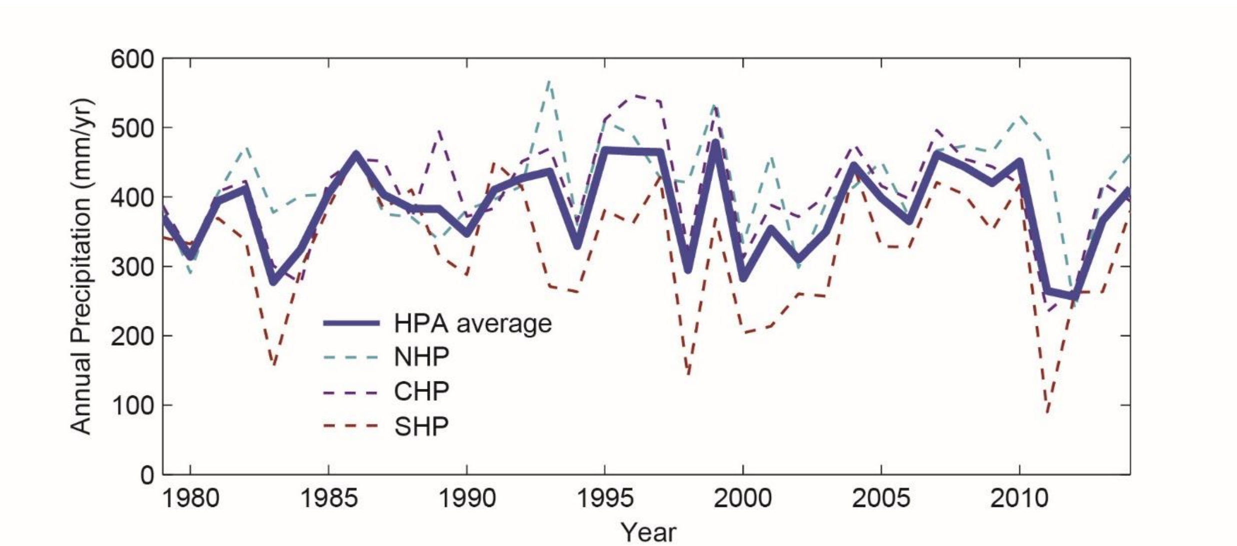


Figure 3. Average annual precipitation for the HPA and its three regions (NLDAS-2 forcing file A).

Discrepancies in the projected GPLLJ strengthening and subsequent precipitation decreases suggest changes in future climate regimes over the HPA (Maloney et al., 2014). Areas of the HPA that are currently limited by water availability will likely be the most affected by these changes (Ng et al., 2010). However, accurately capturing these patterns remains a challenge for predictive models even with knowledge of the major climate controls (Hoerling et al., 2014). For example, the 2012 severe Great Plains drought was suggested to be independent of these climate patterns and likely a result of atmospheric noise alone (Kumar et al., 2013). Future water management strategies would clearly benefit from improved climate prediction skills.

4. The Agricultural Domain

Crop yield in the agricultural domain is the primary indicator of resource efficiency within the water-energy-food nexus, given its dependence on both growing conditions and agricultural management practices. Generally, increased yields through time indicate improved technologies or agricultural practices that allow physical resources to be used more efficiently. However, improved efficiency is not always an indicator of sustainability. Increased crop yields may be a function of efficient practices, but that does not mean they are always less taxing on resources within the physical domain (e.g., water, soil). Cross-domain impacts must be considered to achieve sustainable management strategies in modern agriculture. In this section, we highlight the major agricultural drivers that impact water use, the primary component limited by availability and supply in the physical domain.

4.1. Soil Management

Soil management strategies focus on maximizing crop yield, maintaining long-term soil fertility, and mitigating environmental impacts such as nitrate leaching and greenhouse gas emissions. Example soil management strategies include conventional tillage versus no-till farming (e.g., Ghimire et al., 2012; Hobbs et al., 2008), crop rotations (e.g., Johnston, 1986; Odell et al., 1984), and off-season cover crops (e.g., Allen et al., 2005; Havlin et al., 1990). Conservation agriculture incorporates these land management strategies to increase soil fertility by preserving surface organic carbon, protecting soil from water runoff, and reducing soil loss by eliminating bare exposure (Basso et al., 2006; 2014; Hobbs et al., 2008). Managing soils to improve fertility reduces the demand for additional water applications. However, the potential for soil management to conserve water does not negate the substantial amount of water used for irrigation.

4.2. Irrigation and Crop Yield

A new synthesis of annual irrigated and non-irrigated yield since 1970 across the HPA was conducted using data from the National Agricultural Statistics Service (NASS-USDA), plotted in Figure 4. This synthesis uses annual county-level surveys of yields for the six major commodities grown across the HPA: corn, soybeans, winter wheat, alfalfa, cotton, and sorghum. The analysis of these data highlight: the considerable benefit of irrigation across the HPA (with little difference across subregions), the large increase in yields of corn, soy, and cotton over time due to improved management and crop genetics, and much larger annual variability in yields from dryland relative to irrigated production. The linear trends fit to this data from 1970-2014 show that non-irrigated and irrigated yields have increased by 133 and 96 percent for corn, 74 and 330 percent for cotton, 69 and 89 percent for soybeans, 17 and 26 percent for alfalfa, 11 and 13 percent for sorghum, and 4 and 27 percent for wheat, respectively. Today, *non-irrigated* corn yields are similar to the *irrigated* corn yields of 1970, and irrigated corn yields today are more than double non-irrigated yields (Figure 4A). Similar trends can be seen in cotton yields, although the gap between irrigated and non-irrigated yields has been increasing in recent years (Figure 4B). Alfalfa, sorghum and wheat yields have not rapidly increased since 1970, though irrigated yields are still approximately double the non-irrigated yields of these crops (Figure 4D-F).

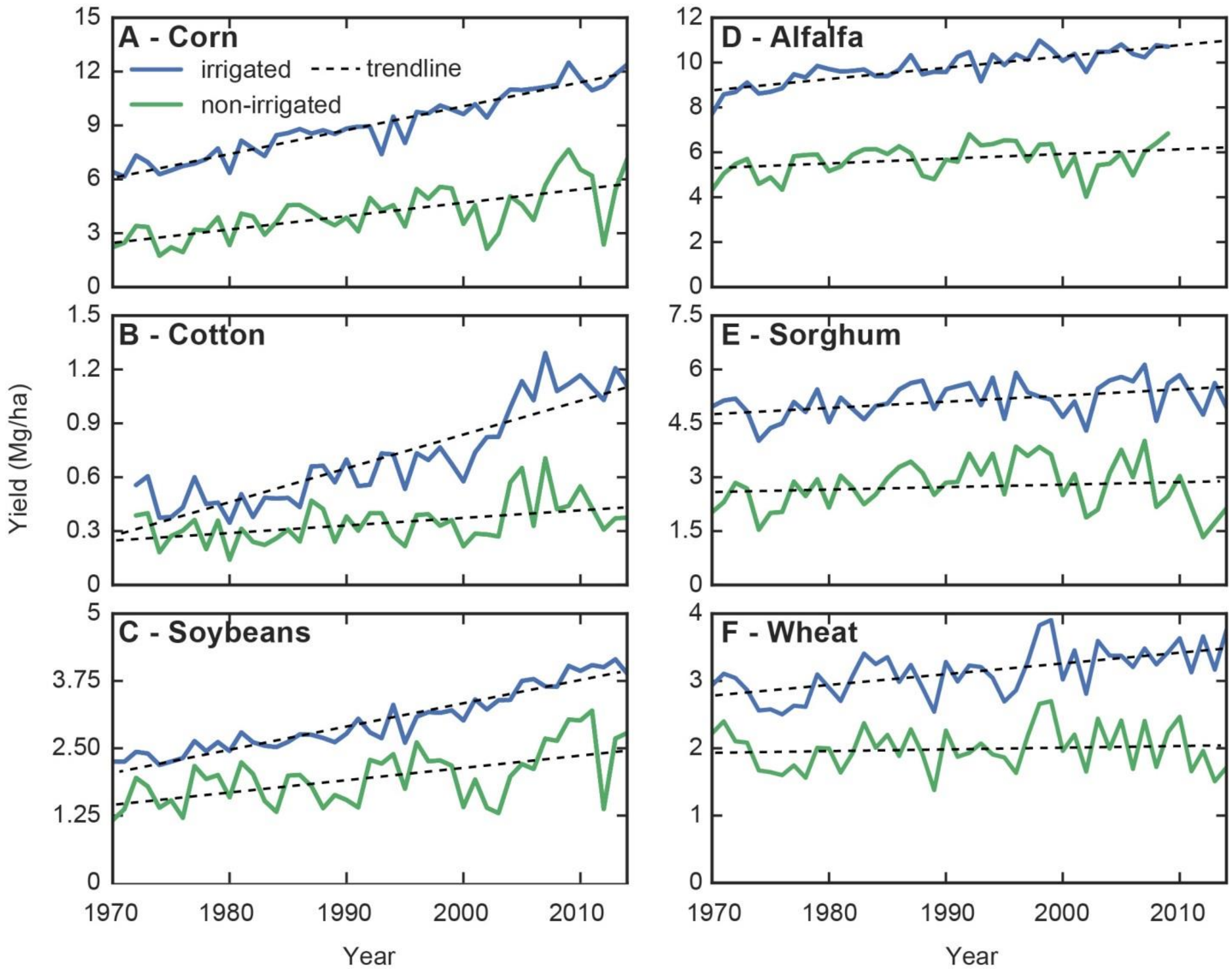


Figure 4. Irrigated and non-irrigated yields for the main commodities grown across the HPA (NASS-USDA). Alfalfa yields were not available after 2009.

In general, we found that irrigation increases yield by a factor of two to four times relative to dryland farming, a significantly larger yield increase than can be generated by other land management strategies (Colaizzi and Gowda, 2009; Colaizzi and Schneider, 2004). This boost in crop yield generates a major economic incentive to irrigate. Today, over 12 million acres of irrigated cropland are fed by the HPA for these six commodities (NASS-USDA). Irrigation over the HPA is so extensive, and high-yield agriculture is such a major component to the regional economy, that widespread transitions to dryland agriculture would cause severe economic consequences for the region (Colaizzi et al., 2009).

4.3. Crop selection

Water demand varies by commodity, and in general, the most water-intensive crops return the greatest short-term profit. For example, cotton demands approximately 69-cm of water for peak yields while corn requires almost 80-cm (Moore and Rojstaczer, 2001). This has resulted in both the widespread selection and the irrigation of more water-intensive crops, such as corn, across the High Plains. To investigate commodity selection trends, we calculated annual irrigated and total acreages from 1970 to 2014 for the six major commodities (NASS-USDA). We used a composite of annual county-level surveys, which may in some years only include a subset of commodities for each county, and the more complete bi-decadal Agricultural Census. Additionally, the noisier annual survey data were bias corrected to match the 5-year Census data. Biases in survey data are calculated for each county relative to the Census values as

$$bias_{year} = (Census_{year} - survey_{year})/survey_{year} \qquad \text{(eq. 1)}$$

and linearly interpolated between Census years. This annual bias was then converted to a multiplicative correction factor as

$$correction_{year} = bias_{year} + 1 \qquad \text{(eq. 2)}$$

which was then multiplied by the annual survey data for each county. Counties partially within the HPA were multiplied by the fraction of each county that falls within three HPA subregions. Adjusted acreages were then summed across the three HPA subregions (Figure 5).

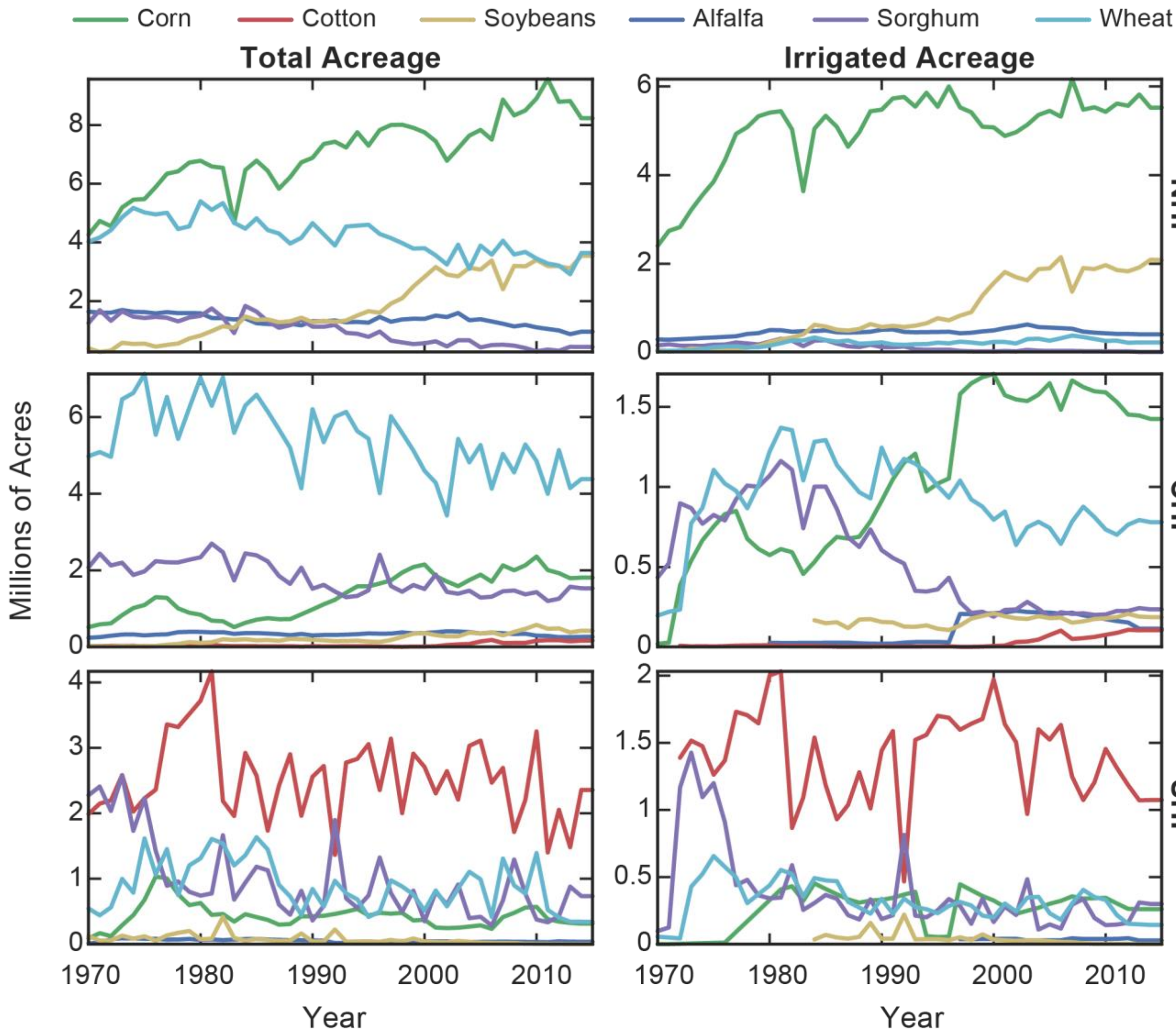


Figure 5. Total commodity acreage (left) and irrigated commodity acreage (right) by region (NASS-USDA).

By the middle of the 1990s, over 7.5 million acres of corn were irrigated across the HPA region compared to just over 2 million acres in 1970. Today, irrigated corn acreage alone is greater than all other major commodities combined for the NHP and CHP regions (Figure 5). While some areas of the HPA have tried shifting from corn to less water-intensive crops in an attempt to conserve water (e.g., Colaizzi et al., 2009), extensively irrigating the crop with the greatest economic return is still widely in practice today. For total acreage, corn is the primary crop in the NHP, wheat is primary in the CHP, and cotton is primary in the SHP. This trend in dominant crop type follows the same gradient of regional water availability, where the most water intensive crop is dominant in the north and the least water intensive crop in the south, further demonstrating how water supply in the physical domain affects decision-making in the agricultural domain. Across the HPA, irrigated corn now accounts for over 50 percent of all irrigation; with approximately 70, 75, and 80 percent of the corn being irrigated in the NHP, CHP, and SHP, respectively.

4.4. Groundwater Pumping

Widespread irrigation is the largest contributor to groundwater decline across the HPA. Steady groundwater level declines across both the CHP and SHP are evidence that irrigation practices in these regions are unsustainable (Figure 6A). Since the late 1930s, saturated volumes of the CHP and SHP aquifers have been reduced by ~30 and ~50 percent, respectively. Our projections based on linear extrapolation of trends in saturated thickness from 1993-2012 (after Haacker et al., 2015) show that irrigable acreage availability (areas with >10-m saturated thickness) will fall below 50 percent of the total SHP and CHP area by the years 2025 and 2065, respectively (Figure 6B). However, irrigation on the NHP has had little impact on the overall decline of groundwater in the region as a whole. This suggests that water in the NHP can generally be treated as a renewable resource (Haacker et al., 2015; Scanlon et al., 2012), except for some portions of the region.

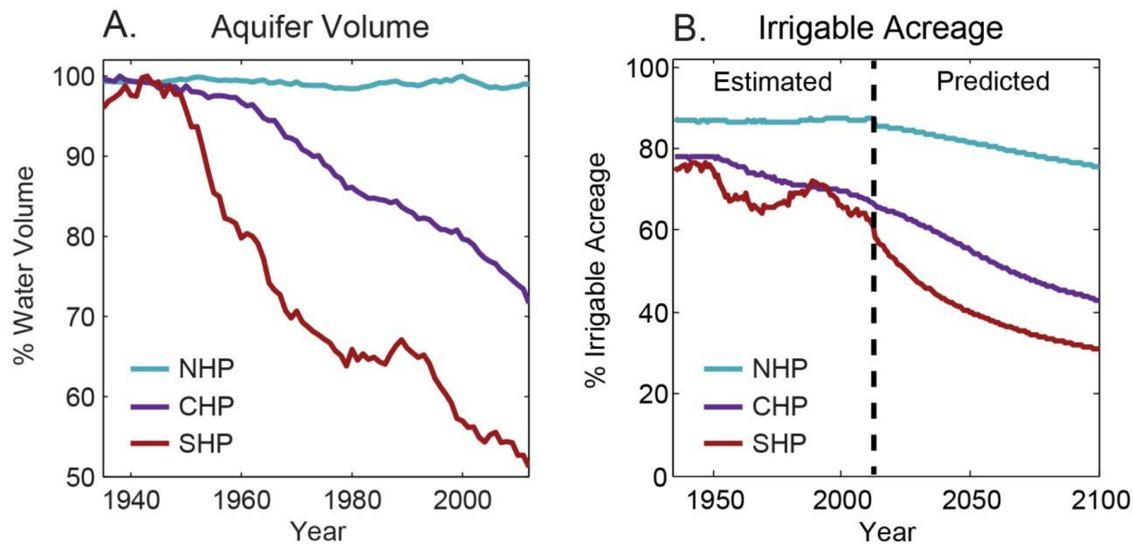


Figure 6. Aquifer decline across the High Plains. A) Saturated aquifer volume for each HPA region since predevelopment. B) Estimated (left) and predicted (right) irrigable acreage based on saturated thickness interpolations for each region (modified from Haacker et al., 2015).

Saturated thicknesses across the NHP have historically varied nonlinearly in a given location, suggesting that overall irrigable acreage may remain relatively stable into the future. However, saturated thicknesses across the CHP and SHP have not evidenced recovery, thus declining saturated thickness estimates are representative of declining irrigable acreage predictions for these regions. Extending the time frame for trend analysis prior to 1993 would allow for more comprehensive predictions of each region, but this dilutes the role of recent agricultural practices on declining groundwater levels. The average projected usable lifespan of the aquifer based on estimated 2007 storage and depletion rates is around 81-yrs for the SHP and 238-yrs for the CHP, while the NHP is relatively sustainable under current irrigation trends (Scanlon et al., 2012).

4.5. Efficient Water Use

Irrigation has become more expensive due to groundwater declines and the increased costs for the energy sources needed to lift groundwater, further supporting the central role of the water-energy-food nexus in modern agriculture. This increase in cost, in addition to the goal of conserving water resources, has led to the development and adoption of increasingly efficient irrigation technologies (i.e., reduction in the percent of water lost to direct evaporation per amount applied). In theory, improved efficiency of water use increases farmer profit by lowering production costs.

Since the 1980's, a common strategy to improve irrigation efficiency has been to modify pre-existing central pivot systems with lower-pressure spray applicators (Colaizzi et al., 2004; Colaizzi et al., 2009; Lyle and Bordovsky, 1983). Low-pressure spray applicators are classified according to the height of the nozzle, as Low-Elevation Spray Applicators (LESA) or Mid-Elevation Spray Applicators (MESA). Systems using an applicator sock dragged along the soil or a sprayer near the soil are referred to as Low Energy Precision Applicators (LEPA), which is also the common name for this entire low pressure applicator class.

We quantified the change in irrigation technologies across Kansas since 1990 (Figure 7) using water rights data from the Kansas Water Information Management and Analysis System (WIMAS). Prior to 1990, adoption of LEPA and related technologies was small, remaining below 5%. While the prevalence of flood irrigation systems steadily declined, farmers were transitioning to traditional high-pressure center pivot systems until 1997 when an abrupt inflection in adoption of LEPA-type systems occurred, along with a steady decline in flood and high pressure center pivot systems. By 2010, LEPA-type systems accounted for almost 65% of all irrigation systems across the HPA region of Kansas. Irrigation technology selections in

Kansas demonstrate the widespread adoption of LEPA technology, trends which are mimicked across the rest of the HPA states.

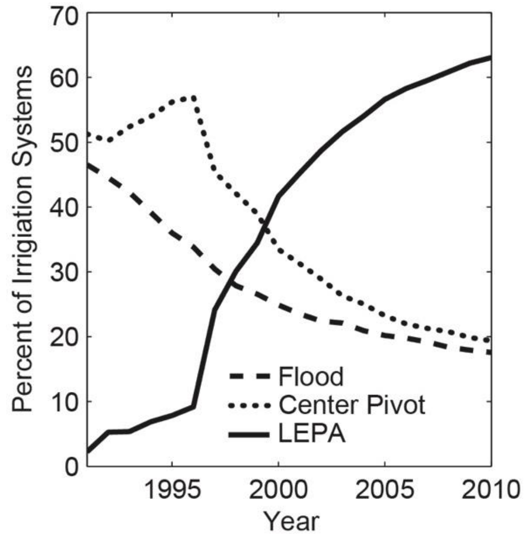


Figure 7. Irrigation technology selections across the HPA in Kansas (Kansas Division of Water Resources).

4.6. Water Use Response to Efficient Technologies

Irrigation technology can have a large effect on water use efficiency (Deng et al., 2006). For example, subsurface drip irrigation can reduce irrigation water use by 35 to 55% (Lamm and Trooien, 2003). However, groundwater level declines have not been mitigated by the widespread conversion to more efficient irrigation technologies; instead, total withdrawals have increased. As improved irrigation efficiency decreases the usage cost for water applications, more acreage can be irrigated at a lower cost, resulting in increased profit margins for farmers and increased incentive to irrigate more acres (Pfeiffer and Lin, 2014; Upendram and Peterson, 2007).

To demonstrate that efficient irrigation technologies have led to increased water use across the HPA, we processed data for total irrigated acreage from 1990-1996, seven years prior to the widespread adoption of LEPA technology, and 1997-2003, seven years directly after

LEPA adoption (NASS-USDA). Total irrigated acreage across the HPA increased by ~11.38 million acres after widespread LEPA adoption; by subregion, the NHP, CHP, and SHP increased by 5.55, 3.63, and 2.22 million acres, respectively (Table 1). Also significant are the trends in irrigated crop choice that directly follow LEPA adoption. For example, NHP farmers focused on irrigating a variety of crops rather than isolating corn expansion, CHP farmers expanded water intensive crops despite regional water level decline, and SHP farmers primarily sought to improve yields on predominant crops like cotton while also capitalizing on the incentive to grow water-intensive corn in the relatively dry region. From 1996 through 2015, there has been an 11 percent increase in irrigated acres on the NHP and CHP; in contrast there has been a 25 percent decrease on the SHP, likely due to the decrease in available irrigable acreage as displayed in Figure 6.

Table 1. Change in irrigated acreage following the widespread adoption of LEPA technology (1990-1996 compared to 1997-2003)

Commodity	NHP		CHP		SHP	
	%	Acres	%	Acres	%	Acres
Corn	-3	-1,300,000	+52	+3,900,000	+43	+693,000
Soybean	+138	+6,240,000	+77	+710,000	-37	-165,000
Wheat	+31	+425,000	-17	-1,330,000	-2	-42,000
Sorghum	-51	-324,000	-34	-1,070,000	+10	+226,000
Alfalfa	+15	+509,000	+684	+1,350,000	+[a]	+260,000
Cotton	N/A	N/A	+256	+74,000	+12	+1,240,000
Total	+11	+5,550,000	+19	+3,630,000	+14	+2,200,000

[a]Irrigated acreage increased from no prior irrigated acreage.

4.7. Other Methods

Past studies have also highlighted how maximizing efficient water use includes more than just improved irrigation technology. For example, efficient water use also includes processes such as fertilizer regimes (Ogola et al., 2002), root zone uptake (Clothier and Green, 1994), pre-existing soil moisture (Panda et al., 2003), and irrigation frequency and intensity (Kang et al., 2002; Nair et al., 2013). Yields have been highest when irrigation applications were frequent with low intensity (Behera and Panda, 2009) and when fertilizer applications integrated with irrigation could offset the additional need for water to maximize yield. Water uptake by plant roots mostly occurs in the uppermost 45-cm of soil, thus irrigation applications that supply water beneath this depth generally add to nutrient and water leaching (Panda et al., 2003). Furthermore, increased irrigation applications, even with efficient technologies, lead directly to increased water loss due to increased evapotranspiration (Howell et al., 2004; Ogola et al., 2002). Improved irrigation regimes are a major focus area for water conservation, and further research is needed that integrates water use with the social drivers behind water management.

4.8. Water Productivity

Improved water use efficiency can both limit the total volume of water applied per area and reduce the total water demanded by the crop system. This movement has been widely linked with “crop per drop” research where the objective is to maximize crop yield for every drop of water applied (Brauman et al., 2013). To quantify the amount of crop returned per water amount of water applied, we conducted a novel synthesis of the benefit of irrigation on yields, irrigated water applications per commodity, and irrigation water use efficiency (Figure 8). The yield benefit of irrigation (Figure 8A), or the difference between irrigated and non-irrigated yields, was calculated for each commodity and averaged across the HPA using the data in Figures 4 and 5. To calculate water applications per commodity, three county-level time series were used: (1) annual irrigated yields per commodity (Figure 4), (2) annual irrigated acreages per commodity (Figure 5), and (3) water use per commodity, which was estimated every five years using Agricultural Census and USGS Water Use data (NASS-USDA, 2012; NWIS-USGS). USGS

Water Use data prior to 1985 are at the state level, so we first disaggregated these to county level by assuming that relative county-level water use remained the same from 1985 back to 1970. Second, we used state level data from the 2013 Agricultural Census on water applied per commodity and assumed that relative water applied per commodity remained the same within each state across the analysis years. Third, we multiplied commodity acreages in each county by relative water use to partition total water among commodities. Finally, we divided the commodity water use in each county by county acreages to get water use per commodity. To estimate the "crop-per-drop" of irrigation water across the HPA, and how it varies across commodities, we divided the irrigated yield benefit (Figure 8A) by the water applied (Figure 8B), yielding Irrigation Water Use Efficiency (Figure 8C), or the benefit of irrigation per unit of applied water, across commodities.

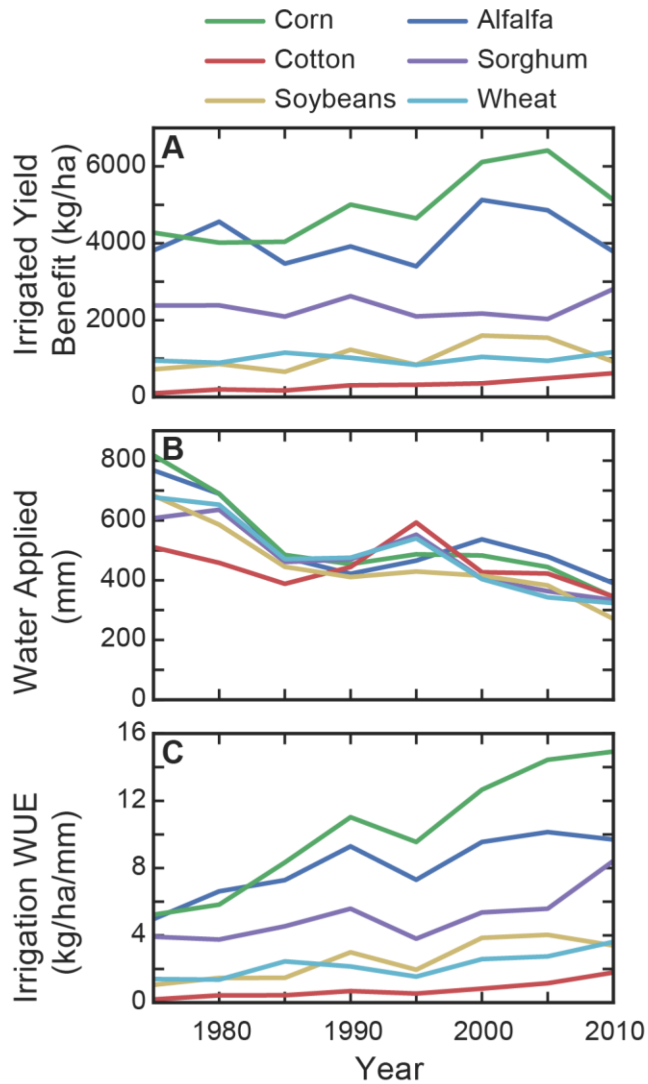


Figure 8. Crop yield, irrigation application data, and water use data (NASS-USDA; NWIS-USGS) were used to quantify crop yield per water spent in 5 year increments since 1970. A) Yield benefit is calculated as the difference in yield between irrigated and non-irrigated yield. B) The HPA-average annual amount of water applied for each commodity. C) Irrigation Water Use Efficiency (WUE), calculated as the irrigated yield benefit divided by applied water.

The incentive to irrigate is obvious based on the irrigated yield benefit. For example, irrigated corn yield is approximately 5,000 kg/ha greater than non-irrigated yield (Figure 8A). However, the productivity of water (i.e., crop yield per drop) has not been well documented across regions. Due to many factors including more efficient irrigation systems, shifts in cropping patterns regionally, and changes in irrigation application practices, the amount of water applied per season has decreased for all common commodities (Figure 8B), and the magnitude of crop yield gained per amount of water applied has steadily increased in recent decades across the HPA (Figure 8C), demonstrating that the productivity of irrigation water has steadily improved. For example, irrigation water use efficiency has nearly tripled in the last 45 years for corn and more than doubled for alfalfa, sorghum, soybeans, and wheat. This boost in regional productivity is directly linked to both the improvement in yield benefit (Figure 8A) as well as reduced water applications. Assuming these positive water productivity trends continue into the future, the incentive to irrigate will continue to increase, further intensifying resource demands in the water-energy-food nexus over the HPA.

4.9. Emerging Strategies

Emerging research to improve water productivity has largely focused on precision agriculture, crop choice, and cultivar improvement (Basso et al., 2011; 2013; Ritchie and Basso, 2008). In recent years, the emphasis on cultivar development has increased given the expected cross-domain implications of climate change such as decreased crop yields due to increased water stress (Basso et al., 2015; Basso and Ritchie, 2014). New crop cultivars may result in increased yields despite growth challenges posed by climate change by allowing for some traditionally water-intensive crops to be grown in regions where water is scarce (Hu and Xiong, 2014; Lobell et al., 2014). As more drought-resistant crop cultivars enter the market, growth of these cultivars in water-deficient areas will likely become more profitable (Benson et al., 2011).

Precision agriculture has generated significant interest among researchers and farmers given its potential to improve long-term production even at the small farm scale, although the adoption of precision agricultural practices has only grown moderately since their introduction in the 1990s (Daberkow and McBride, 2003; McBratney et al., 2005). Precision agriculture uses discretized, site-specific information based on factors including crop choice and soil type to develop strategies that are unique to that site, such as only applying irrigation to moisture deficient sections of a field (Basso et al., 2001; Bongiovanni and Lowenberg-Deboer, 2004). One challenge for large scale increases in water productivity using precision agriculture is that variable rate technologies are still under development and have not been widely applied in areas such as the HPA. The implications of precision agriculture on water productivity are likely most beneficial when considering adaptive, full-field irrigation strategies that respond to low soil moisture conditions.

4.10. Natural Viability

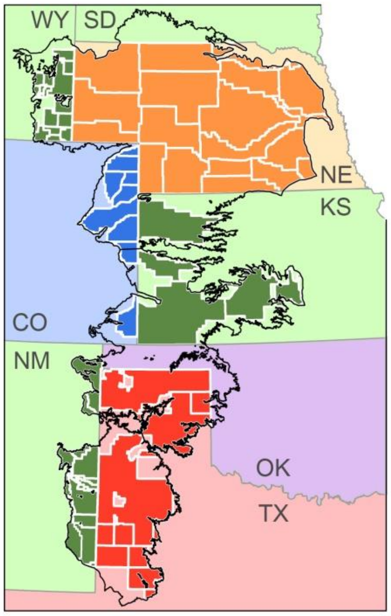
Crop type selection is a natural solution to water conservation. For example, switching from water-intensive corn to a less water-intensive crop mitigates the need for excess irrigation. In the northern Texas region of the SHP, switching half of irrigated corn to irrigated cotton could reduce water withdrawals by 8% (Colaizzi et al., 2009). Growing water-intensive crops in regions that need supplemental irrigation generates the largest demand for water withdrawal from the HPA aquifer. Crop selection based on the natural variability of the regional climate is the most effective method of water conservation. However, natural crop selection generally results in less farmer profit.

5. The Socioeconomic Domain

The socioeconomic domain both motivates and regulates how water is used within the water-energy-food nexus. In other words, this domain defines the incentives and social penalties for water use. Farmers generally aim to maximize profit, meaning the nature and location of economic incentives within the nexus can be useful indicators of potential water use. At the same time, legislation and political actions define to what extent, and sometimes at which locations, water can be used. Understanding how drivers within the socioeconomic domain may impact cross-domain trends in the physical and agricultural domains is a challenging but critical task in modern agriculture. We highlight historical socioeconomic and policy trends that provide key insights into areas where future management strategies can improve within the context of water conservation.

5.1. Historical Water Policy

In the United States, water allocation laws are made at the state level except where subject to federal rules such as interstate commerce (Peck 2007). Among U.S. states, there are four predominant doctrines governing water policy: (1) the absolute ownership doctrine: all water beneath a property owner's land belongs to the landowner, (2) the correlative rights doctrine: landowners must share underlying water with other owners of land over an aquifer, and each owner has equal rights to groundwater, (3) the reasonable use doctrine: the landowner can use underlying water without restriction as long as it is beneficial to the overlying land, and (4) the prior appropriation doctrine: priority belongs to the most senior claim, often phrased "first in time, first in right." The dominant legal doctrines governing water rights across the HPA states are displayed in Figure 9.



State	Doctrine	Control	Strategic Plan
CO	Modified Prior Appropriation[1]	State,[1] Local	Planned depletion[3]
KS	Prior Appropriation[1]	State,[1] Local	Sustainable yield,[2] planned depletion
NE	Reasonable Use/Correlative Rights[1]	Local[2]	Set by NRDs[2]
NM	Prior Appropriation[1]	State[1]	Planned depletion[4]
OK	Reasonable Use[1]	State[1]	Planned depletion[4]
SD	Prior Appropriation[1]	State,[1] Local	Set by state and[4] Indian Reservations
TX	Absolute Ownership[1]	Local[1]	Set by monitoring[2] program
WY	Prior Appropriation[1]	State[1]	Long-term monitoring[5]

[1]McGuire et al., 2003; [2]Peck, 2007; [3]White and Kromm, 1996; [4]Gutentag et al., 1984; [5]Taboga, 2015

Prior Appropriation
Reasonable Use
Modified Prior Appropriation
Absolute Ownership
Correlative Rights/Reasonable Use
Darker shades represent local management boundaries

Figure 9. Dominant groundwater doctrines, local management boundaries, primary control levels, and prevailing management plans across the HPA in each state.

Most HPA states have developed localized management areas to enact further protection for groundwater after decades of following a state-first control model (Fipps, 1998; Peck, 2007). HPA states have intensified groundwater management strategies by implementing plan requirements, regulating case-specific problems, and establishing critical watershed areas in efforts to address groundwater decline issues not met by pre-existing allocation policies (Ashley and Smith, 1999; Kaiser and Skillern, 2001; Mace et al., 2006). Despite a more localized and defined management approach, allocation policies have failed to adequately protect against groundwater depletion (Kaiser and Skillern, 2001). The control levels of all HPA states are further summarized in Figure 9.

Surface water connections have also become increasingly prominent in modern groundwater policy and legislation. For example, in 1999, Kansas sued Nebraska and Colorado alleging that reduced flow in the Republican River due to large-scale groundwater development violated the Republican River Compact of 1942. Although groundwater was not explicitly addressed in the Compact, the US Supreme Court ruled that groundwater use was restricted if it depleted transboundary streamflow. The resulting restrictions for this region of Nebraska included a suspension on drilling new water wells, mandatory metering of irrigation wells in the watershed and certifying irrigation acreages, restrictions of groundwater pumping volumes, and a framework to use groundwater modeling to assess compliance on five-year running averages (Kuwayama and Brozović, 2013; Peck, 2007). Future water management can capitalize on improved observations and numerical models to formulate strategies that integrate surface water and groundwater as a preemptive step to groundwater conservation.

5.2. Motivation for Policy Changes

Historically, water policies in the High Plains states were created during periods of limited demand on water resources. These initial policies still exist as political frameworks, and policies have been fit within the structure of these outdated philosophies. Most HPA states acknowledge that under current policy, it is more realistic to manage groundwater as a nonrenewable resource or mined commodity, rather than a sustainable and renewable resource (Waskom et al., 2006). As a result, many areas on the High Plains have implemented "manage for depletion" regimes where calculated water withdrawals are permitted based on an extraction formula, rather than targeting aquifer sustainability (McGuire et al., 2003; Peck, 2007; Waskom et al., 2006). Management strategies across the HPA region are summarized in Figure 9.

HPA states have attempted to modify federal, state, or local governance models to fit within the limiting frameworks of historical policy and mitigate groundwater decline, but the limitations of these adjustments are frequently debated as groundwater depletion has continued under both large and small-scale control (Haacker et al., 2015; Kromm and White, 1987; Peck, 2003; Scanlon et al., 2012). A challenge is that large-scale control often overlooks localized needs, but local management bodies can be reluctant to self-impose overwithdrawal sanctions (Peterson, 1991). For example, the absolute ownership doctrine in Texas grants the landowner flexibility in water withdrawals, but little protection exists for neighbors against overwithdrawals. Localized permit systems discourage overwithdrawals, providing greater state and local control, but little flexibility is granted to the landowners and extensive government resources are necessary to administer the complex system of water rights and allocation. Given these challenges, traditional management strategies emerging from past policies are unlikely to meet the water demands of the future (NRCS, 2001; 2004).

5.3. Farmer Profit

Water use across the HPA region is intimately linked to short-term farmer profit. This concept is demonstrated by irrigating corn, the commodity most likely to return the greatest profit, in the water-stressed SHP region, the HPA region least suited for the crop, despite the understood implications of groundwater decline. This suggests that future management strategies focused on water conservation should also take farmer profit into consideration through various economic policies; these policies can be broadly sorted into: (1) ***direct policies***, where direct restrictions are imposed on human behavior (e.g., restrictive water use legislation), and (2) ***indirect policies***, where economic incentives are used to encourage a change in behavior (e.g., subsidies for water-conserving practices). An ideal economic policy should be designed to simultaneously encourage farmer profit protection and water conservation, all while staying within the pre-existing frameworks of direct policies.

Farmer profit is a function of global market demand, production costs, and the variability or risk involved in crop growth. From the perspective of a farmer, risk and variability linked to decreased yields are often the biggest concern for decreased revenues (Barry, 1984). In general, agricultural risk can be divided into: (1) production risk (associated with yield, input costs, and weather variability), (2) market risk (uncertainty about future market value of the harvest), and (3) institutional risk (the potential for change in agricultural policies; Babcock and Shogren, 1995; Barret; 1996; Eakin, 2005). By reducing risk and variability through indirect policies, expected revenue and production costs can be balanced to provide a substantial influence on crop choice.

Crop insurance provides one method to mitigate production risk (Hazell et al., 1986), but the long-term success of this strategy is often questioned (Duncan and Myers, 2000; Miranda et al., 1997). Few other risk mitigation methods exist despite the critical link between risk management and best management practices. For the High Plains, many of the active indirect policies and risk management strategies are defined in the U.S. Farm Bill, a comprehensive agricultural bill passed by congress every five years.

The U.S. Farm Bill includes market supports that boost the value of particular commodities, subsidies that provide incentives for best management practices (e.g., switching to high-efficiency irrigation systems), and crop insurance that decreases the risk of profit loss during a variable growing season. For example, the 2014 US Farm Bill includes the Stacked Income Protection Plan (STAX), which allows enrolled cotton farmers to receive payments if regional yields fall below 90% of the expected level, ultimately decreasing the risk for growing cotton. Another example is the Conservation Reserve Program (CRP), which was first introduced in the 1985 Farm Bill and has significantly affected the HPA region by encouraging the retirement of marginal farmland through rental and cost-share payments to farmers (Osborn, 1993). However, despite the long history of the U.S. Farm Bill, only a few studies (e.g., Rao and Yang, 2010) have examined how indirect policies have influenced water availability.

Most indirect policies have done little to protect HPA groundwater, given that the incentive to increase profits is antithetical to water conservation. In fact, current indirect policies may increase the demand for water use across the HPA. For example, the Renewable Fuel Standard (RFS) of 2005 required that 7.5 billion gallons of renewable fuel be blended into gasoline by 2012 (Schnepf and Yacobucci, 2013). This biofuel mandate generated a profitability incentive to farmers, ultimately increasing the planting of water-intensive biofuel crops (e.g., corn). This increased water burden may or may not be reduced in the future as less water-intensive biofuel crops (e.g., sorghum) become more profitable. Indirect policies concentrated on water conservation will be more realistic if factors such as irrigable acreage and total water use are considered (Caswell and Zilberman, 1985). Interdisciplinary research that integrates social and natural sciences will be necessary to help develop successful future water management strategies that incorporate indirect policies and still mitigate groundwater decline.

5.4. Market Prices

Effective groundwater management strategies must capture spatially and temporally dynamic drivers, making it difficult for uniform policies to be effective. Market prices, for example, have strongly fluctuated over the last fifty years (Figure 10). Commodity prices during the 1970s were much higher than the 1990s, but values increased in the early 2000s to those similar to the early 1980s. More recently, record grain production in 2012 and 2013, coupled with unusually high grain prices in 2012, generated substantial bumper crops and subsequently a sharp decline in grain prices prior to the 2014 season, demonstrating that short-term factors can compromise management strategies even at the seasonal scale (USDA, 2014). Value fluctuations have direct implications on irrigation demand through revenue incentives, particularly when water intensive crops have a high market value. These dynamic complexities in management strategies remain challenging to capture for long timescales; this challenge is intensified by the unknowns linked in other domains such as climate variance and irrigation technologies.

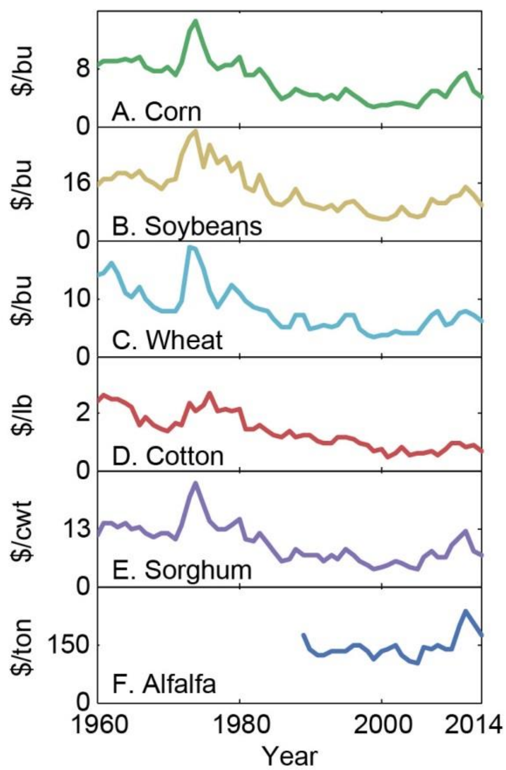


Figure 10. 2014 price adjusted market values for common HPA commodities. Commodity prices are synthesized for HPA states, with the exception of cotton which did not have official state data for the HPA region (NASS-USDA). Cotton prices are derived from national market values. Official alfalfa prices are not available prior to 1989 for the HPA states.

5.5. Irrigation Value

One challenge to cross-commodity analyses is finding an equal metric for comparison. For example, comparing the irrigation water use efficiency for corn and cotton (Figure 8A) would suggest that it is much more efficient to grow corn rather than cotton. But without an economic value for efficiency, it is not an even comparison (i.e., a kilogram of cotton is not equal to a kilogram of corn). To allow for cross-commodity comparisons, we calculated the value of irrigation by multiplying irrigation water use efficiency in kg/ha/mm (Figure 8C) by market value for each commodity converted to $/kg (Figure 10), linearly interpolating the irrigation water use efficiency data annually. The result is a time-series of annual irrigation per commodity (Figure 11). It is no surprise that the irrigation value is high for corn given the large irrigated yield benefit (Figure 8A) and high water use efficiency associated with the crop (Figure 8C), but the irrigation value of cotton is also high despite the relatively low irrigated yield benefit (Figure 8A) and low market value (Figure 10) on a per-mass basis. Thus, quantifying the economic value for irrigation can offer key insights that highlight incentives within the water-energy-food nexus.

Our results indicate that given high irrigation values for both corn and cotton, restructuring a management plan or subsidy program around the production of irrigated cotton instead of water-intensive corn may provide an economic opportunity for farmers in regions like the SHP to switch from corn to the less water-intensive commodity. Another example is the irrigation value of wheat, which has yield benefits and water use metrics similar to those of cotton, but its irrigation value is substantially less (Figure 11). This suggests that economic incentives aligned with the production of irrigated wheat may not be very beneficial to either farmer revenues or water conservation. By understanding the value of drivers like irrigation value, management plans can be designed to promote both farmer profit and the mitigation of groundwater loss by anticipating the most economical decisions for farmers.

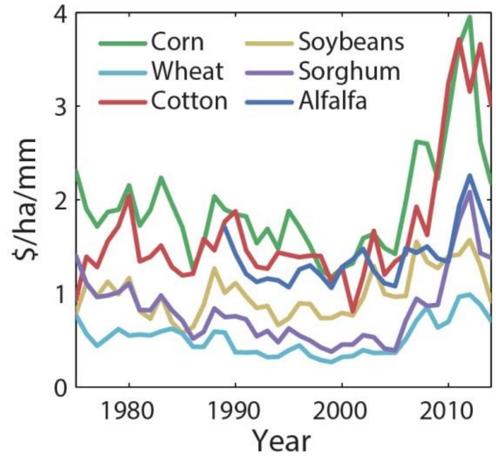


Figure 11. The value of irrigation per commodity across the HPA. Irrigation value is calculated as water use efficiency for each commodity (Figure 7C) multiplied by its corresponding market value in $/kg (Figure 9).

5.6. Adaptive Management and Innovative Strategies

The High Plains would also benefit from adaptive management that is responsive to short-term drivers (e.g., government subsidies, drought) within a long-term framework. Recently, sustainable management approaches have defined long-term goals for desired conditions 50-100 years into the future and used backcasting to inform short-term objectives and water use limits (Gleeson et al., 2012). By adapting short-term regulations to meet long-term goals, water management can be tailored to regional challenges and newly implemented programs. This allows for spatially and temporally relevant adjustments that adapt to the regional needs across the HPA while still maintaining groundwater sustainability as an objective. However, implementation of these strategies is too recent to evaluate the effectiveness of this approach.

Innovative strategies have also been integrated into current policies, but the benefits of these trials have been mixed. Current attempts have included: (1) heterogeneous tax policies, where water during dry years is seen as more valuable than during wet years, thus subsidies are given in exchange for groundwater conservation (e.g., Ashwell and Peterson, 2013), (2) restrictions on new drilling and pumping, and (3) voluntary restrictions to total water use that are self-imposed through personal or local initiatives (Mulligan et al., 2014). However, these strategies have also been shown to increase water use and streamflow depletion because they do not accurately capture changes in practice by users (Ashwell and Peterson, 2013; Scheierling et al., 2006; Ward and Pulido-Velazquez, 2008). Innovative strategies designed to mitigate water use must include the preferences of farmers if groundwater conservation is to be achieved.

When combined with innovative methods and regional markets, adaptive management strategies could significantly alter water use across the High Plains. For example, the Twin Platte Natural Resources District in Nebraska implemented the first groundwater permit trading market in the United States in 2014 to maintain streamflow in the Platte River (Young and Brozović, 2016). Because the marginal cost of water reductions varies across users, permit trading theoretically allows each unit of water pumped out of the system to be used at the lowest overall cost to the system (Brozović and Young, 2014). This contrasts with uniform quotas on groundwater pumping across users, which can force some users to make costly reductions while other low cost solutions are overlooked. Permit markets have the potential to be cost-effective while maximizing flexibility for water users (Palazzo and Brozović, 2014). A longer implementation period is needed for full evaluation, but permit trading highlights a cost-effective groundwater management strategy that promotes farmer profit, includes farmer values, and could be implemented in other regions of the HPA.

6. Discussion and Conclusions

Agricultural water use is depleting the High Plains Aquifer, yet current water management strategies will not prevent future declines. Increased climate variability will likely increase the stress on water resources across the High Plains, specifically through changes in precipitation patterns and drought intensification. We found that irrigation tends to at least double commodity yield when compared to their non-irrigated counterparts, placing a large economic incentive on irrigated water use across the semi-arid High Plains region. Additionally, we found that efficient irrigation technologies can reduce total groundwater conservation, as irrigated acreages substantially increased after the widespread introduction of efficient irrigation technologies and groundwater level declines continued at rates similar to those prior to the efficient systems. Future decades will require significant changes in agricultural practices for the SHP and CHP regions, as irrigable areas are predicted to decline ~30 to 50 percent by 2100 relative to current irrigable areas. We further quantified irrigation water use efficiency and found that the amount of crop per unit irrigation, often called crop per drop, has increased through time for every major commodity. We multiplied these crop per drop values by market prices to quantify the unit value of irrigation water for each commodity. Based on our results, cotton and corn have the highest irrigation value, followed by alfalfa, soybeans, sorghum, and lastly wheat. These new datasets provide a basis to evaluate the influence of major water use drivers across domains and develop key insights into the water-energy-food nexus for modern agriculture.

Based on the trends analyzed in this study, our main conclusion is that future water management strategies would benefit most from: (1) prioritizing farmer profit as an incentive for change in practice, (2) managing water as an input in the water-energy-food nexus, (3) focusing on adaptive frameworks, (4) adopting innovative strategies that function within current policies, (5) reducing production risk, and (6) increasing political desire for resource sustainability.

Short-term farmer profit is the primary driver to water use across the High Plains. As long as there is an economic incentive to irrigate, farmers across the HPA have largely demonstrated that extensive groundwater extraction will continue regardless of the potential risk for resource collapse. While aquifer depletion may be inevitable in some locations, water conservation provides an optimal economic path, giving the region's economy time to diversify and maximize both crop per drop and profit per drop. Introducing restrictive caps and regulations can reduce groundwater use, but these efforts also result in decreased crop yields which pose direct threats of food, fiber, and fuel shortages, as well as local economic hardship. Instead, future strategies should attempt to shift the economic incentive away from immediate groundwater extraction by placing incentives in the growth of less water-intensive crops as a way to encourage sustainable management. This requires development of alternative biofuels, increased demand for the commodities that generate alternative biofuels, and the implementation of government programs or market adjustments to make these alternative crops valuable. Farming practices will follow economic incentives, thus management strategies should incentivize farmers to profitably reduce water use rather than making it more difficult to maintain livelihoods through water use restrictions.

Water is the limiting component to agricultural production within the water-energy-food nexus, and past practices on the HPA demonstrate that overlooked water use incentives will be exploited if not properly accounted for in management strategies. Groundwater sustainability goals can only be met when water use is balanced as an input within the nexus, where food and fuel are functions of water use. In budget terms, groundwater sustainability goals can only be met when annual groundwater use is nearly equal to the annual recharge supplied. Given that the agriculture industry across large portions of the HPA has historically been established using unsustainable practices, future management strategies must compound multiple water conservation methods to offset the extensive reliance on groundwater pumping.

Adaptive frameworks capture temporally dynamic water use drivers (e.g., extended droughts, new government incentives, and market price fluctuations) by granting decision-making freedom in response to changing circumstances. Thus, adaptive management strategies must incorporate short-term objectives that align with long-term goals to remain relevant at extended timescales, and must react to changing physical and social drivers that caused past strategies to become outdated. By allowing for heterogeneous, short-term flexibility in a long-term framework, strategies can be tailored to dynamic drivers even at the seasonal timescale to meet long-term goals.

Widespread groundwater decline is enabled by pre-existing political frameworks that govern water law. These frameworks are outdated and often irrelevant on the High Plains, as evidenced by the shift from regional- to local-scale groundwater management over many areas of the HPA. Management strategies that follow this traditional political framework will also likely fail. Traditional frameworks are too restrictive to capture every critical water use driver, allowing farmers to exploit these overlooked areas and capitalize on the economic incentives they leave unregulated. Instead, innovative and nontraditional strategies should be designed to fit within existing legislation, but designed to capitalize on the decision-making behaviors that follow economic incentives. Innovative strategies do not need to capture every driver; rather, they need to manage for the decisions that follow the key driver: farmer profit. New strategies that align farmer profit with reduced water use may prove more effective within the legislative framework.

Reduced production risk is another way to encourage farmer behavior by placing an economic incentive toward ensured revenue. If the risk associated with a change in practice (e.g., less irrigation) is reduced, then farmers will be more likely to adopt new practices that align with water conservation objectives. Reduced risk can come through mechanisms including the enhanced development of climate resistant cultivars or more effective insurance programs.

There must be the political will to promote groundwater conservation. Past strategies can mitigate groundwater declines to a certain extent, but they cannot fully succeed if there is not the will to implement them. Given that some portions of the HPA are already managing for depletion, there appears to be a conflict between these areas and a regional desire for groundwater sustainability. Management strategies must be constructed with the necessary tools to succeed, but they must also be implemented in a political framework that promotes and advocates for successful implementation.

Acknowledgments

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CHAPTER 3

Agricultural and Economic Implications of Providing Soil-Based Constraints on Urban Expansion

Abstract

Urbanization onto adjacent farmlands directly reduces the agricultural area available to meet the resource needs of a growing society. Soil conservation is a common objective in urban planning, but little focus has been placed on targeting soil value as a metric for conservation. This study assigns commodity and water storage values to the agricultural soils across all of the watersheds in Michigan's Lower Peninsula to evaluate how cities might respond to a soil conservation-based urbanization strategy. Two Land Transformation Model (LTM) simulations, one representing traditional urbanization and one representing soil conservation-based urbanization, are used to create urban area maps from 2010-2050 at five year intervals. The expansion of urban areas onto adjacent farmland is then evaluated to quantify the conservation impacts of soil-based development. Results indicate that a soil-based strategy significantly conserves total farmland, while also conserving the more fertile soils within each soil type. In terms of revenue, ~$88 million was conserved in 2050 when using soil-based constraints, with the projected savings for 2011-2050 totaling more than $1.5 billion. Soil-based urbanization also increased urban density for each major metropolitan area. For example, there were 94,640 more acres directly adjacent to urban land by 2050 under traditional development compare to the soil-based urbanization strategy, indicating that urban sprawl was more tightly contained when using soil value as a metric for development. Based on this study, implementing a soil-based urbanization strategy would better satisfy future agricultural resource needs than a traditional approach to urbanization.

Keywords: Urbanization, Land Transformation Model, soil conservation, land use change, urban planning, agriculture, soil value

1. Introduction

As population and subsequent urbanization increases, the amount of land available for crop production decreases (Tayyebi et al., 2015; 2016a). This discordant relationship is increasingly essential as crop demand is expected to roughly double by 2050 (Tilman et al., 2011), while total urban area may increase almost 300 million acres by 2030 based on current urbanization trends (Seto et al., 2012). Furthermore, future climate change is expected to negatively impact the availability, stability, utilization, and access of global agricultural production, making it increasingly difficult to meet this emergent crop demand (Schmidhuber and Tubiello, 2007). Food security is a major global concern (Godfray et al., 2010), yet a critical knowledge gap exists between current urbanization patterns and the increased demand for soil-based resources (Setälä et al., 2014).

Urbanization onto adjacent farmlands largely follows economic drivers where farmland is sold to developers based on profit incentives for both the farmer and developer (Satterthwaite et al., 2010; Devadoss and Manchu, 2007). Two major problems exist with these types of transactions: 1) land values are often based on the *current* market value of production and risk rather than a market value based on *future* resource demands (Goodwin et al., 2003), and 2) land values are driven by location (e.g., proximity to urban areas) and not explicitly the relative quality of soil across possible development sites (Livanis et al., 2006; Huang et al., 2006); both discount the long-term resource needs of a growing society. Thus, it is common for the most fertile soils in a region to be the first converted to urban land as cities expand onto adjacent farmland. This is a particularly important oversight because once farmland is converted to urban land, it can no longer function as a food and energy depot for the public (Dunlap and Jorgenson, 2012; Thompson and Prokopy, 2008), leaving behind the soils with average or below average yield potentials to grow the crops necessary for a growing population. Yet, the value of fertile soil is still not adequately included in long-term urban planning (Blanco-Canqui and Lal, 2008).

Every state has policies in place to preserve farmland (e.g., tax relief, right-to-farm laws, development rights, or zoning restrictions; Nelson, 1992), but these policies generally do not differentiate soil types based on yield potential. This generates a fundamental disconnect between current development practices and future resource needs. One reason why soil-based development is not widely implemented is that urbanization is considered to only impact a small fraction of total agricultural land across a region (Thomspon and Prokopy, 2008; Chen, 2007; Hart, 2001). However, at smaller spatial scales, urbanization can have a substantial impact on the total area of key commodities and the total revenue of local economies. Furthermore, lost production and revenue is compounded annually as maximum annual crop yields can no longer be achieved due to the development of the most fertile soils.

Here, we simulate the agricultural and economic impacts of urbanization when a soil-based development constraint is implemented across Michigan's Lower Peninsula (LP) region. This study uses the Land Transformation Model (LTM) to simulate two development scenarios: 1) non-penalized urban expansion (i.e., traditional development), and 2) penalized urban expansion using soil value as a way to develop lower-valued soils first (i.e., soil-based development). Results from this study provide an initial framework for soil-based urbanization planning, which could be used to inform land development policies and resource conservation strategies.

2. Methods

2.1. Study Area

The study region includes Michigan's Lower Peninsula (LP) and adjoining areas that drain to the Great Lakes, including parts of Illinois, Indiana, and Ohio (Figure 12). This region has a total population of more than 15 million, most of which is in major metropolitan areas including Chicago, Detroit, Toledo, Lansing, Grand Rapids, Kalamazoo, South Bend, and Traverse City (US Census, 2010). Agricultural land covers 36% of the study area; 98% of which is used to grow 11 main commodities: corn, soybeans, wheat, hay, cherries, apples, blueberries, potatoes, cucumbers, dry beans, and sugar beets (Homer et al., 2015; CDL, 2014). These 11 commodities were selected as the most valuable for the region given their largest total dollar production; in 2014, corn production accounted for $2.87 billion, $1.94 billion from soybeans, $893 million from dry beans, $691 million from hay, $415 million from wheat, $169 million from cucumbers, $159 million from apples, $132 million from potatoes, $114 million from cherries, $73 million from sugar beets, and $58 million from blueberries. Collectively across all commodities, agriculture in this region accounts for more than $9 billion per year (NASS-USDA).

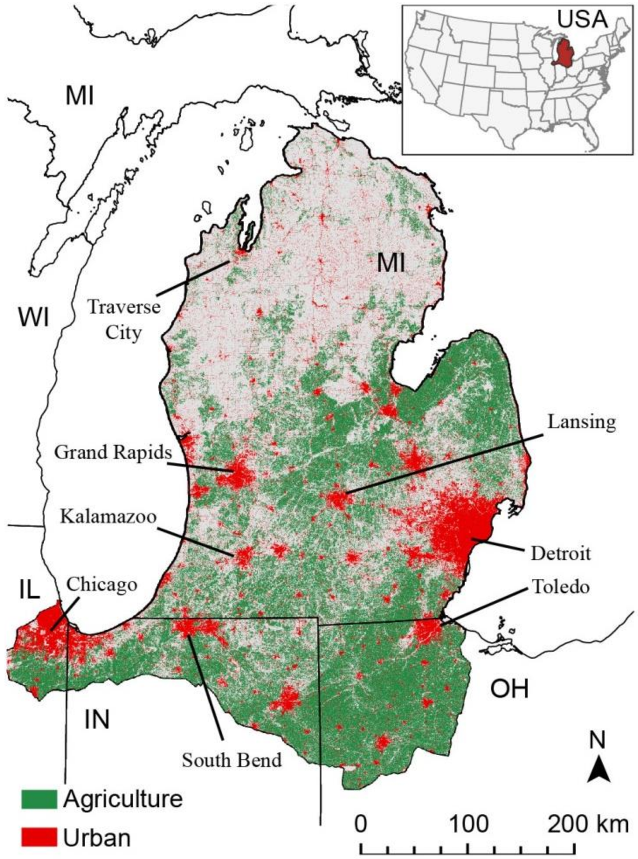

Figure 12. Site map of the Lower Peninsula watershed region (Homer et al., 2015).

The southern half of the region contains the majority of the population and most of the agricultural land. Crops in this southern portion primarily consist of the major row crops (corn, soybeans, wheat, and hay), while the western and eastern portions respectively grow mostly specialty fruits (cherries, blueberries, apples, and grapes) and other commodities (dry beans, cucumbers, potatoes, and sugar beets). The southern half of the boundary is widely characterized by flat lands with highly fertile, silty soils, where the northern half of the boundary includes more distinct glacial topography and sandy soils. All of the major metropolitan regions are situated within agricultural land, except for Traverse City, which is located in the northern LP adjacent to cherry farms and vineyards; almost all urbanization must expand into nearby farmlands. Adjacent land other than agriculture (36% of the study area) primarily consists of other urban land (14%), forest (25%) or grassland (12%; Homer et al., 2015).

2.2. Simulation Period and Early Trends

We simulate urbanization from 2011-2050 in 5-yr intervals, building first from development trends between 1992 and 2001. During this 1992-2001 period, population in the study area increased nearly 7% (US Census, 1990, 2000), while total urban area increased over 50% (Vogelman et al., 2001, Homer et al., 2007). Heavy development occurred primarily in suburban areas, where lateral expansion greatly extended away from city centers. The largest increases in relative urban land occurred in the Grand Rapids, Kalamazoo, and Traverse City metropolitan areas, though each major metropolitan region increased in urban area. In 2011, average distance of urban area to the centroid of urban mass was 27.6 km for Detroit (24% increase since 1992), 22.3 km for Chicago (10% increase), 14.1 km for South Bend (7% increase), 13.4 km for Grand Rapids (49% increase), 10.8 km for Toledo (24% increase), 8.4 km for Lansing (20% increase), 7.1 km for Kalamazoo (28% increase), 7.1 km for Saginaw (24% increase), and 6.7 km for Traverse City (52% increase). By 2011, urban area in Chicago increased by 84% when compared to 1992, 95% for Detroit, 204% for Grand Rapids, 155% for Kalamazoo, 119% for Lansing, 89% for Saginaw, 217% for South Bend, 118% for Toledo, and 373% for Traverse City. This study uses the Land Transformation Model to build on these trends as a way to forecast development into the future.

2.3. Land Transformation Model (LTM)

The Land Transformation Model (LTM) forecasts spatially-explicit land use transitions through time using two primary input datasets: 1) GIS maps of potential spatial drivers, and 2) spatially- and temporally-variable transition demand (i.e., population growth estimates for urbanization). The LTM weights spatial drivers into temporally-variable suitability maps, and then assigns transitions according to transition demand. Here we briefly describe the functioning of LTM, more details can be found in Pijanowski et al., 2002; 2010; 2014; and Tayyebi et al., 2015.

LTM creates suitability maps by weighting spatial drivers of transitions using an artificial neural network (ANN), which is trained using two land use maps. The ANN adjusts the weights of the drivers, and assigns temporal transitions to the highest weighted cells, until the spatial patterns of transitions most resemble those encountered in the training datasets (here, transitions from 1992 to 2001). In this analysis, National Land Cover Dataset (NLCD) land use data for the years 1992 and 2001 were used to train the LTM to predict urbanization. The LTM allows for a spatially and temporally dynamic neural network that reacts to a changing landscape of drivers, where weights vary in time and space and are updated every at every 5-yr interval (Pijanowski et al., 2000; 2002).

This study incorporated eight variables to build the ANN (Figure 13) distance to: A) urban areas (Vogelman et al., 2001; Homer et al., 2007; 2015), B) major highways (TIGER/line 1998; 2013), C) secondary highways (TIGER/line 1998; 2013), D) major rivers (NOHRSC, 1999), E) all inland lakes (NWIS-USFWS), F) the coast, G) population density (US Census, 1990; 2000; 2010), and H) soil value. The baseline urbanization simulation used variables A-G, while the soil-based development simulation added variable H. Each distance raster was calculated using Euclidean distance in ArcGIS. All layers have rasters with 25,000 rows and 25,000 columns with rectangular cellsizes. Optimum model parameters were estimated by iterating the LTM model through training cycles, where a training run continues until two successive mean squared error differences reach a specified stable point (Tayyebi and Pijanowski, 2014; Pijanowski et al., 2006; 2009).

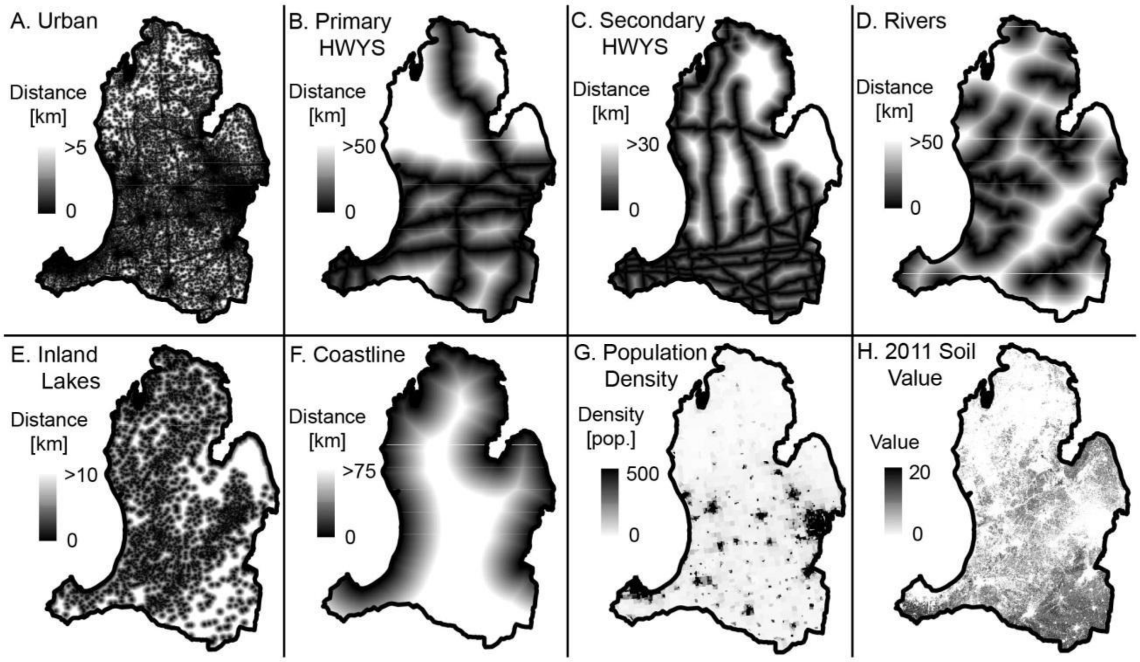

Figure 13. Primary variables used in the ANN. Panels A-G were selected as the most influential drivers to development across the region, and Panel H was integrated as the soil-based development driver.

The estimated parameters were then applied to the independent predictor variables in 1992 (Panel A-G, Figure 13) to predict urbanization suitability and urban area in 2001. The predicted transformation map in 2001 is combined with NLCD in 2001 to create a composite land use map that is compared to the NLCD in 2001 to evaluate the predictive ability of the LTM. The total number of urban transition cells to be added was quantified from the training sets, and the most suitable cells were assigned to urban class until the urban area was met. Urbanization within the LTM is only allowed in a forward-moving sense, meaning once a cell becomes urban it does not change back to a different land type. The LTM also works horizontally in space and not vertically, indicating that the upward development of dense city areas is not simulated.

The performance of the LTM for modeling urbanization was evaluated using Percent Correct Match statistics (PCM) and Relative Operating Characteristic curves (ROC). PCM describes the proportion of the reference map where urban change, and no-urban, have been correctly predicted by the LTM (Pijanowski et al., 2002), and ROC calculates accuracy by plotting the relationship between false positives vs. true positives. Both PCM and ROC returned over 80% accuracy when comparing predicted 2001 NLCD to actual 2001 NLCD, which were satisfactory for the remaining urbanization predictions. We reserved 2011 as an additional training set if 1992-2001 proved unsatisfactory, but it was not included given the results of the agreement. We then used LTM trained between 1992 and 2001 to predict urbanization from 2011-2050 in 5-yr intervals for both the baseline urbanization and the soil-based development simulations. Results from each simulation were then used to quantify the subsequent urbanization of adjacent agricultural land.

2.4. Soil Value Penalty Layer

Soil value was quantified using two components: 1) a commodity value, and 2) a water storage value. Each cell has a commodity value and water storage value between 1 and 10. The final summed soil value layer thus has values of 2-20, where 2 is the least valuable and 20 is the most valuable soil (Figure 13, Panel H).

The commodity value was determined by the public as reflected in the market prices ($/kg) for each commodity multiplied by the average production per area (kg/acre) to get commodity values in $/acre. This value was calculated for each commodity using the 2011 state-level yields and prices paid per harvested weight downloaded from the National Agricultural Statistics Survey (NASS-USDA) data. Each commodity was ranked from 1 to 10, where 1 was the least valuable and 10 was the most valuable. Production costs were not included in the ranking (Table 2), since spatially complete data are not available at the state or county level. We assumed that farmers were using best practices and that the commodity planted would have the highest return on revenue for the underlying soil type. Commodity rankings were then applied to each corresponding commodity cell within the study domain using the 2011 Cropland Data Layer, which is a map of the location of crops grown.

Table 2. Commodity rankings based on market value and production weight per area ($/acre)

Commodity	**$/acre**	**2011 Value**
Soybeans	344	1
Wheat	372	2
Hay	395	3
Dry Beans	494	4
Corn	540	5
Sugar Beets	1219	6
Potatoes	3221	7
Cherries	3372	8
Apples	5526	9
Blueberries	8413	10
Cucumber	3224	*not in CDL

Each water storage value was determined based the plant available water (PAW) characteristic of the underlying soil within each commodity. PAW is a measure of the potential soil moisture storage available for root uptake, as calculated in Equation 1. Within each commodity, we assumed that a higher PAW value would result in greater yields, as more water is available for the crop. Thus, calculating the water storage value includes two steps: 1) calculate PAW everywhere, 2) determine the distribution of PAW values spatially within a given commodity, and 3) break each distribution into decile bins and map these back across the domain.

$$\text{PAW} = \text{field capacity} - \text{wilting point} \tag{1}$$

Field capacity and wilting point were calculated using information from the SSURGO (Soil Survey Geographic Database, 2016) dataset, along with soil hydraulic properties from the ROSETTA database (Schaap et al., 2001). For each SSURGO polygon (map unit, or MUKEY) there are potentially several types of soils (components, or COKEY), which can each have multiple horizons (CHKEY). If present in the database, percent sand, silt, and clay values for each component horizon were mapped to values of field capacity and wilting point extracted from the ROSETTA database. If percent textural values were missing, then textual descriptions (i.e. fine sand, or sand) were mapped to percentages using the centroids of the USDA soil textural triangle. These were then mapped to component horizon field capacity and wilting point values. Each component within a map unit can have different horizon layering, so within each map unit, and indeed across all map units for this analysis, the horizons were standardized into a 0-25 cm layer. Individual horizons were combined and weighted to obtain average wilting point and field capacity values within the standardized layer. Then, for each map unit, the components were averaged according to their specified component fractions to obtain a single map unit value for the standard layer. The map units were then rasterized at 30 m resolution. Finally, PAW was calculated according to Equation 1.

Using ArcGIS, each commodity was extracted from the 2011 CDL raster dataset and projected onto the PAW raster for the region. PAW values and the corresponding number of cells were extracted for each commodity. A weighted average was then calculated using the PAW values and cell count, and each PAW value was normalized to the weighted average. The normalized PAW values were then divided into 10 equal frequency bins, and the corresponding cell counts were carried along into each subsequent bin. Each cell within a bin was assigned a corresponding value, where 1 was the least valuable and 10 the most valuable. Each cell was then reassigned back to its original location and added to its commodity value.

Cucumbers were added into the analysis after the simulation results were completed by the LTM. Cucumber acreage could not be included in the 2011 penalty layer since it was not designated in the 2011 CDL. Instead, cucumber acreage was listed as a combination of corn, soybeans, sugar beets, wheat, and potatoes. An updated 2014 layer was constructed for the post-LTM analyses using the 11 main commodities and subsequent PAW values; in other words, the penalty layer for the LTM included 10 commodities and 10 PAW bins, and the penalty layer used in the analyses inserted cucumber and sorted PAW values into 11 corresponding bins. With this addition, the relative order of PAW value does not change, though the commodity value of cucumber is ranked 8th most valuable. In the LTM, this commodity value for cucumber could not be included, although a large concentration of high-valued commodities are included in the 2011 layer in place of cucumbers (9 for potatoes, 6 for sugar beets, and 5 for corn) indicating that the 2011 score used in the LTM is largely suitable as a proxy for 2014 cucumber values.

When integrated into the LTM, the penalty layer, a sum of the commodity and water storage values, conditioned the ANN to preferentially develop soils with lower values relative to those with higher values. Soil value consideration was coded into the ANN to lower the suitability of a higher-valued soil cell, effectively promoting nearby lower-valued cells as more suitable relative to the higher-valued cell. Development onto high-valued cells was not prohibited, where the ANN was left to determine whether the suitability of a high-valued cell was logical given the influence of the other drivers. All results of the LTM simulations were analyzed using ArcGIS.

3. Results

3.1. Urban Development without a Penalty

Urban development mimicked the trends seen prior to 2011 under the non-penalized scenario, where widespread development occurred onto adjacent farmland and urban areas continued to rapidly extend further away from the mass centroids of the metropolitan areas. Total urban area across the region increased by 2.46 million acres (an increase of 53%) compared to 2011, where 224,000 acres were added to the Detroit metropolitan area (23%), 139,000 acres to Chicago (26%), 68,000 acres to Grand Rapids (35%), 53,000 acres to Toledo (36%), 46,000 acres to South Bend (26%), 29,000 acres to Lansing (33%), 24,000 acres to Kalamazoo (39%), 20,000 acres to Traverse City (73%), and 16,000 acres to Saginaw (30%). Average distances from the mass centroid increased by 14.3 km (52%) for Detroit, 9.2 km (41%) for Chicago, 6.6 km (50%) for Grand Rapids, 6.2 km (57%) for Toledo, 4.3 km (51%) for Lansing, 4.0 km (28%) for South Bend, 3.9 km (56%) for Kalamazoo, 3.4 km (48%) for Saginaw, and 3.2 km (48%) for Traverse City. Urbanization by 2050 consumed over 1.3 million acres of active cropland in 2011, reducing the overall acreage by 8%; blueberries acreage was reduced by 17%, cherries by 16%, apple and cucumbers by 12%, hay by 10%, wheat by 9%, soybeans by 7%, corn and dry beans by 6%, potatoes and sugar beets by 5%.

3.2. Penalty Layer

Results from the summation of the commodity and water storage values across the study area are displayed in Figure 14. The highest value soils are related to the specialty fruits, located primarily in the Traverse City and Grand Rapids regions (displayed using insets in Figure 14, Panel D). There is a clear shift from low- to mid-value soils moving from the southeast to the southwest of the study region, as crop selection transitions from a higher soybean concentration to a higher corn concentration. Mid-Michigan has the widest spectrum of soil values, from low to high, as it is located at the nexus of the low-value soils in the southeast, the mid-value soils in the southwest, and an increasing concentration of specialty crops in the northern half of the region. While a high concentration of low-valued commodities (i.e., soybeans and wheat) are found in the eastern thumb of the region, this area also has the highest concentration of the most valued row crops (i.e., potatoes and sugar beets), generating a wide variety of soil values in a small concentrated area. Panel A highlights the distinct low vs. mid-value commodities, and Panel B highlights the relative soil water available to the overlying commodity. Panel C illustrates the summation of Panels A and B, where the highest valued soils are largely concentrated east of the major metropolitan region. Panel C also illustrates a distinct difference in the value of soils directly along the edges of the metropolitan region. Conceptually, as Toledo urban area expands, the penalty layer serves to encourage the development of the low-value soils first (i.e., purple), followed by the turquoise-colored soils (i.e., mid-value). Panel D displays regional soil-values relative to all urban land.

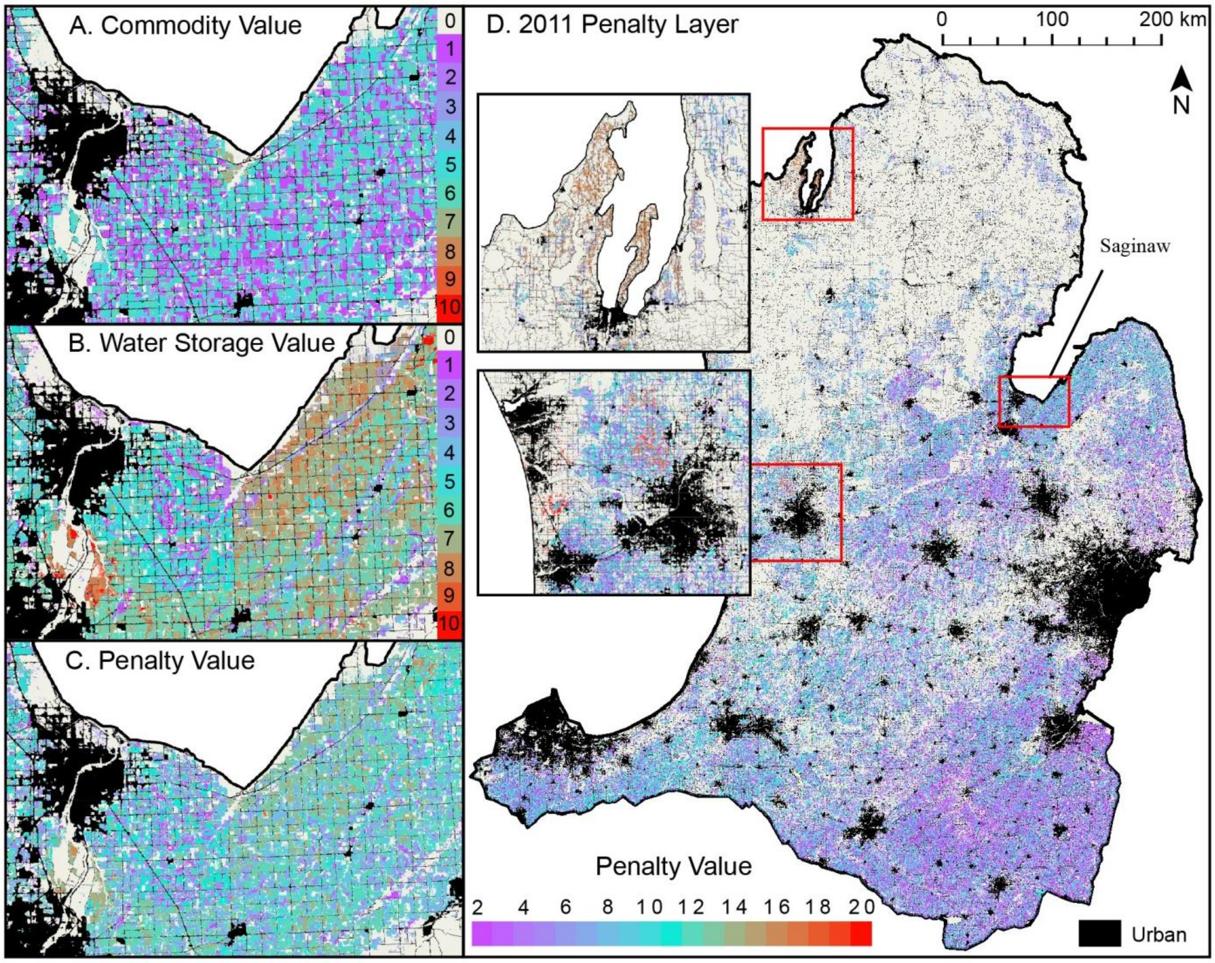

Figure 14. Soil-based penalty layer constructed using commodity and water storage values. Panel A is the commodity value based on market prices and yield per hectare, Panel B is the water storage value based on the moisture available for root uptake, Panel C is the summation of A and B, and Panel D is the regional summation of the commodity and water storage values.

3.3. Urban Development with Penalty

Simulation results for the soil-based development regime are displayed in Figure 15, where panels A-I illustrate each major metropolitan region through 2050.

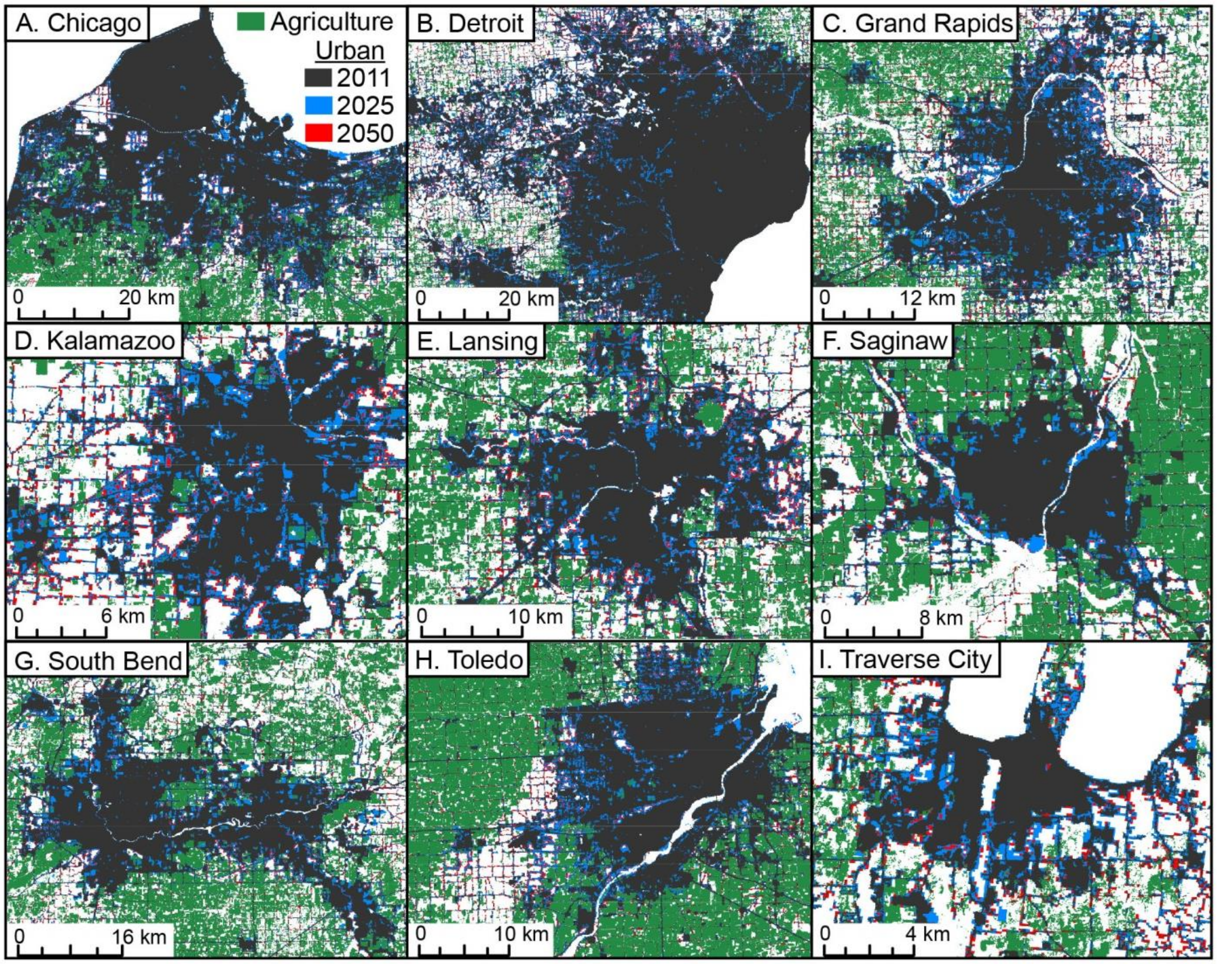

Figure 15. Soil-based development maps for major metropolitan regions. Urban landscapes are depicted on top of 2011 agricultural land.

While the total area of urban land does not drastically increase, trends primarily show intensification in the density of pre-existing urban area through 2025 and expansion onto adjacent farmlands through 2050. Compared to 2011, the percent change in total urban area was 20.0% by 2025 and 53.2% by 2050. Total urban area increased by the same amount when the penalty layer was excluded from the simulation, though the development patterns were not inclined to protect fertile lands in the baseline scenario. Panel A shows that any Chicago area expansion beyond 2050 would directly consume cropland, where Panel B shows that the Detroit region has room to expand without compromising large amounts of agricultural land. Future expansion in Grand Rapids (Panel C) can comfortably occur in the east, while Kalamazoo (Panel D) can develop in the west. Lansing (Panel E), Saginaw (Panel F), and Toledo (Panel G) are tightly locked by agricultural land, meaning any expansion would directly reduce farmland in those regions. Traverse City (Panel H) does have room to expand without compromising farmland, but the farmland that does exist is mainly high value commodities (e.g., cherries, and wine grapes).

3.4. Soil Conservation and Conversion

When comparing simulation results for urban transition areas with and without soil-based development, there was 79% agreement between the location of urban areas in 2025 and 49% in 2050. The disagreement between the simulations was due to the development of alternative land types other than cropland in response to the soil penalty. The percent changes in land cover for summarized NLCD classes are displayed in Table 3 and were calculated using the total areas for each land type at each time period.

Table 3. Percent change in NLCD land types since 2011

	2025		2050	
Land Cover	No Penalty	With Penalty	No Penalty	With Penalty
Barren	-7.9	-8.0	-15.1	-16.1
Forest	-3.7	-4.2	-8.7	-10.4
Shrubland	-2.4	-2.9	-7.7	-9.8
Grassland	-4.8	-5.4	-11.9	-14.3
Cropland	**-2.6**	**-2.3**	**-8.0**	**-6.6**
Wetlands	-3.1	-3.4	-6.7	-7.9

The greatest change in both scenarios was due to conversion of grassland into urban land, followed by shrubland and forest. Adding the soil penalty increased the development of all land types other than cropland from 2011-2050, indicating that the soil-penalty was effective in preserving total agricultural area available for future resource production.

Agricultural land which was developed into urban land during the baseline simulation but not during the soil-based simulation is referred to below as "*conserved*". Effectively, this is the agricultural land that would have been urbanized under traditional development behavior, but instead was not developed due to the soil-based penalty. Agricultural land which was developed into urban land in the soil-based simulation but not the baseline simulation is referred to as "*converted*". This is the agricultural land that would have remained untouched had the soil-based penalty not been integrated. While the soil-based penalty did cause some agricultural soils to become *converted*, much more acreage was *conserved*. Figure 16 illustrates the *conserved* and *converted* soils around Toledo from 2011-2050.

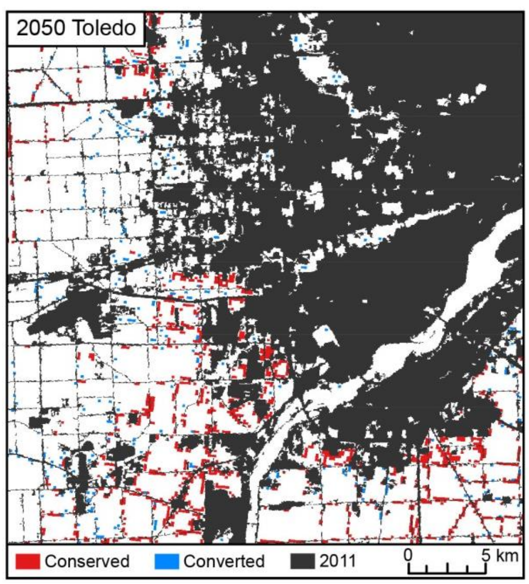

Figure 16. *Conserved* and *converted* soils around the Toledo metropolitan area.

Total *conserved* agricultural land across the study region is divided into major commodities and displayed in Table 4, where positive values represent *conserved* area, and negative values represent *converted* area.

Table 4. Conserved area per commodity with soil-based urbanization since 2011. Percent conserved is calculated based on initial 2011 commodity areas.

Total Acres Conserved Since 2011												
YEAR	Appl.	Blueb.	Cher.	Corn	Cucum.	Dry Be.	Hay	Pota.	Soyb.	S. Bee.	Wheat	Total
2020	-59	-5	-24	13367	19	149	303	64	16064	91	1759	31726
2030	3	-7	222	28613	49	557	1470	194	32332	289	3676	67398
2040	-166	-31	-342	50723	90	1242	3005	374	55971	869	7577	119313
2050	-166	-11	-43	74449	114	2206	4203	484	82632	1656	11159	176684
Percent Area Conserved Since 2011												
YEAR	Appl.	Blueb.	Cher.	Corn	Cucum.	Dry Be.	Hay	Pota.	Soyb.	S. Bee.	Wheat	Total
2020	-0.57	0.07	-0.40	1.90	0.58	0.55	0.22	0.89	2.60	0.25	1.14	0.66
2030	0.06	-0.08	1.21	3.93	1.67	1.54	0.80	3.03	5.02	1.11	2.41	1.88
2040	-1.56	-0.08	-3.24	6.90	3.33	3.26	1.74	5.86	8.47	3.65	4.98	3.03
2050	-1.30	-0.03	-0.30	8.86	3.30	5.36	2.13	7.68	11.44	5.85	7.00	4.55

Both the total number of acres and total percent area *conserved* across all commodities increased as a result of the soil-based development regime. The total area *conserved* compounded through time as urban areas continued to increase in size, further validating the difference between soil-based development and traditional development over long timescales. Corn, soybeans, and wheat are the most widely planted commodities across the region, and they were also three of the most *conserved* commodities. The mid-valued commodities (potatoes, sugar beets, dry beans, and cucumbers) are generally less *conserved* than the major row crops, although potatoes are highly *conserved* relative to their small planted area. The high-valued specialty fruits (apples, blueberries, and cherries) actually lose acreage and are instead *converted* into urban land. However, since conserved and converted values are relative to the simulation without the penalty layer, values can switch from *converted* to *conserved* and back to *converted* (i.e., negative to positive to negative) if less acreage is developed in one year followed by more acreage the next, relative to traditional development. For example, a positive apple and cherry value in 2030 can be explained by high development of those areas during the traditional development simulation over the same time period, relative to the development that occurred in the soil-based simulation. Negative values indicate more development on those areas during the soil-based simulation relative to the traditional simulation over the same time period. Collectively across the major commodities, 176,684 acres are conserved by 2050, the equivalent to 4.55% of their cumulative area.

3.5. Yield Conservation

While spatially distributed yield data on commodities were not available, we can assess the likelihood of preserving higher yielding soils within each commodity--assuming that higher within-commodity PAW values correspond to higher yields. Total conserved PAW values were again grouped into equal frequency bins using the 2014 CDL, and the results are displayed in Figure 17.

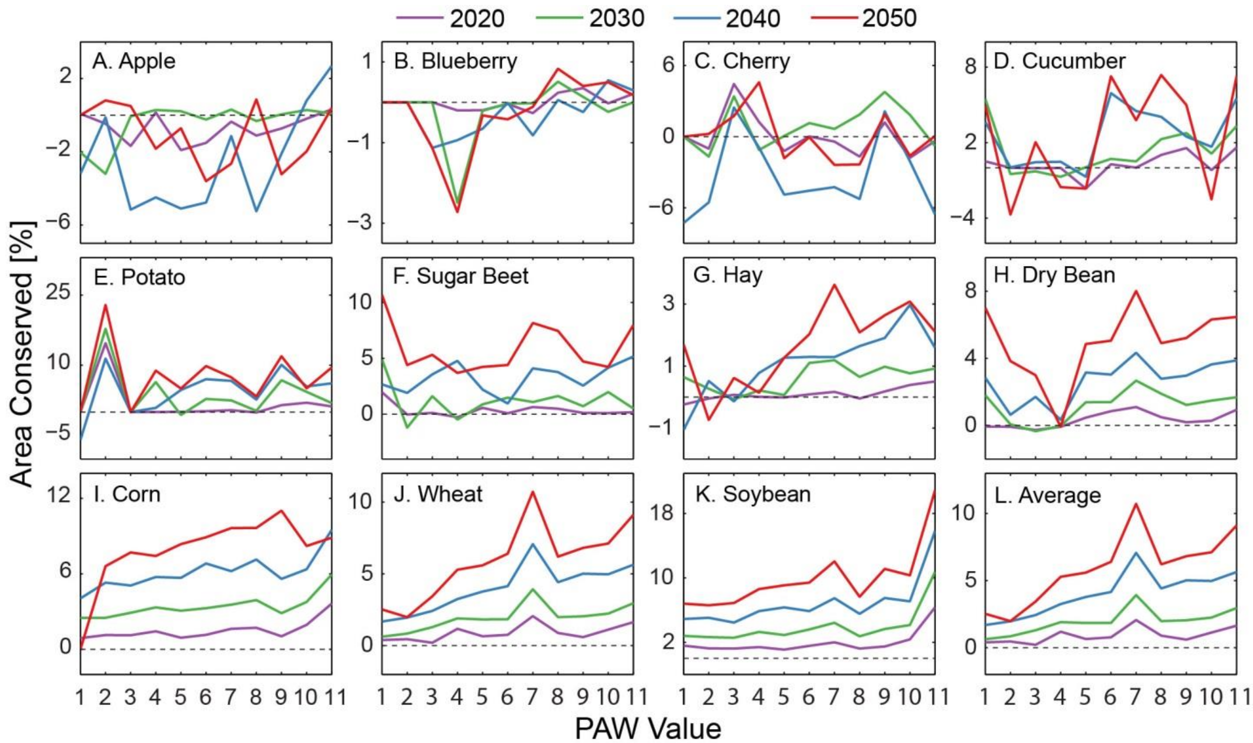

Figure 17. Total percent of soil conserved per PAW bin per decade. Bins are scored 1-11, where 1 is the lowest amount of PAW and 11 is the highest. Percentages are both positive and negative, positive being conserved land and negative being converted land.

Nearly all commodities (panels D-K) demonstrate an increasing trend in the conservation of higher PAW soils, indicating that the soil-based penalty layer did conserve more fertile soils when compared to less fertile soils. This is particularly clear when averaged across all commodities (panel L), where the only exception is PAW bin 7 that is heavily influenced by the conservation of wheat, hay, dry beans, and sugar beets. This increasing trend indicates that the conservation of high fertility soils is not temporally variable across the commodities, but rather magnified through time. Some exceptions include the specialty fruits (panels A-C) which exhibit widely variable conservation behavior and ultimately have more soil converted to urban land than is conserved. However, opposite to *conserved* soils, *converted* land is not linear through time. This is indicated by the temporal disorder of area conserved compared to the consistent order (2020 < 2030 < 2040 < 2050) largely found across other commodities. Disorder likely indicates that the penalty layer does have influence in resisting the conversion of these soils, as evidenced by some relatively high peaks across years and bins, but other factors override the soil penalty in these regions, as evidenced by the low peaks and differing peaks in each year. In other words, there are periods where these commodities are conserved, indicating that the penalty layer is preventing development. However, at other periods, the soil layer is not influential enough to prevent development, and urbanization consumes these commodity areas. This relationship indicates a pattern where soils are conserved until other incentives outweigh the penalty layer and development occurs.

High-valued soil conservation is also evident at the metropolitan scale. The total developed acres on each PAW value for both non-penalized and penalized scenarios in 2050 compared to 2011 are displayed if Figure 18, where both the left and right sides within panels are positive acreage values.

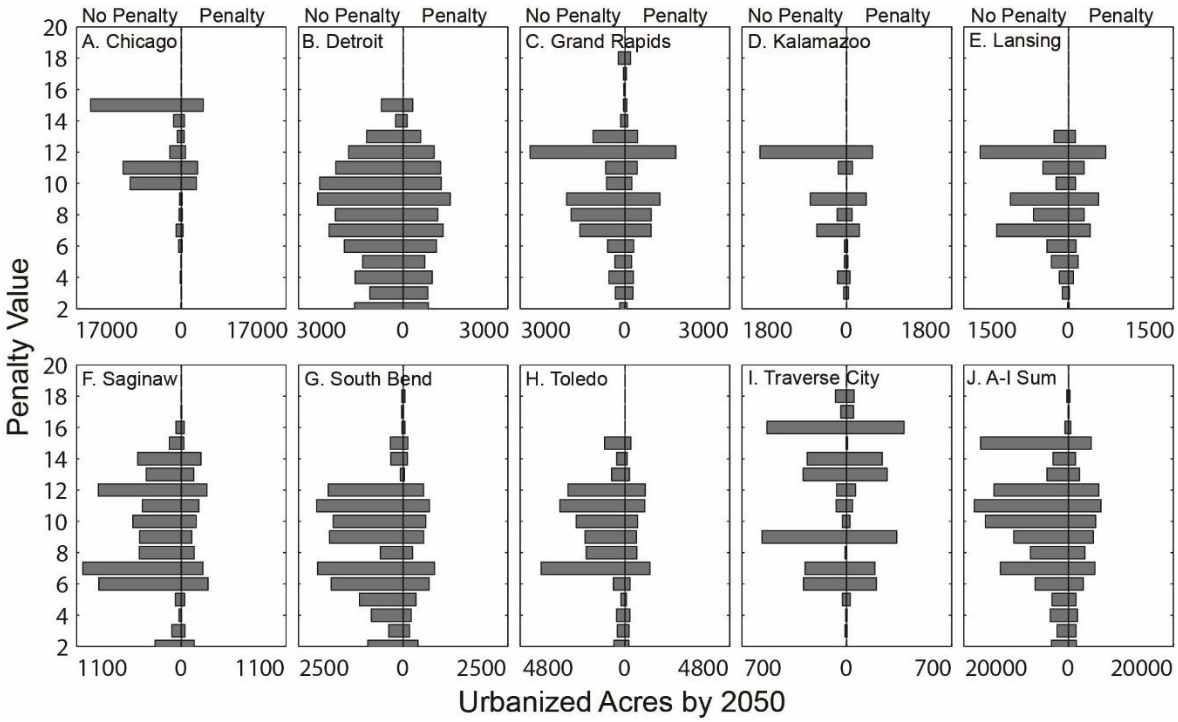

Figure 18. Total urbanized acres per penalty value for non-penalized (left within panels) and penalized (right within panels) scenarios in 2050 compared to 2011.

Considerably more commodity acreage was developed during the simulation without the penalty layer, and the total impact of the layer is also highlighted by the largest differences occurring within soil values of 8-15. In particular, major differences occur in Chicago (Panel A), Kalamazoo (Panel D), both of which are surrounded by a high percentage of the mid- to high-value corn acreage. Other major differences occur in Saginaw (Panel F), South Bend (Panel G), and Toledo (Panel H), though these occur across a wider margin of soil values due to the increase in commodity diversity, both at lower values (e.g., soybeans and potatoes) and mid-values (e.g., potatoes, sugar beets). The development of the high value soils is inconsistent between each simulation, where these soils were less-developed in some regions and more developed in others. The total magnitude of developed high value soils is also much less given significantly less total harvested area of the specialty fruits. Overall, nearly all soil values were less developed when implementing soil-based development (Panel J).

3.6. Revenue Conservation

The economic impact of the soil-based development can be calculated for each commodity by multiplying the average yield per acre, the acres conserved or converted, and 2014-adjusted market values of the commodities. Conserved revenue results are displayed in Table 5, where positive values represent revenue retained through *conserved* land, where negative values represent revenue lost through *converted* land.

Table 5. Conserved revenue per commodity with soil-based urbanization since 2011

Revenue Conserved ($) per Commodity Since 2011 (x 1,000)												
YEAR	Appl.	Blueb.	Cher.	Corn	Cucum.	Dry Be.	Hay	Pota.	Soyb.	S. Beets	Wheat	Total
2020	-340	-39	-75	8,073	60	80	134	210	6,281	120	707	15,211
2030	20	-61	699	17,282	159	299	651	642	12,642	379	1,478	34,189
2040	-957	-257	-1,078	30,637	291	667	1,331	1,236	21,885	1,140	3,046	57,941
2050	-958	-87	-137	44,967	371	1,185	1,862	1,600	32,309	2,171	4,486	87,769

Corn and soybeans are the main sources for conserved revenue, as they conserved ~5-40 times the revenue of the other commodities. Apples, blueberries, and cherries result in negative revenue as their area was converted rather than conserved under the soil-based development regime. By 2050, the total revenue conserved is ~$88 million, up more than $50 million when compared to 2030. This effectively is revenue that would be lost if a soil-based development plan were not to be implemented. However, the $88 million valuation in 2050 is only for one year. When extrapolating between 2011 ($0), 2020, 2030, 2040, and 2050 to calculate annual values, the total revenue conserved since 2011 with soil-based development is more than $1.5 billion.

Conserved revenue roughly follows a linear trend line, where the trend is highly dependent on market prices. It is also important to note that conserved revenue is only a valuation of the crop production and does not include the value of the urban land. In reality, the value of urban land may outweigh the value of crop production, indicating that revenue was in fact not conserved by preserving the farmland. However, the economic value of urban land is widely variable and not addressed in this study.

3.7. Urban Density

The total number of acres touching urban land by 2050 in the baseline simulation was 94,640 acres more than the soil-based simulation, confirming that urban density increased when a soil-based penalty was integrated. The densification of urban areas at the metropolitan scale can also be represented by the distances of unique (i.e., developed in one scenario and not both) urban transition acres relative to the mass centroid of the given metropolitan area. The average distances of each unique urban area in 2050 compared to 2011, and between the non-penalized and penalized scenarios, are displayed in Figure 19, where both the left and right sides within panels are positive acreage values.

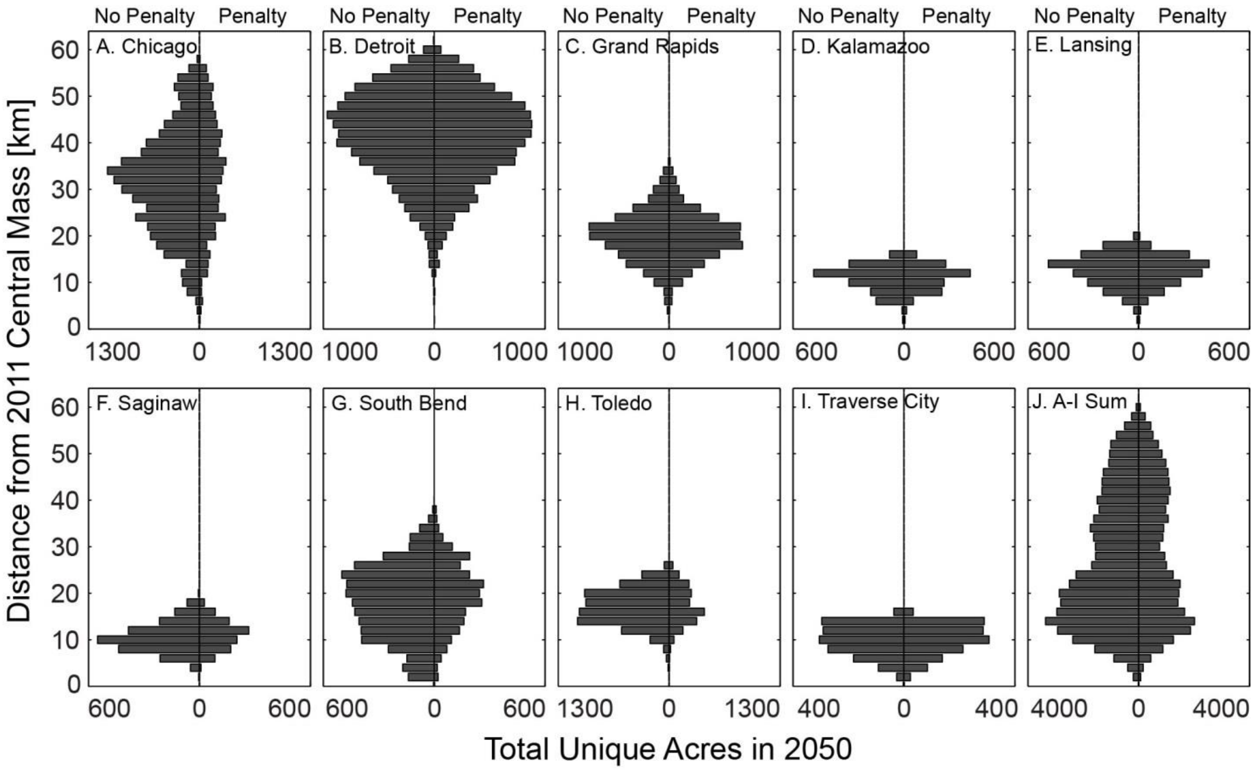

Figure 19. Distance of total unique urban transition acres in 2050 relative to 2011 metropolitan centroids for non-penalized (left within panels) and penalized scenarios (right within panels).

Increased densification of metropolitan areas under soil-based development is evidenced by the decreased distance of urban transitional acres relative to the non-penalized scenario. In particular, Chicago (Panel A), Saginaw (Panel F), and Toledo (Panel H) depict the greatest difference between scenarios. These metropolitan regions are also tightly surrounded by agricultural land, which would inhibit expansion away from the centroid. Contrarily, Detroit (Panel B), Grand Rapids (Panel C), Kalamazoo (Panel D), Lansing (Panel E), and Traverse City (Panel I) have distance patterns that closely mirror the opposite scenario. These regions are much less confined by adjacent commodities, relative to Chicago, Saginaw, and Toledo (refer to Figure 15). It is plausible that these patterns will become less symmetrical beyond 2050 as agricultural land becomes more confining.

4. Discussion

Urban expansion is inevitable with a growing population located in an unbounded location, like the Lower Peninsula of Michigan. Thus, protecting high-value soils for agricultural use requires conversion of other land types (e.g., forests, shrub, and barren). While the conservation of farmland may help protect revenue and food security, it is important to keep in the mind the implications that occur with the development of other land types. For example, deforestation can lead to a reduction in overall biodiversity (Seto et al., 2012) and can alter regional hydrologic and biogeochemical cycles (Chadwick et al., 2006; Tayyebi et al., 2015). Outdoor recreation is a major industry in the Lower Peninsula, and a reduction in natural lands can also have a negative impact on the local economy. Other land conversion data also indicated that traditional urbanization heavily altered forested and grassland regions (Jantz et al., 2005), and here we found that our soil-based regime even more intensely altered these other land class types. Notably, our penalty designed to conserve cropland was successful, suggesting that similar penalty layers could also be designed for the protection of forests, grasslands, or other land class types. The compilation of all land class types can also be constructed into a comprehensive penalty layer if the economic value, or value as a future resource, can be comparably calculated for all land covers. A comprehensive penalty would be a benefit to the overall conservation of the most valuable lands.

Corn, soybeans, wheat, and hay are the most widely planted commodities across the region, thus they are the most likely to be conserved. This is also due to their close proximity to the major metropolitan regions, both of which occupy the southern half of the study region. The mid-valued commodities (potatoes, sugar beets, and cucumbers) are less conserved compared to the major row crops, but this may also be attributed to their location relative to other metropolitan regions. These commodities are primarily located around the Saginaw Bay area along the eastern border of Michigan's Lower Peninsula, suggesting that conservation will only occur if unprecedented amounts of development were to occur along the eastern side of the study region. Conservation of these commodities may be more evident in years beyond 2050, when populations are higher and space for development is more competitive.

The soil-based penalty layer failed to conserve large areas of the specialty fruits, even though they were the soils with the highest value. We suggest this may have occurred due to two reasons. First, the location of these fruits is directly adjacent to popular metropolitan regions along the western half of the boundary (e.g., Grand Rapids, Traverse City), suggesting that expansion onto these soils is likely unless specific policies are put in place to protect them (e.g., a nonlinear soil-based constraint). Regardless, the penalty layer was designed to encourage development away from these high-value soils, and they ultimately were not conserved. This indicates that an alternative behavior may have been captured. Here, the cost-savings of developing a different soil type may be lessened by the increase in value of other nearby commodities. For example, if an acre of blueberry land were in a prime development location, that acre could have a high price tag under a traditional development scenario when compared to a nearby acre of corn. However, if the nearby acre of corn had high water storage value (high PAW), the soil-based constraint would increase the value of that corn acre, shrinking the cost savings between the two acres of land. This reduction in savings may thus encourage a buyer to develop the prime blueberry acreage, if the location were more desired than the corn. This is evidenced by the temporal disorder across PAW values for these commodities (Figure 18), suggesting that the soil-based constraint may be successfully inhibiting development for a short duration of time, as indicated by the conserved land for apples and cherries in 2030, until pressure to develop the valuable soils outweighs the incentive to develop lower-valued soils. Development onto these lands then occurs in a nonlinear order, resulting in the variable PAW conversion patterns. If these specialty fruits are to be conserved under a soil-based constraint regime, additional protection or incentive away from these commodities may be necessary.

The soil-based development was also successful at conserving the most fertile soils compared to the least fertile soils per commodity. This is a significant result given that others have found most urban development in the United States to occur on the most fertile soils (Imhoff et al., 2004). Here, we also translated fertile conservation to revenue conservation. However, the amount of conserved revenue was calculated using state-wide yield averages for each commodity. If the least fertile soils are being developed, as demonstrated, this would increase the average yield, ultimately increasing the average revenue conserved per acre. The revenues reported here are thus under-representations of the actual revenue conserved.

Revenue and soil value are also dependent on market prices, which are highly variable through time. Our penalty layer was constructed using only the market prices from 2011. If a soil-based penalty were to be implemented into actual practice, it could benefit from being revalued at shorter timescales. Integrating a temporally dynamic penalty would likely better represent the soils that society deems valuable.

One limitation to this study is that the penalty layer is not variable through time. As mentioned, market prices are widely variable and would result in temporally variably commodity values. However, what is more important is the value assigned to an ever-increasing commodity. For example, as cherry acreage becomes urbanized, the remaining cherry acreage would likely become more valuable. This study does not incorporate a value based on a depleting supply. A second limitation to this study is that this was designed to be conceptual and all land was available for development, including protected lands (e.g., state forests and wetlands). In reality, these areas would not be permitted for development. This methodology can integrate protected lands prior to actual implementation. A third limitation is that the penalty layer did not incorporate a technology driver, which may artificially change the value of a commodity. For example, a low-valued PAW soil may have access to irrigation, effectively converting its true production to levels seen at high-valued PAW soils.

Future research prior to the application of this methodology would require a full analysis on the impacts of other ecosystem cycles. For example, while this study discussed increased urban density as a significant result, this would likely intensify stream flows, increase runoff, and increase localized surface water contamination (Hatt et al., 2004; Eigenbrod et al., 2011). A valuable second research pursuit is to isolate the high-valued soils and investigate the dynamic that is causing high-valued soils to be consumed. Lastly, if all land covers are to be integrated, it is necessary for values to be normalized across land types (e.g., an acre of forest compared to an acre of grassland). While much research has been completed to evaluate the economic or societal value of these land resources (e.g., Adams, 2014, Woodward and Wui, 2001; Pearce, 2001), most attention has been paid to the services provided by each land cover and little has been paid to the connection of soil-value. Furthermore, soil values are highly variable across both space and time. More work on the topic of land value is critical for the successful implementation of future land conservation strategies.

5. Conclusion

This study sought to integrate soil-based urbanization as a way to conserve valuable soils and maximize production and annual revenues across the LP watershed region. There are six key conclusions from this study:

(1) Over $1.5 billion of agricultural value could likely be conserved by 2050 if soil-based urbanization had been implemented in 2011 and stayed in effect. Revenue savings come from the yield on farmlands that would have been otherwise consumed under traditional urbanization patterns. Conserved revenue will only increase more rapidly beyond 2050, as more agricultural land remains available for production. These conserved soils also act as a reserve to meet the future resource demands of a growing society.

(2) The most fertile soils are more conserved than the least fertile soils, including at longer timescales, and the rate at which high fertility soils are conserved is greater than low fertility soils. The best soils for high-yield crop production are conserved under soil-based urbanization simulation, further indicating that the available agricultural land will be the most capable of meeting future resource demands.

(3) Soil-based urbanization promotes the intensification of urban density. By 2050, the total number of acres touching urban land in the baseline simulation was 94,640 acres more than the soil-based simulation. Furthermore, average distance of a new urban acre in each major metropolitan area increased relative to its corresponding mass centroid during the baseline simulation compared to the soil-based scenario.

(4) Based on the successful conservation of agricultural land in this study, the soil-based constraint methodology presented here can be transposed to the conservation efforts of other individual land types (e.g., forests, grasslands). Other land covers were consumed to accommodate for the urban development that would otherwise have occurred on adjacent croplands. If other land cover protection is desired, then individual land types will need to establish value metrics that can be assigned within a constraint layer. These methods can also incorporate all land types simultaneously if soil value can be normalized across land covers.

(5) The highest valued soils may need additional protection within a soil-based constraint, as evidenced by the increased consumption of specialty fruit acreage during the soil-based constraint. It is plausible that the incentive to conserve high-valued soils becomes less in actual practice as nearby soils, not traditionally valued by non-socially derived parameters such as available water storage, are made more valuable. This is a particularly relevant dynamic across this study region as the specialty fruits are restricted to very specific growing conditions found only in relatively small niche areas across the state.

(6) As agricultural land is consumed, it would be valuable to implement a differential weighting system rather than a linear valuation. In other words, as commodities become scarcer, their value may increase. For this method to be implemented in a real management scenario, it would be important to allow for a dynamic valuation that follows commodity availability and market fluctuations.

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CHAPTER 4

Increased Dependence on Irrigated Production across the CONUS

Abstract

Efficient irrigation technologies, which seem to promise reduced production costs and total water consumption in heavily irrigated areas, may instead be driving increased applications in areas not traditionally irrigated. As a result, the total dependence on supplemental water applications for both crop production and total revenue may be steadily increasing across the continental United States. However, comprehensive irrigated and dryland yield and acreage data are unavailable outside of major aquifer regions, despite the importance and long history of irrigation applications in agricultural practices. This study uses a linear regression model to predict average irrigated and dryland yields at the state level for 5 major row crops: corn, cotton, hay, soybeans, and wheat. For 1945-2015, we quantify crop production, irrigation enhancement revenue, and irrigated and dryland acreages in marginal states where both irrigated and dryland farming practices are implemented. In 2015 alone, we found that total revenue due to irrigation enhancement was $7 billion and 37% of total production relied on irrigated lands. There was a clear response to biofuel demand, with the addition of more than 3.6 million hectares of irrigated corn and soybeans in the last decade. Since 1945, over $465 billion in revenue has resulted from the yield enhancement due to irrigation relative to predicted dryland yields, and over 4 million irrigated hectares were added since 2002. Here, we establish a baseline of irrigated and dryland acreage trends in marginal states and identify a growing market and agricultural dependence on irrigation, which ultimately leads to increased risk in economic and agricultural markets.

Keywords: Irrigation, Food Production, Irrigation Efficiency, Water Use, Agriculture

1. Introduction

Average annual crop production across the continental United States (CONUS) from 2010-2014 was valued at ~$205 billion (NASS-USDA, 2012); $145 billion of which came from only 5 commodities: corn, cotton, hay, soybeans, and wheat. Approximately $75 billion in annual sales can be attributed to just 4 major aquifer regions (California's Central Valley, the Snake River Valley/Columbia River Plateau, the High Plains Aquifer, and the Mississippi River Alluvial Plain) demonstrating that high crop production is directly correlated to the accessibility of extensive freshwater supplies that can be used for irrigation. Irrigated yields are more than double dryland yields in some regions (Smidt et al., 2016; Steduto et al., 2012), and farmers often take advantage of this incentive as evidenced by the high volume of total withdrawals across the CONUS (>115 billion gallons per day; USGS, 2016). In some regions like the High Plains and California's Central Valley, crop production is so reliant on irrigation withdrawals that farmers continue to expand irrigated acreage despite significant groundwater depletion (Haacker et al., 2015; Scanlon et al., 2012). This connection between irrigation, high-yield crop production, and groundwater level decline has led to the development and adoption of efficient irrigation technologies, where the primary goals are to reduce production costs and water consumption, while increasing or maintaining yields (Schuck et al., 2005; Howell, 2001).

Modern agricultural practices appear to be evolving in response to increased irrigation efficiency and lower production costs. While many researchers have documented that efficient irrigation technologies have reduced water consumption relative to inefficient systems (e.g., Schaible and Aillery; 2012; Thomspon et al., 2009; Seo et al., 2008), others have documented that efficient irrigation technologies have actually led to more water extraction, as the cost of water-use declines (Smidt et al., 2016; Pfeiffer and Lin, 2014; Ward and Pulido-Velazquez, 2008). This behavior is most evident in major aquifer regions, but recent trends indicate that farmers outside of major aquifer regions may also be adopting such irrigation practices. For example, irrigated area and total water withdrawals have increased by 24% and 8%, respectively, from 2000-2010 across the 25 least irrigating (USGS, 2016). One explanation is that areas with a marginal need to irrigate can do so at a higher return on investment, and areas with little or no need to irrigate can do so as a cost-effective way to mitigate drought risk (Baker et al., 2012; Wilhelmi and Wilhiti, 2002). Crop selection in some regions has transitioned from regionally-suited dryland commodities to more water-intensive crops, as irrigation allows farmers to produce the commodity with the greatest market value and return on investment, regardless of water demand. For example, corn and soybean areas across the Great Lakes Region (WI, IL, IN, MI, OH) increased by 672,000 and 142,000 hectares, respectively, from 2000-2015, despite the overall area for the major row crops decreasing by 832,000 hectares within this same region (NASS-USDA). An evolving industry suggests that the dependence on irrigation for both production and revenue has steadily increased in recent decades. However, comprehensive irrigated yield and acreage data is critically lacking within major aquifer regions and is nonexistent across much of the CONUS.

Here, we identify: (1) the economic risk associated with artificially high yields due to human-induced water applications, and (2) how states have evolved in their irrigated crop selection since the introduction of overhead irrigation systems. We use historically comprehensive average yield data to construct a mixed-end member linear model as a way to quantify irrigated and dryland production for the 5 major row crop commodities across the CONUS from 1950-2015. We present 65 years of unprecedented irrigated and dryland decision-making trends that highlight the patterns in irrigation of major row crops and the economic risk linked to water-use dependency, particularly in marginal states outside of major aquifer regions.

2. Irrigation Enhancement

The end of World War II in 1945 signaled the start of a new era in agriculture, both in agricultural technology and crop production. The rapid increase in crop yield post-1945 can be seen in every state across the CONUS, but a very distinct change in the rate of yield increase is observed in states with heavy irrigation practices when compared to those with traditional dryland practices. To demonstrate this yield change, the 48 CONUS states were grouped into one of two categories, either irrigated or dryland, defined by a pumping threshold of 150 Mgal/day per 4,000 hectares of agricultural land; there are 21 irrigated and 27 dryland states using this classification. The post-1945 "hockey stick" effect is displayed in Figure 20 for corn yields across the CONUS.

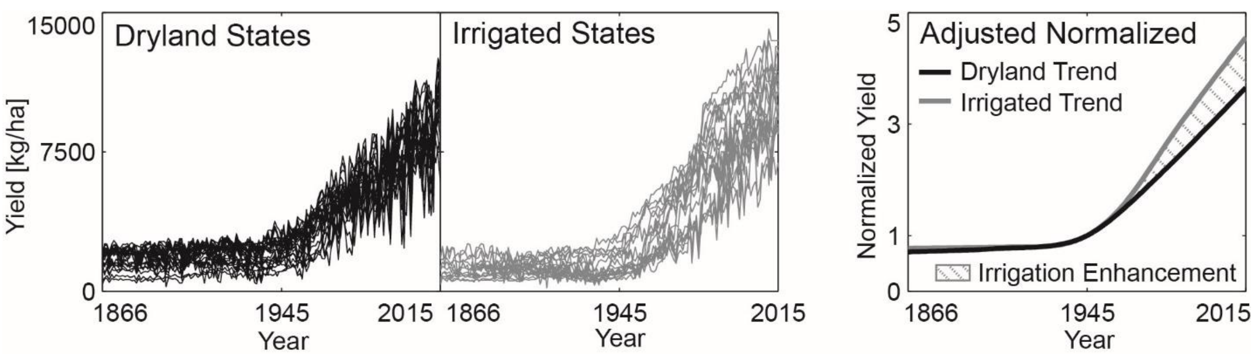

Figure 20. Corn yields for dryland and irrigated states (left) and mean, 1945-normalized yield enhancement due to irrigation (right).

In theory, both groups of states should have similar yield trends in the period before and after 1945, assuming new irrigation technologies had not entered the market and all states had access to the same innovative agricultural technologies that would have been introduced. However, many arid states (e.g., California) do have access to expansive groundwater reserves that allow for irrigation pumping at a magnitude large enough to be economically feasible using center-pivot technology. As a result, the rate of yield increase for irrigated states outpaces the increase for dryland states. This difference is likely due to the yield enhancement associated with irrigation. In other words, the only variable that is unique between dryland and irrigated states after 1945 is access to irrigation. Thus, any boosts to yields in irrigated states relative to dryland states must be attributed to irrigation enhancement. The differences in yield for each state group and the subsequent irrigation enhancement for corn are displayed in Figure 20, where dryland and irrigated trends are shown for the normalized group average, and the irrigated trend is adjusted to the 1945 dryland value to identify irrigation enhancement.

3. Modeling Framework

To investigate irrigation enhancement and evaluate cropping patterns from 1945-2015, we developed a mixed-end member linear model to predict dryland and irrigated yields for states using both dryland and irrigated practices (i.e., marginal states with moderate demand and availability for irrigation). “Irrigated” end members were assigned using a threshold of less than 200 cropland hectares per Mgal/day of irrigation pumping (USGS, 2016). “Dryland” end members were assigned as those with more than 20,000 cropland hectares per Mgal/day of irrigation pumping. “Mixed” states, or referred to as marginal states, with both dryland and irrigated practices, were those that fell between these two thresholds. According to this classification, there are 11 dryland, 8 irrigated, and 29 mixed states. One exception was Florida, which classifies as “irrigated” but was switched to “mixed” since most of its irrigation is used for commodities outside of the 5 major row crops analyzed in this study.

A series of predictor variables were assigned to each end member group to correlate irrigated and dryland yields relative to state-specific characteristics. Predictor variables included a combination of time (a proxy for improvements in technology, management, and crop genetics), growing season precipitation (March to October; NOAA-CDO), growing season temperature (NOAA-CDO), state-centered latitude, state-centered longitude, average annual recharge (calculated in ArcGIS as the statewide average recharge beneath agricultural lands; NLCD, 2011; USGS, 2003), and the state-average yields from the other remaining row crop commodities (NASS-USDA). A multiple linear regression model was then run between the predictor variables and the end-member yields to derive coefficient estimates to be used to predict dryland and irrigated yields for each mixed state, based on the predictor variables corresponding to each mixed state.

In summary, our model estimates four main values, each at the state level: (1) irrigated yield, (2) dryland yield, (3) irrigated acreage, and (4) dryland acreage. Given that total agricultural production can be calculated by multiplying yield and acreage, we were also able to estimate total production, total irrigated production, and total dryland production for each state.

4. Model Validation

Preliminary calibration and validation of the linear model was conducted in two steps: (1) using every-other year yield data and statistically significant predictor variables to derive coefficient estimates and (2) applying the coefficient estimates to the other set of every-year year data and variables to make yield predictions. Predicted vs. actual yield, and correlation strength, for each commodity across mixed states are displayed in Figure 21.

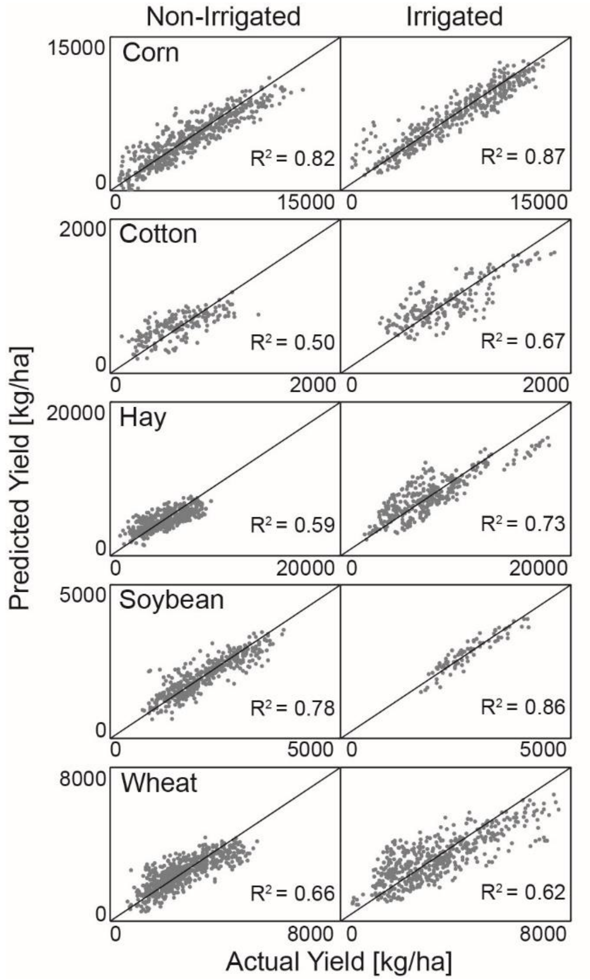

Figure 21. Multiple linear regression model calibration between actual and predicted yields across all irrigated and dryland states.

In addition to the calibration and validation using every other year data, model validation was conducted using both: (1) the 2013 Farm and Ranch Irrigation Survey (USDA, 2014) dryland and irrigated yield data (NASS-USDA) compared to 2013 predicted values, and (2) the actual annual production compared to predicted annual production using the NASS Survey data (NASS-USDA).

The only year the NASS has complete dryland and irrigated yield data, from either the Agricultural Census (every 5 years) or annual Survey, for each commodity and state is the 2013 Farm and Ranch Irrigation Survey (USDA, 2014), part of the 2012 Agricultural Census (NASS-

USDA). We compared our predicted 2013 yield values to the 2013 Farm and Ranch Irrigation Survey values, and the percent difference is displayed in Table 6. Given that yield in any year is a widely variable statistic, the percent difference between predicted and observed production is small enough to have confidence in the model predictions. Dryland yields had closer agreement than irrigated yields due to irrigation having a wider range of impacts on yields across each mixed state, where the upper limits to dryland yields are limited by natural precipitation.

Table 6. Percent difference between 2013 predicted and observed dryland and irrigated yields for each commodity across all mixed states

Commodity	Dryland [% diff.]	Irrigated [% diff.]
Corn	-6.6	1.8
Cotton	8.3	21.2
Hay	-2.6	24.0
Soybean	-2.6	7.4
Wheat	-5.3	8.8

To evaluate the predicted irrigated and dryland areas, we compared them relative to observed production data. The 2013 Farm and Ranch Irrigation Survey contains irrigated and dryland areas, but agreement between the NASS Survey and the 2013 Farm and Ranch Irrigation Survey for these values was no greater than the 58% agreement among wheat acreages. As a result, actual production was calculated as the Survey yield values multiplied by the area for each state and commodity. Predicted production was calculated as the predicted dryland and irrigated yields multiplied by the predicted dryland and irrigated areas, respectively, for each commodity and each state. The observed and predicted production values are displayed in Figure 22, along with the average annual percent difference. Predicted production closely follows actual production, including capturing specific peak patterns. Given that predicted production relies on highly variable yields and predicted areas derived from these highly variable yields, the strong correlation between predicted and actual production indicates a robust model. The largest differences between actual and predicted production are found in hay and wheat post-1990, which each have the greatest variability in upper limits of irrigated yield across mixed states.

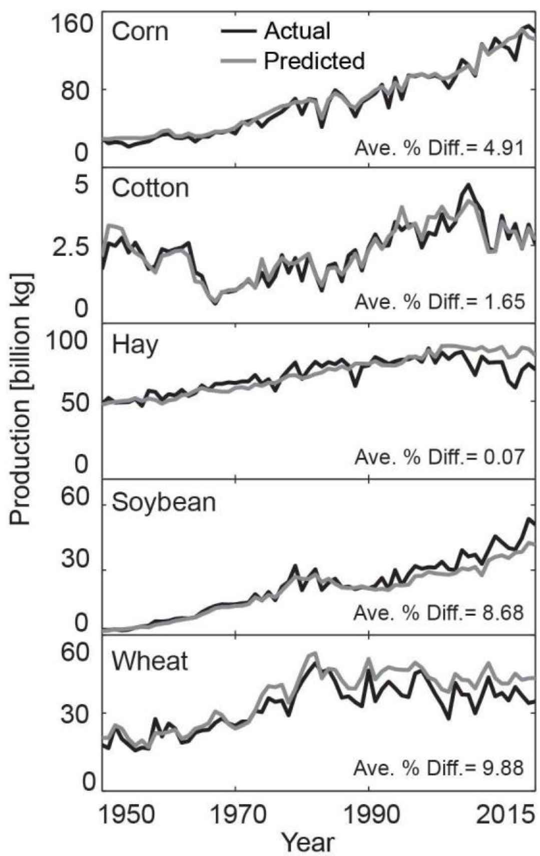

Figure 22. Observed total annual production compared to predicted (dryland + irrigated) production for each commodity across all mixed states

5. How dependent are marginal states on irrigated production?

Irrigated dependence indicates the percentage of the total crop production that is derived from irrigated area. In 2015, total irrigation dependence across mixed states was 37%, indicating that over a third of the production was derived from irrigated practices. Irrigation dependence was calculated using the predicted dryland and irrigated production values as upper and lower limits within each state. Here, the upper limit represented 100% irrigated production, and the lower limit represented 0% irrigated production. The percent production from irrigation for each state and commodity was then quantified using eq. 1:

$$\frac{\text{Actual Production} - \text{Predicted Dryland Production}}{\text{Predicted Irrigated Production} - \text{Predicted Dryland Production}} \qquad \text{(eq. 1)}$$

Total percent irrigated production for each mixed state and commodity was then area weighted and summed to quantify total production dependence for all mixed states.

Two major peaks are displayed for total irrigation dependence: one in the late 1950's with a peak dependence of 41% in 1959, and another starting in 2007 with a peak dependence of 38% in 2013 (Figure 23). Corn production steadily increased in irrigation dependence until ~1980 when dependence stalled and maintained a relatively steady value around the 2015 value of 38%. Cotton production has been the most dependent on irrigated production, peaking once in 1958 at 73% and again in 2015 at 77%, while rising consistently since the late 1970's. Hay production was very dependent on irrigation from ~1960-1980, with a peak value of 57% in 1968, when a steady decline began that has continued to a dependence value of just 4% in 2015. Soybean production was originally dependent on irrigation, peaking at 40% in 1960, but quickly declined to nearly zero from 1975-1990. Since then, this value steadily increased to 26% in 2015. Wheat production dependence has remained fairly stable at approximately 10% since 1950.

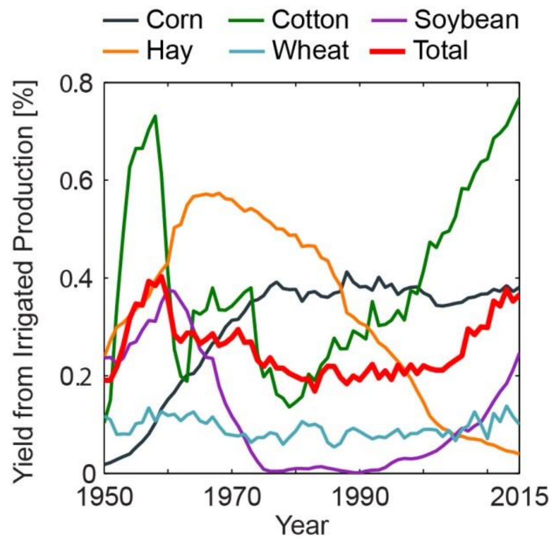

Figure 23. Irrigation production dependence for each commodity across all mixed states

Dependence values are independent from market prices, indicating that any rise or fall in dependency occurs for one of two reasons: (1) irrigated yields increased relative to dryland yields, or (2) irrigated area increased relative to dryland area, both of which may be influenced by technology or policy changes.

6. How have irrigated crop trends changed?

Predicted dryland yields, irrigated yields, and observed production were used to quantify dryland and irrigated areas using equations 2, 3, and 4.

$$\text{Production} = \text{Dryland Yield} * \text{Dryland Acreage} + \text{Irrigated Yield} * \text{Irrigated Acreage} \qquad \text{(eq. 2)}$$

$$\text{Total Area} = \text{Dryland Area} + \text{Irrigated Area} \qquad \text{(eq. 3)}$$

$$\text{Irrigated Area} = \frac{\text{Production} - \text{Dryland Yield} * \text{Total Area}}{\text{Irrigated Yield} - \text{Dryland Yield}} \qquad \text{(eq. 4)}$$

Calculated dryland and irrigated areas for each mixed state were then summed together to quantify the total dryland and irrigated areas for each commodity from 1950-2015. Changes to dryland and irrigated areas (Figure 24) identify changes in crop selection as well as additions or reductions in total irrigation practices.

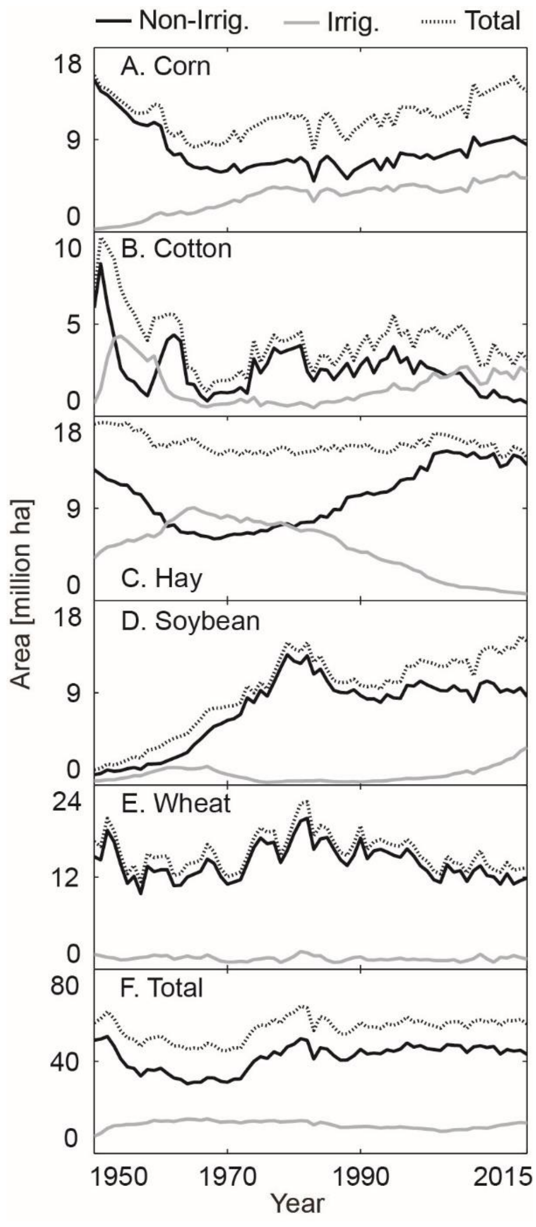

Figure 24. Predicted dryland, irrigated, and total areas for each commodity across all mixed states.

The most notable change is increase of both irrigated and total area for corn and soybeans over the last two decades, indicating that farmers have switched to growing these irrigated commodities at the expense of other declining crops. Total and irrigated wheat, and hay to a lesser extent, have both declined in recent years, suggesting that this land has transitioned into irrigated corn and soybeans. Total irrigated area has also increased while total dryland area has decreased, further demonstrating the increase in irrigation dependence.

Total irrigated area was the highest from 1955 to 1985, when a dip occurred that lasted until 2002 (Figure 24F). Since then, there has been a steady increase to 13.3 million irrigated hectares in 2015 relative to 9.6 million hectares in 2002. Total dryland area has also slowly declined since 2007, dropping from 46.3 million hectares to 43.0 million hectares in 2015. Irrigated corn area has gradually increased from 273 thousand hectares in 1950 to 5.2 million hectares in 2015 (Figure 24A). Dryland and total areas have also increased for corn, peaking at 9.2 and 15.1 million hectares in 2013, respectively. Dryland cotton area has rapidly declined since 1995, dropping from 3.8 million hectares to 716 thousand hectares in 2015. However, irrigated cotton has steadily increased since 1983 (Figure 24B), rising from 451 thousand hectares to 2.4 million hectares in 2015. Total cotton area has remained fairly stable around 4 million hectares, as new irrigated area has replaced depleting dryland area. Initially, hay was largely irrigated, peaking at 9.1 million hectares in 1965, but has since rapidly declined to 631 thousand hectares in 2015 (Figure 24C). Total hay area has remained fairly stable since 1945, totaling ~15 million hectares. Soybean area has primarily been dryland, but there has been a rapid increase in irrigated land and an addition to total soybean area over the past two decades (Figure 24D). Irrigated area has increased to 3.38 million hectares in 2015 from 17.8 thousand hectares in 1989. Dryland soybean area has remained relatively stable since 1986 at ~9.0 million hectares. Irrigated wheat has remained relatively low, near 1.3 million hectares, while total and dryland areas have steadily declined to since 1982, both declining over 39%.

7. How much revenue is from irrigation enhancement?

Irrigation enhancement revenue is a market-based measure of the production due to irrigation, relative to what would have been expected using production practices similar to those in dryland states (i.e., the difference between actual irrigated production by state and the average production trend derived from the normalized average of dryland states). When irrigated states are also included with mixed states (mixed + irrigated), we estimate there has been total irrigation enhancement revenue since 1945 of $466 billion across the five major row crops; $187 billion for hay, $118 billion for corn, $115 billion for cotton, $32 billion for wheat, and $14 billion for soybeans. Revenues for each commodity are displayed in Figure 25.

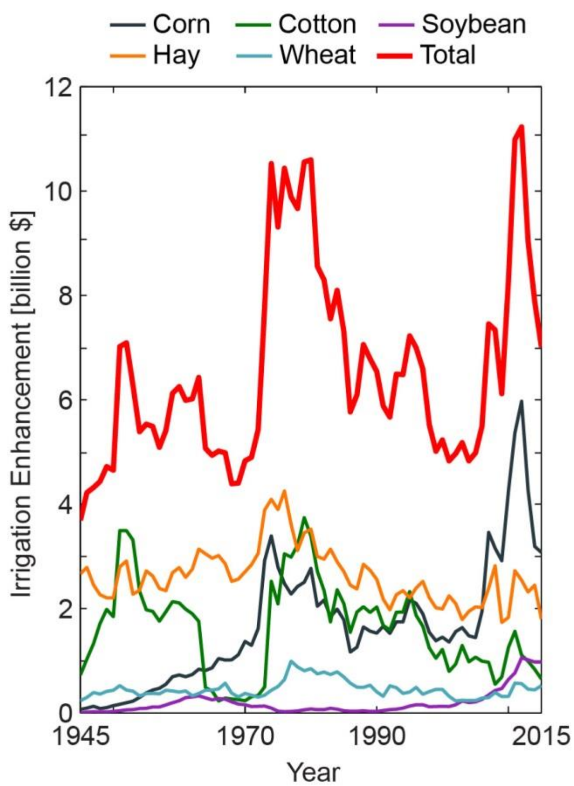

Figure 25. Irrigation enhancement revenue for each commodity across all states with irrigation.

Total revenue from irrigation enhancement has increased since 1945, with large dips in 1968, 1983 and 2001-2006. In 2015, total irrigation enhancement revenue was $7.01 billion, with peaks of $11.2 billion in 2012 and $10.6 billion in 1980. Total revenue patterns after 1970 closely resembled the patterns observed for corn, where irrigation enhancement revenue peaked at $6.0 billion in 2012, and is currently $3.1 billion. Prior to 1970, revenue patterns resembled cotton, where revenue peaked at $3.5 billion in 1951. Wheat has fluctuated the least in enhancement revenue, maintaining an annual value of under $1 billion. Peak wheat revenue was $992 million in 1977 with a low revenue value of $229 million in 2002. Initially, cotton observed the greatest boost in revenue from irrigation enhancement, peaking once at $3.49 billion in 1951, followed by a second peak of $3.74 billion in 1979, but revenue has steadily declined to a value of $648 million in 2015. Hay has largely remained the greatest source of enhancement revenue, ranging from $2-4 billion from 1945-2015. Much of this early revenue is attributed to the predicted inability to grow hay without irrigation in some states, resulting in all wheat production becoming enhancement revenue. Soybean revenue observed a notable decline from 1960-1990, with a low value of $28.3 million in 1989, but has seen a rapid increase in enhancement in recent years, peaking at $1.05 billion in 2012.

Given that revenue is directly correlated to market values, the wide variability seen across commodities can be partially attributed to fluctuations in annual market prices. For example, corn prices peaked in the early 1970's, which correlates with the first major peak in revenue increase. There is a subsequent lag between the early 1970's when commodity prices were highest and irrigation revenue, as farmers continued to irrigate at high production rates throughout the 70's. As commodity prices quickly declined through the 80's and 90's, so did the widespread intensity of irrigation applications. However, other peaks are found outside of the early 70’s (i.e., in times outside of high market prices) indicating that other factors also influence revenue. For example, other peaks may be attributed to technology changes, which lead to increased irrigated yields relative to dryland yields, such as new seed cultivars, or a reduction in the overall variability of crop production during, which ultimately can lead to a more consistent irrigated yield relative to dryland yield (Kucharik and Ramankutty, 2005). It is also likely that after the spike in commodity prices during the 70’s, farmers with irrigation systems were ultimately able to revamp applications once production costs were made lower due to technological advancements. Thus, peaks in irrigation enhancement revenue are most likely due to: 1) an increase in crop price, where any irrigation enhancement is magnified by dollar amounts (e.g., 2 kg x $2/kg = $4 vs. 2 kg x $5/kg = $10) or 2) an increase in yield margin relative to dryland yields (e.g., 2kg x $2/kg = $4 vs. 5kg x $2/kg = $10), which inherently can include changes in climate conditions like droughts.

8. Correlation to Biofuel Demand

Three major trends are notable since 2007: (1) increased irrigation enhancement, (2) increased percent irrigated production, and (3) increased total irrigated area. All three trends are tightly correlated to the recent increase in market demand for corn and soybeans as biofuel crops. This demand increase began in the mid-2000’s and accelerated in 2008 after the introduction of the 2008 US Farm Bill, which encouraged the production of biomass commodities and may also have resulted in improved cultivar technologies. This connection to biofuel demand is further validated by the decline in other commodity areas, while corn and soybean areas increase. Since total irrigated area has also increased, this indicates that farmers are choosing to produce irrigated corn and soybeans over other commodity types. A change in crop choice can be further validated by the recent increases in irrigation enhancement for corn and soybeans, the only two commodities noticeably increasing in revenue. Prior to the mid-2000's, corn and soybean have no apparent patterns between their irrigated areas and enhancement revenues, indicating an intentional selection to irrigate corn and soybeans during the current biofuel-era.

9. Correlation to Drought

The largest increase in irrigation enhancement revenue occurred in 2011-2012, which corresponds to the largest U.S. drought post-1945. Naturally, irrigated yields are much higher than dryland yields during a significant drought, indicating that the rapid peak in irrigation enhancement were due to climatic controls, coupled with the timing of recent increase in biofuel demand. This is an important foretelling observation, as drought intensity and frequency is expected to increase under future climate scenarios (IPCC, 2014; Karl and Trenberth, 2003). 2011-2012 revenue increases are most notable for corn and soybeans, though increases can also be documented in cotton, wheat, and hay. One explanation for why corn and soybeans experienced the greatest spike in enhancement revenue is that a large percentage of dryland areas may also have converted to growing corn and soybeans in regions that do not naturally accommodate these crops to capitalize on the biofuel demand (Smidt et al., 2016). Thus, the difference between irrigated and dryland yields were exacerbated when the already dry conditions for growing corn and soybeans were intensified by a significant drought. Meanwhile, areas of irrigated corn and soybean were able to maintain steady yields.

10. Efficiency Leads to Increased Water Use

Peak irrigation withdrawals at the national-level were highest around 1980, and have since been slowly declining (Maupin et al., 2014). Our model indicates that peak withdrawals in mixed states are currently climbing and not reflective of the national peak. We indicate here that water withdrawals were relatively low among mixed states during the national peak and instead are currently in a period of high pumping based on the recent increase in irrigation dependence and total irrigated area. It is likely that efficient technologies have made irrigation economically feasible in these marginal states, ultimately leading to increased use in areas outside of major aquifer regions.

The concept of increased efficiency leading to increased use and extraction is often referred to as Jevon's paradox (Jevons, 1865). This paradox originated from coal and oil extraction; few have paralleled the trends in coal and oil extraction to the extraction of water for irrigation (e.g., Dumont et al., 2013; Gómez and Gutierrez, 2011). Given that water is often referred to as "the new oil", it is important to validate that the efficiency-extraction relationship holds true in groundwater extraction for farming, just as in mining. Center pivot systems have recently been upgraded with high-efficiency adaptations, such as Low-Energy Precision Applications (LEPA) and Low-Energy Spray Applications (LESA), helping farmers in marginal states expand irrigation practices by lowering production costs (e.g., Figure 6, Smidt et al., 2016). Similarly, the steady increase in irrigation enhancement across all irrigated states, including those that are past peak groundwater extraction, suggests that Jevon's Paradox would remain true in heavily irrigated states if water resources were not limited.

11. Coupled Human and Natural System

One concern for increased irrigation in mixed states is the negative environmental implications that can occur with water-use intensification, in addition to declining groundwater levels. Increased groundwater extraction can cause significant land subsidence, disconnect streams and rivers, require increased energy consumption to pump water from deeper elevations, have notable impacts on regional hydrologic budgets, and significant regional climate effects (e.g., Scanlon et al., 2012; Kustu et al., 2011; DeAngelis et al., 2010; Pei et al., 2016). However, irrigation in mixed states may never have the same environmental implications as other heavily irrigated regions due to site-specific aquifer and climate characteristics. For example, specific yields across mixed state aquifers may not allow for the same intensity of irrigation as seen in heavily irrigated states. Also, more humid climates in mixed regions may be able to tolerate pumping at industrial levels due to rapid recharge, as is the case in the Northern High Plains Aquifer relative to the Central and Southern High Plains Aquifer (Haacker et al., 2015, Breña-Naranjo, 2014). As a result, promoting irrigation in mixed areas that can support high levels of irrigation may be a feasible pathway for meeting future food, fiber, and biofuel demands, as water-stressed regions begin to decline in irrigated production and availability.

12. Conclusion

Comprehensive irrigated and dryland production data is critically lacking prior to 2013, despite the widespread use of irrigation in modern agriculture as early as the 1940's. This study used long-term average statewide yield values to estimate irrigated and dryland yields and areas across the CONUS from 1945-2015. We then quantified the revenue earned from irrigation enhanced-yields along with the overall dependence on irrigated production for states with mixed irrigated and dryland practices.

Based on this analysis, we have four main conclusions:

(1) Irrigation enhanced revenue has totaled over $465 billion since 1945, a considerable amount of revenue solely from the application of water during the growing season. Revenue has consistently increased since 1945, considerably adding to the economic risk associated with intensive irrigation practices. While many irrigated regions are not at risk for groundwater depletion, other major aquifer regions are dangerously water stressed. Any imminent large-scale

decline in irrigated area could result in a considerable economic loss to overall production revenue. Future drought will also have a major impact on the value of irrigation, as evidenced by the peak enhancement revenue during the 2012 drought, which may further increase the dependence and risk associated with artificial water applications.

(2) Mixed states have become more reliant on irrigation for crop production since the turn of the century, meaning the national and global market has also become reliant on irrigation for crop production. This also increases the economic risk associated with any declines in irrigated practices across the CONUS. Dependence on irrigation production is currently at 37% and rising.

(3) Total irrigated area has steadily increased across mixed states over the two decades. Irrigated area increased 36% from just 2002-2014, adding more than 1.4 million irrigated hectares in just a 12-year period. Total agricultural area has remained fairly constant, indicating that dryland areas are being converted to irrigated areas, rather than expanding irrigated agriculture onto new soils.

(4) Crop selection in recent years has mimicked the market demand for biofuel crops, trending toward irrigated biomass production. Since 2006, irrigated corn and soybean areas have increased by 33.7% and 207%, respectively. This is a total addition of more than 3.6 million hectares in just the past decade.

This study highlights the increased dependence on irrigated crop production in states with both dryland and irrigated practices, as well as the increased dependence on irrigation enhancement revenue across all states with irrigation. While irrigation applications can be a significant economic and food security benefit in times of intense weather conditions (e.g., the 2011-2012 U.S. drought), it is important for future policies to consider the economic risk associated with water loss, in addition to food loss. There are several critically stressed aquifer systems (e.g., the High Plains Aquifer and Central Valley aquifer system), and this study indicates that irrigation in marginal states may be able to help reduce the water stress in these regions while also helping to mitigate the economic risk associated with crop production at the national level. As the gap in irrigated crop production between irrigated states and mixed states continues to become smaller, it is important for future policies to consider tailored strategies that are specific to the natural conditions of each state. Tailored strategies will need to include the promotion of dryland strategies in irrigated states, as the risk of economic collapse becomes a more tangible certainty when coupled with the reality of groundwater depletion.

13. Methods

13.1. Data and Processing

All available yield and acreage data for each commodity were downloaded at the state level using the National Agricultural Statistics Survey (NASS-USDA). There are a few states that have recorded irrigated and dryland yields and acreages, though these data are isolated to states above major groundwater aquifers (primarily the High Plains Aquifer states). All available irrigated and dryland yield and acreage data were also downloaded and included in the model construction for each end-member, even if the state with data was a mixed state. Water use data used to define irrigated, mixed, and dryland states was downloaded from the United States Geological Survey for 2010, the most recent year with pumping data (USGS, 2016). All data were processed and modeled using MATLAB version R2010a.

13.2. Conceptual Irrigation Enhancement

An area weighted average was calculated for the dryland state category and smoothed using a locally weighted scatterplot smooth function (lowess) in MATLAB. This function uses a regression weight function for the data points to fit a smoothed surface representative of the data, similar to a polynomial best-fit trend line. Each irrigated state was then smoothed using the same function, normalized by its own 1945 value, and compared to the normalized trend observed across all dryland states. Results are displayed in Figure 20.

13.3. Building Model Predictors

The variables used in the linear model to establish estimation parameters and predict irrigated and dryland yields for mixed states were a combination of the predictor variables that met three criteria: (1) returned the strongest correlation, (2) were individually significant at a 95% confidence interval, and (3) were statistically significant as a group at a 95% confidence interval. We also required predictor variables to return a complete list of predicted values. For example, every state has data available for hay, but less than half of the states have cotton data available. As a result, hay could be used as predictor variable for cotton, but the reverse could not be done. When irrigated and dryland yields from the Agricultural Survey were known for mixed states, they were included as part of the corresponding end-member groups.

13.4. Irrigation Dependence

Actual yield trends for each state and commodity were smoothed using the locally weighted scatterplot smooth function (lowess) as described above. We assumed that no high volume irrigation enhancement was present before 1945, so both predicted irrigated and dryland values were adjusted to the actual 1945 value as a calibrated starting location. If actual yield trends fell below the predicted lower limit (predicted dryland) after 1945, the yield trend was assigned the lower limit value. If actual yield trends rose above the predicted upper limit (predicted irrigated), the yield trend was assigned the upper limit value. The yield trends for each state and commodity were then multiplied by the total area within the state for the corresponding commodity to quantify total production. The predicted upper and lower yield limits were also multiplied by the corresponding areas to quantify the upper and lower limits of production. After using eq. 1, percent irrigated production for each mixed state and commodity was then area weighted and summed to quantify total production dependence on irrigation for mixed states across the CONUS for each year.

13.5. Irrigation Revenue

The total revenue generated by irrigation was calculated using the predicted dryland yields for all states with significant irrigation practices (mixed + irrigated). The predicted dryland yield was subtracted from the actual yield to get the increase due to irrigation. This difference was then multiplied by total area and 2015-adjusted market prices. This process was summed across all states to get cumulative values for each commodity.

13.6. Model Limitations

A limitation to the applicability of this model is that crop yields are widely variable, thus finding several statistically significant predictor variables available at every year and for every state can be difficult. For example, a commodity like sorghum may have great correlation for predicting irrigated corn, but its data are spatially and temporally discontinuous, rendering sorghum unusable as a predictor variable. Furthermore, this study originally included sorghum as a commodity to analyze, but a statistically significant correlation could not be determined using the available predictor variables, rendering sorghum unusable as both a predictor variable and analyzed commodity. This method may be successfully applied to other commodities, but new predictor variables may be necessary for statistically correlated yields across the CONUS.

Acknowledgements

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CHAPTER 5

Natural and Artificial Recharge across the Southern High Plains Aquifer

Abstract

Regional recharge estimates across the Southern High Plains must capture both the complex hydrology and intensive land use due to widespread irrigation applications. Here, we use a fully coupled landscape hydrology model to capture the entire water and energy cycles at the regional scale. This analysis includes simulated recharge estimates under both dryland and irrigation practices for three time periods of interest: (1) averages from 2001-2014, (2) an unusually high precipitation year (2004), and (3) a historic drought (2011). We found that recharge beneath playa lakes had the highest recharge per area, but total recharge throughout interplaya regions was greater than the total recharge beneath the playas. From 2001-2014, the average playa recharge in the irrigation simulation was 149 mm while interplaya recharge was 21 mm. However, interplaya recharge accounted for 71% of the total regional recharge, while playa recharge accounted for the remaining 29%. Playa recharge is notably more consistent than interplaya recharge and is much less impacted by fluctuations in precipitation. We also found that most regional irrigation return flow is very small (less than 1%), but ~15% of irrigated area has significant irrigation return flow (>15%). In particular, interplaya recharge and irrigation return flow are significant in areas that have identified paleochannels or course soils. Results from this simulation can be used as a baseline for water supply across the region, with utility of preparing for climate change or planning for dryland agriculture.

Keywords: Recharge, High Plains Aquifer, Irrigation, Playa, Ogallala, Return Flow

1. Introduction

Agriculture on the Southern High Plains (SHP) is the basis for a substantial portion of its regional economy, averaging nearly $9 billion per year since 1997 (NASS-USDA). However, much of this agricultural production is reliant on groundwater extraction from the underlying SHP Aquifer (SHPA), and recent projections suggest current water use is unsustainable despite conservation strategies (Smidt et al., 2016; Haacker et al., 2015; Scanlon et al., 2012). Total saturated volume of the SHPA has already declined by nearly fifty percent since predevelopment (Haacker et al., 2015), and the average lifespan of the aquifer is projected to be only 81 years under current extraction rates (Scanlon et al., 2012). Further reductions in groundwater levels pose a major threat to the long-term economic stability of the region (Almas et al., 2006). Thus, quantifying groundwater recharge across the region is critical for future groundwater projections and management strategies. However, limited information exists that integrates the complex characteristics necessary to fully quantify areal recharge for the SHPA.

A main challenge to quantifying regional recharge is that perennial streamflow across the SHP is uncommon; instead, nearly 90% of the region is drained to ~30,000 playa lakes, most of which have episodic ponding (Bolen et al., 1989; Nativ, 1992; Scanlon and Goldsmith, 1997; USGS, 2013). Several studies have indicated that almost all SHPA recharge exists as point-source percolation beneath the playa lakes (McMahon et al., 2006; Scanlon and Goldsmith; 1997; Osterkamp and Wood, 1987; Wood and Osterkamp, 1987), but documented recharge rates directly beneath playa lakes are widely variable (Gurdak and Roe, 2010). Physical approaches, water budget calculations, and groundwater models have provided playa recharge estimates between ~0.25 and 219 mm/yr (Nativ and Riggio, 1989; Mullican et al., 1994; Gurdak and Roe, 2010) where studies using chemical approaches have described recharge rates between ~6 and

600 mm/yr (Scanlon and Goldsmith, 1997; Gurdak and Roe, 2010). Part of this variability in recharge estimates is due to the heterogeneous geologic conditions across the playa basins, where site-specific conditions (e.g., clay-rich beds or the presence of large macropores) have resulted in varying flowpaths for recharge. Past researchers have found recharge occurs both along the boundary of a playa basin (known as the annulus) during periods of high ponded water levels (Osterkamp and Wood, 1987; Wood and Osterkamp, 1987; Scanlon and Goldsmith, 1997) and through macropores or cracks directly in a playa bed when water is present in the basin (Gurdak and Roe, 2010; Fryar et al., 2001; Scanlon and Goldsmith, 1997). However, prediction of playa-specific recharge is a challenge since macropore pathways may become clogged due to sedimentation (Tsai et al., 2010; Bolen et al., 1989) and evaporation, episodic ponding, or limited precipitation can result in dry playa basins (Scanlon and Goldsmith, 1997; Bolen et al., 1989; Gurdak and Roe, 2010). Most research about SHP recharge has focused on the playa lake scale; few studies have attempted to quantify the hydrologic fluxes, including those linked to agriculture, at the regional scale (Stovall et al., 2001; Deeds and Jigmond, 2015; Strassberg et al., 2007).

Another challenge to quantifying regional recharge is that the SHPA is a coupled system with the hydrologic and energy fluxes on the land surface (e.g., irrigation). Several regional groundwater models have been developed for the SHP (e.g., Deeds and Jigmond, 2015; Stovall, 2001), but these do not capture the major surface fluxes critical to recharge in a dynamically coupled framework (e.g., climate, soil moisture controls to irrigation). Capturing the coupled water and energy balance for such a system in a regional model is important for two reasons: (1) they play a critical role in the economic production and groundwater supply of the region, and (2) they have profound impact on the hydrology and energy budgets elsewhere across the country. For example, dry soils prompt groundwater extraction for irrigation, and irrigation return flow has been documented to supplement playa lake (McMahon et al., 2011; Fryar et al., 2001), and irrigation applications have been linked to changing precipitation and temperature patterns, both in the region and across the country (DeAngelis et al., 2010; Adegoke et al., 2003; Moore and Rojstaczer, 2001; Pei et al, 2014, 2016). Particularly across the SHP where agricultural production is actively responding to hydrologic and energy cycles, regional models must capture these fluxes to provide realistic estimations of water availability and to help develop effective management strategies. To the best of our knowledge, no regionally-complete studies yet exist that fully couple the water and energy cycles in a process based manner (e.g., evapotranspiration).

The objective of this study was to couple energy and hydrologic cycles at the regional scale to quantify SHP recharge in both playa and interplaya areas. We use the Landscape Hydrology Model (Hyndman et al., 2007; Luscz et al., 2017) to simulate the complete water and energy balance across the entire SHP; this allows us to dynamically estimate recharge across space and time in a manner that incorporates major drivers both on the surface and in the near subsurface. Model simulations span from 2001-2014. Results from this study can be used to supplement future hydrologic models and water management strategies, across the entire High Plains and other regions.

2. Methods

2.1. Study Site

The semi-arid SHPA (~171 km^3; Smidt et al., 2016) covers approximately 90,000 km^2 and includes portions of New Mexico, Texas, and Oklahoma (Figure 26). While it is hydrogeologically connected to the rest of the High Plains aquifer, the vast majority of surface water is not connected to an external surface drainage network, making the region hydrologically distinct. Instead of flow within a developed river network, nearly all surface water flows to playa basins where it is temporally stored as playa lakes. Water within playa lakes is then removed as evaporation or percolation into the subsurface. The average playa lake across the SHP has a lake basin area of 0.05 km^2 and a drainage basin area of ~47 km^2 (USGS, 2013). Clay-rich soils are also very prominent across the region, leading to low permeability and slow percolation. Thus, percolating water often evaporates, leaving behind a layer of leached carbonate minerals referred to as caliche, or calcrete (Reeves Jr., 1970). Caliche acts as a barrier to flow, although fractured or dissolved areas are common (Osterkamp and Wood, 1987; Reeves Jr., 1970). Most recharge is believed to occur as percolation through permeable caliche areas or as leakage beneath playa lakes (Weeks et al., 1988; Dennehy et al., 2002; Osterkamp and Wood, 1987; Scanlon and Goldsmith, 1997; Wood and Osterkamp, 1987).

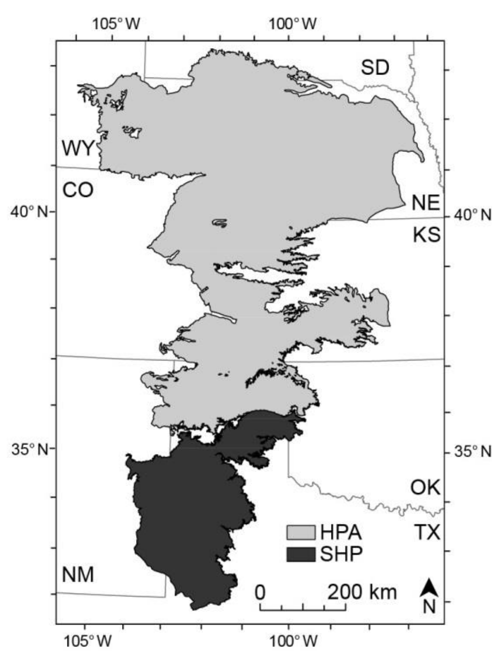

Figure 26. Site map of the Southern High Plains Aquifer.

Agriculture is the dominant land use across the SHP, accounting for 94% of the total land area (NLCD, 2011). Row-crop farming and livestock rangeland are the most common uses, including ~1.2 million hectares of irrigated cropland (Figure 27); irrigated acreage accounts for 32% of all row crops and 13% of the total regional area (Qi et al., 2002). Irrigation across the SHP occurs almost exclusively as groundwater pumping from the High Plains aquifer, and the aquifer is subsequently in decline. Average groundwater levels for the region have dropped 12 m since predevelopment, and total aquifer volume has been reduced by ~50% (Haacker et al., 2015). Trends indicate the average lifespan of the aquifer is ~75 years under current pumping rates, though many areas have already been depleted or have a lifespan of less than 10 years (Scanlon et al., 2012; Haacker et al., 2015; Cotterman et al., in review). Due to its semi-arid climate, irrigation is the major driver of soil moisture, which plays a key role in the regional energy and water cycles (Berg et al., 2014). Irrigation applications often increase with low soil moisture, which in turn leads to increased soil moisture level, increased evaporation, and possible secondary recharge due to irrigation return flow (McMahon et al., 2006; Scanlon et al., 2005). Irrigation also plays a key role in drought management, as artificial water applications help lower the risk of underperforming crop yields. Consequently, agriculture and recharge are inherently linked across the SHP.

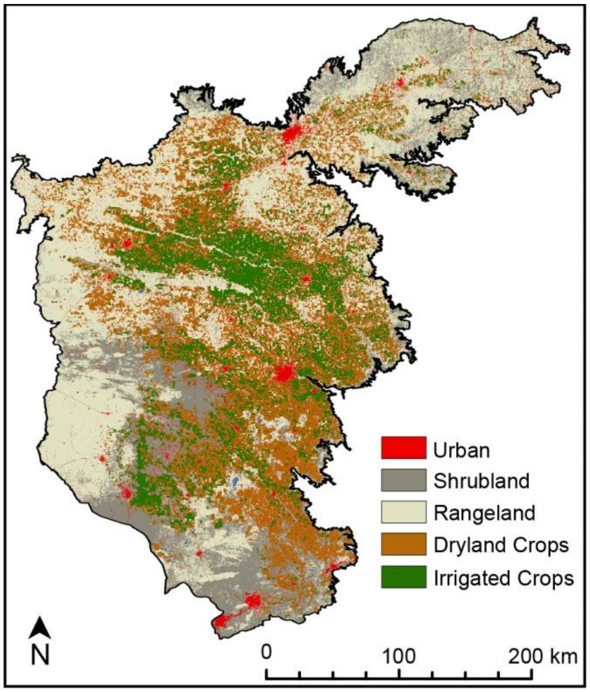

Figure 27. Land cover across the Southern High Plains Aquifer (NLCD, 2011).

2.2. Landscape Hydrology Model

A coupled landscape-hydrology model is necessary to comprehensively address the hydrology of the SHP, including the impact of irrigation and land use on regional water flux. The Landscape Hydrology Model (LHM; Hyndman et al., 2007; Kendall, 2009) simulates the complete water cycle across large regions, including irrigation; it thus provides a framework to quantify recharge and groundwater availability across the SHP. LHM is categorized into four coupled domain modules: (1) surface water routing (e.g., infiltration, precipitation, and evaporation), (2) canopy and root zone moisture (e.g., transpiration, leaf area index, and soil moisture), (3) deep unsaturated zone (e.g., deep percolation and throughflow), and (4) saturated groundwater via a linkage to MODFLOW (Harbaugh et al., 2017). Within each domain module are a suite of process-based codes that allow for a fluid mass balance based on the processes that drive water movement within each module boundary. Here, we focus on the shallow fluxes (modules 1-3) across the SHP and do not use saturated groundwater (module 4) in the analysis.

First, LHM was fed widely available gridded input data [precipitation (NOAA-CDO, NLDAS, 2012), air temperature (NOAA-CDO, NLDAS, 2012), land cover (NLCD, 2001), LAI (MODIS), soils (SSURGO-NRCS), elevation (NED-USGS, 2009), wetlands (NWIS-USGS), and streams (NHD-USGS)]. These data were then grouped into LHM-specific modules [surface water, soil moisture, potential evapotranspiration, canopy, and snowpack] and then routed through coupled mass- and energy-balance flow component modules [evapotranspiration, runoff, throughflow, deep percolation, recharge, and streamflow]. Module outputs were then grouped into additional LHM-specific modules [saturated zone and unsaturated zone] and processed for final regional outputs [recharge and streamflow]. Here, we isolate all cells into two categories: upland (interplaya) and wetland (playa). Upland cells are essentially those without ponded water and are treated as unsaturated soils. Wetland cells are those with ponded water and are defined by the hydraulic conductivity of the underlying soil and evaporation of the ponded water (refer to the next section for more details). All data were processed using Matlab R2010a, ArcGis 10.2.2, and Python 2.6. More detailed information on LHM construction and function can be found in Hyndman et al. (2007) and Kendall (2009).

Here, we used hourly timesteps for our simulations over the 2001 - 2014 period. The scope of the model boundary included the entire SHP, with 500m x 500m cells (~1 million active cells). Complete model runs typically took ~4 days, or ~7 hours per model year on a typical workstation (3 GHz, quad-core processor). Two scenarios were evaluated: (1) no irrigation applications (acting as a baseline), (2) irrigation applied based on a soil moisture threshold (representing current land practices). We analyze recharge in this study using cell populations as model outputs were spatially gridded raster files. We report cell count data in the form of cumulative distribution functions (CDFs). Results are reported in both standard scale (to interpret the upper order end members) and x-axis semi-log scale (to interpret the lower order end members). One modification within the standard LHM module code specific to this study was the simulation of playa lake drainage, as discussed in the next section.

2.3 Playa Routing

Nearly 30,000 playa lakes across the SHP have been processed into a geospatial layer as part of the National Hydrography Dataset by the USGS (USGS, 2013). Playas are mostly located in the central and eastern portions of the boundary, as shown in Figure 28 (Panel A), and their size and distribution is variable across the landscape (Panel B). SHP playa locations also heavily overlap the regions of irrigated cropland. Most hydrologic models treat playas as stationary bodies and do not accurately capture regional water fluxes. In reality, water storage in playa lakes is often ephemeral and not predictable in space or time (i.e., knowing when, where, and how much water will enter a playa lake is largely unpredictable). We used the central locations of playa lakes and the regional elevation model (NED-USGS, 2009) to construct drainage basins for each playa lake basin. Instead of treating playa lakes as stationary points, we allowed playa lakes to drain and fill independently by routing excess flow to playa drainage basins only when the playa lake basin was occupied with ponded water (i.e., runoff accumulation). All internally-drained basins were identified, and split into two subsets: those with a mapped playa according to the National Hydrography Dataset, and those without. For playa basins, all flow from the internal drainage was routed to the playa and split amongst its cells. For those without playas, surface runoff was routed to sink cells, but then treated as upland type, rather than as a wetland.

If a pre-existing lake was not present in a drainage basin, water was routed across the landscape similar to interplaya regions. Once ponded in a playa, infiltration was time-limited by the hydraulic conductivity of the underlying soil and the depth of ponded water. Ponded water was either infiltrated or lost to evapotranspiration, so the total amount in each such play was directly related to the duration of ponding. For example, more water was lost to evapotranspiration during a wet year than dry year, as more water was left ponded due to the limiting hydraulic conductivity of the underlying soil. It is also important to note that LHM does not simulate complex playa processes such as infiltration through the annulus of the playa bed (e.g., Gurdak and Roe, 2010). In conceptual terms, playa water was treated as a perched lake, where infiltration and evapotranspiration were a function of hydraulic conductivity and depth of ponding.

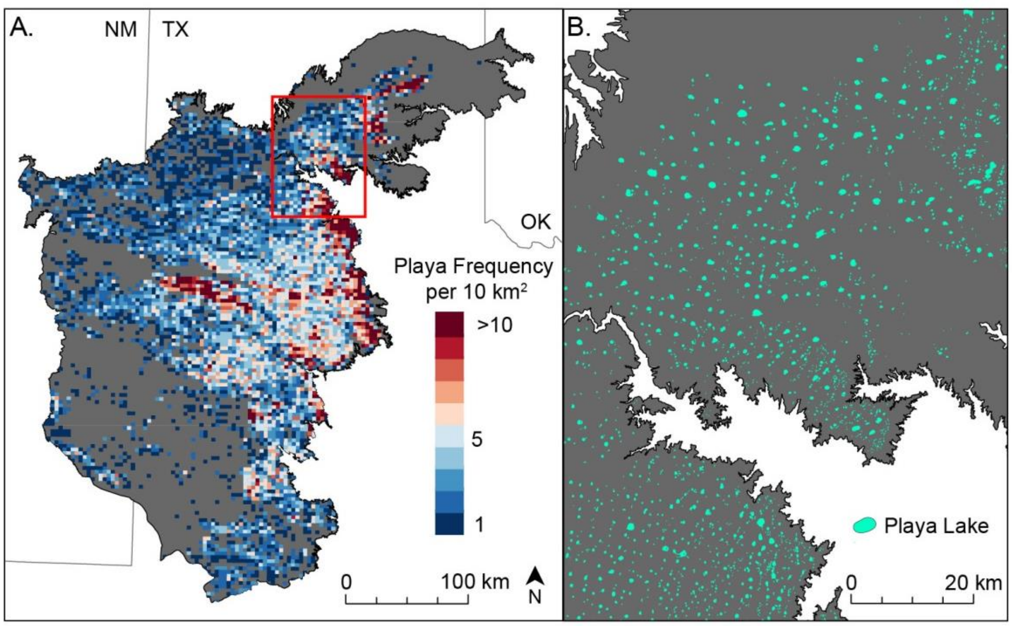

Figure 28. Playa lake density across the SHP per 10 km^2 cells (Panel A) and sample playa lake distribution (Panel B; USGS, 2013).

2.4 Irrigation and Precipitation

Irrigation is common across the SHP and is deeply engrained into the regional economy, thus making it a critical component to incorporate in regional recharge estimates. We defined the growing season as mid-March through mid-October (days 133-290) to define the transpiration period in these areas, where the start date indicated when root mass would start to develop and the end data indicated when crops were no longer present. The locations of irrigated areas derived from satellite imagery were downloaded from Qi et al., 2002. Irrigation scheduling within LHM was triggered if four criteria were satisfied: (1) the leaf area index (LAI) of irrigated cropland was greater than a defined LAI threshold (0.0005 m^2 leaf/m^2 ground), (2) the time since the last irrigation event is sufficient to warrant a new event, (3) when soil moisture in the first 3 layers dropped below 0.33 of plant available water (PAW = field capacity – wilting point), and (4) if accumulated evapotranspiration is above a defined threshold. Irrigation events were assigned standard center pivot pumping rates of 599 gpm (2,265 L/s), and LHM assumes that an entire cell was irrigated. In reality, partial cells are irrigated. In response, a threshold for defining entirely irrigated cells was assigned using the total irrigated acreage. Threshold construction first sorted all cells with irrigation, including partially irrigated cells, from most irrigated area to least. Cells with the greatest irrigated area were then converted to entirely irrigated until the total area of irrigation matched the total irrigated area prior to cell conversion. The result of this process converted cells with at least 48.1% irrigated area to entirely irrigated. Cells with less irrigated area were converted to dryland. Once these cells were assigned, irrigated areas were stationary in space and time throughout the simulations, as there is not sufficient information available to determine irrigation locations through time.

Average precipitation across the SHP ranged from 200 mm in the west to 700 mm in the east (Figure 29, Panel A). However, regional precipitation in 2004 was much higher than average and 2011 much lower. We use these two years to evaluate changes that likely occur in high and low precipitation years. High precipitation in 2004 ranged from 500 mm in the west to 1000 mm in the central and eastern portions, ~300 mm increase from the average across the region (Figure 29, Panel B). There was a historical drought in 2011, when annual precipitation dropped to just ~100 mm across most of the region (Figure 29, Panel C). We isolate three precipitation levels as focus periods for the remainder of this study: (1) annual average, where each year is individually simulated and then averaged together (2) 2004, acting as a proxy for high precipitation years, and (3) 2011, acting as a proxy for low precipitation years.

We compare two simulations for each of these precipitation focus periods, one with the irrigation module enabled and one with no irrigation (baseline). The baseline scenario represents natural conditions, where the precipitation received is the only source of water added to the landscape. During the irrigation simulation, artificial water applications are added to the irrigated cells to simulate crop production strategies across the region. In both simulations, all other processes (e.g., evapotranspiration) perform consistently, although estimates between each simulation can vary due to the addition of water to the landscape during the irrigated simulation.

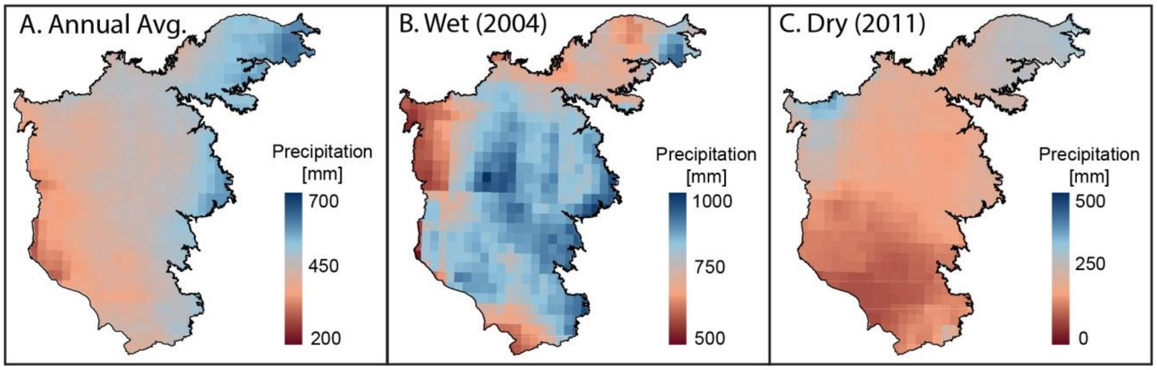

Figure 29. Annual precipitation across the SHP.

3. Results

3.1. Baseline Recharge with No Irrigation

Recharge results for the baseline simulation are displayed in Figure 30 as a CDF of interplaya (only cells outside of playa lakes) and playa cells (only cells with playa lakes). When comparing interplaya and playa regions, average recharge was 20 mm and 53 mm, respectively. Playa basins had the greatest concentration of recharge, where 10% of cells generated at least 100 mm of annual recharge, compared to 5% of interplaya cells. Interplaya regions followed a similar recharge curve compared to playa basins, though cells with the highest recharge remained at lower values than in playa basins, and lower recharge values were more frequent in all cells when compared to playa cells; 66% of cells received less than 10 mm of recharge in playa basins and 68% in interplaya basins.

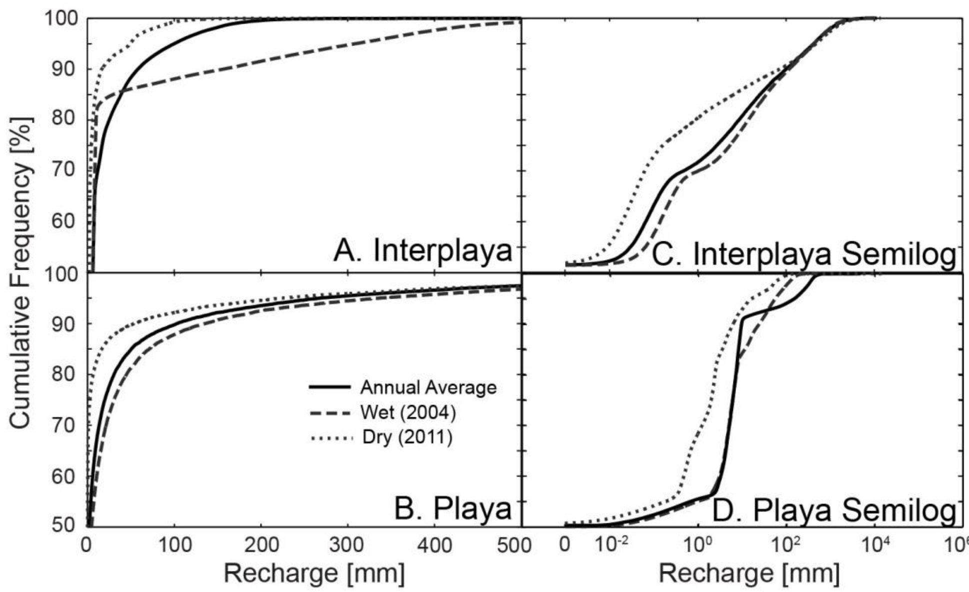

Figure 30. CDF of recharge for interplaya and playa cells during wet, dry, and average years.

Changes in recharge in response to precipitation fluctuations are much larger in interplaya regions than in playa basins. In 2004 (wet year), recharge in playa basins was only slightly greater than an average year, despite the large increase in regional precipitation. However, recharge in interplaya regions was significantly larger, particularly in the top 15% of cells. Here, recharge largely deviated from the average and increased to 156 mm for the top 10% of cells and 308 mm for the top 5%. The average recharge in interplaya regions during 2004 increased to 42 mm, where average recharge in 2004 only increased to 65 mm in playa basins.

A similar, but declining, pattern was observed during the 2011 drought. Interplaya regions fluctuated the most compared to average recharge, where playa recharge trends remained fairly stable compared to average recharge. In both regions, average recharge dropped to 8 mm in interplaya regions and 44 mm in the playa basin. During 2011, 95% of cells received less than 50 mm of recharge in interplaya regions and 90% in playa basins. On average the wet year (2004) had 39% and 138% more recharge in playa basins and interplaya regions respectively compared to the dry year (2011); this indicates a strong correlation between precipitation and recharge for areas outside of playa basins, and consistent recharge for playa regions despite precipitation changes.

3.2. Irrigation and Recharge

Irrigation demand is driven by soil moisture, indicating that there greater demand for irrigation in dry versus wet years. Our simulated estimates for irrigation are displayed in Figure 31. Annual simulated irrigation was 420 mm in average precipitation years (Figure 31, Panel A); 380 mm in a wet year (Figure 31, Panel B) and 515 mm in a drought year (Figure 31, Panel C). The most notable difference between the three periods is the widespread intensification of irrigation in 2011. Here, not only did the magnitude of applied water increase, but the regional intensity also increased in areas that are significantly less irrigated during an average year. In particular, the eastern portion of the study region drastically increased in irrigation applications, and the central band of irrigated acreage also intensified its applications. This central band was still evident in 2004, but across the region, irrigation demand and application significantly declined during the above-normal precipitation regime.

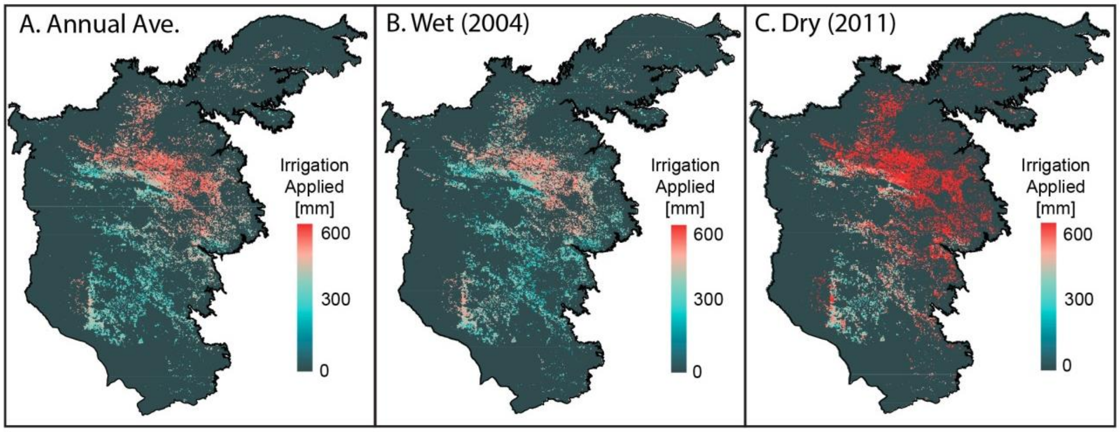

Figure 31. Simulated irrigation estimates across the SHP a) averaged across individually simulated years from 2001-2014, b) in 2004 which was wet year, and c) in 2011 – a drought year.

Average annual recharge across the region from 2001-2014 was 149 mm for playa regions and 21 mm for interplaya regions (Figure 32, Panel A); annual recharge was 146 mm for playa regions and 45 mm for interplaya regions during 2004 (Figure 32, Panel B), and 170 mm for playa basins during 2011 and 8 mm for interplaya regions (Figure 32, Panel C). Noteworthy observations are the large bands of interplaya recharge in the central and southwestern portions of the study region. While still less than the simulated recharge in many playa basins, average annual recharge many of these interplaya cells was ~300 mm/year. The inset isolating the central band highlights moderate interplaya recharge adjacent to playa recharge cells (Figure 32, Panel A). In this inset, points of playa recharge were nearly double interplaya recharge, although significant interplaya recharge did still occur. Interplaya recharge also occurred in the northeastern portion of the study area in similar magnitude to the interplaya recharge found in the central and southeastern portions of the boundary.

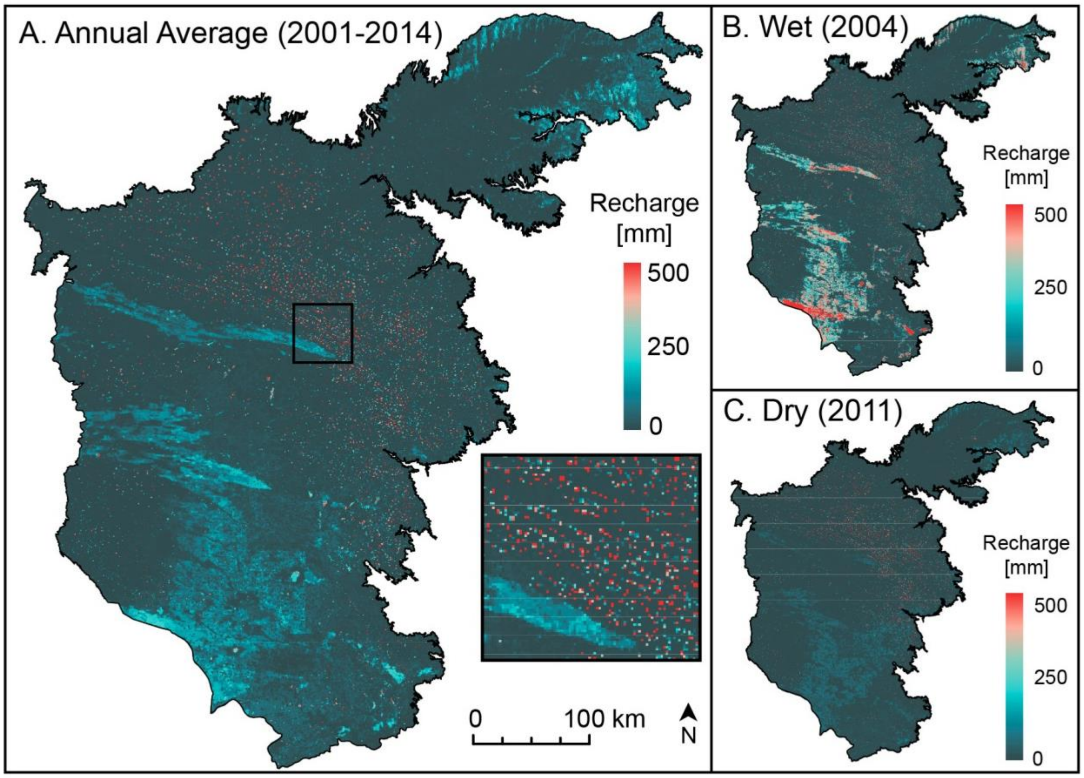

Figure 32. Simulated recharge estimates with activated irrigation for the annual average, 2004, and 2011 across the SHP.

It is important to note that even though interplaya regions have much less recharge than concentrated playa basins, the large area of interplaya regions ultimately sums to a large percentage of average annual recharge budget compared to highly concentrated playa basins. Our results indicate that ~71% of all regional recharge occurs as percolation through interplaya areas, with the remaining 29% occuring as percolation through playa basins. Playa cells account for less than 6% of the entire boundary, yet they account for nearly one-third of all recharge. Playas are a significant contributor to recharge, but our simulations indicate that interplaya regions likely have a larger role in regional recharge than currently understood.

Another notable observation is the drastic variability in interplaya regions again in the central and southwestern bands (Figure 32, Panel B). In 2004, recharge due to high precipitation was similar to the recharge simulated in playa basins for average years, and was as high as 500 mm in some portions of the bands. Despite large amounts of precipitation, recharge beneath playa lakes largely remained consistent with the annual average. The dry year indicated almost no interplaya recharge, while playa recharge remained consistent (Figure 32, Panel C). Interplaya recharge in the central bands declined from an average of ~300 mm to ~100 mm, and in many areas with already minimal recharge, declined to 0 mm. A CDF of playa recharge cells for the irrigated simulation are displayed in Figure 33; interplaya recharge cells were not significantly altered by irrigation, and therefore were not statistically different than interplaya recharge during the baseline simulation.

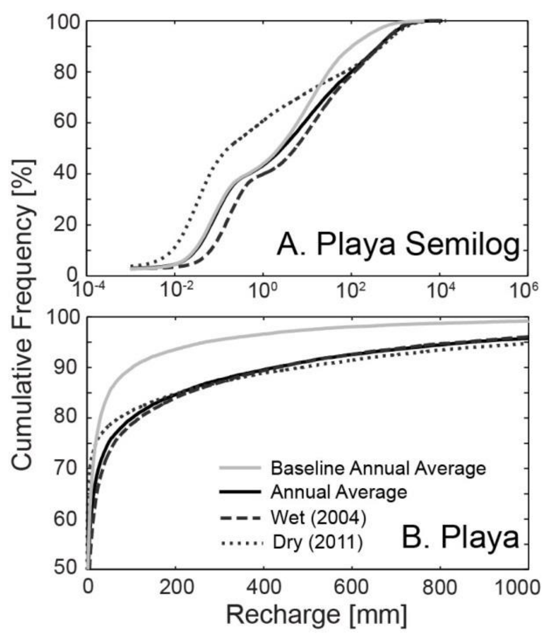

Figure 33. CDF of recharge across the SHP when irrigation is included in the simulation.

Playa recharge followed a similar curve as the baseline simulation (Figure 30); only here a greater percentage of cells generated a larger magnitude of recharge. A much larger percentage of cells generated higher recharge, with 13% of cells were > 300 mm. Playa recharge during the irrigation simulation for the three focus periods (i.e., annual average, 2004, and 2011) was most variable in lower recharge zones and was consistent in zones of higher recharge. The number of cells that generated more than 300 mm of recharge was almost identical for the three periods. However, less recharge was generated in 2011 compared to 2004 in cells with less than 300 mm of recharge. Also displayed in Figure 33 is the baseline annual average. Compared to the irrigation simulation, less average recharge occurred during the baseline simulation, as evidenced by a greater percentage of cells generating lower recharge. Thus, irrigation had a notable impact on recharge within playa basins, partially due to irrigation return flow.

3.3. Irrigation Return Flow

Excess irrigation application that returns as recharge is referred to as return flow and is defined using equation 1.

$$Return\ Flow = \frac{Irrigated\ Recharge - Baseline\ Recharge}{Irrigation\ Applied} \qquad \text{(eq. 1)}$$

Areas of estimated return flow are highlighted in Figure 34. Across irrigated cells in the SHP, more than 81% of the average return flow from 2001-2014 was less than 1% of the applied water and 77% had nearly no return flow (Figure 34). However, irrigated return flow was significant in some portions of the study region, particularly in the southwestern region of the boundary (Figure 34). Here, return flow values were mostly around 15% with some cells reaching as much as 30% return. Irrigation return flow in the main central band of irrigated cropland was mostly less than 1%, with the exception of isolated areas within playa basins. This central band corresponds to a high density of both playa lakes and irrigated cropland, indicating that irrigation runoff to playa basins may be significant, as evidenced by the high return flow values inside playa basins. The greatest return flow percentages, which occur in the southwestern and central portions, directly correspond to areas with highest interplaya recharge. This relationship is accentuated during 2004, when notable return flow occurs at a broader spatial scale across interplaya regions, and return flow is almost entirely eliminated in these regions during 2011. Regardless of precipitation, return flow in playa basins remained fairly stable throughout each of the focus periods, further indicating a connection between irrigated runoff and playa storage.

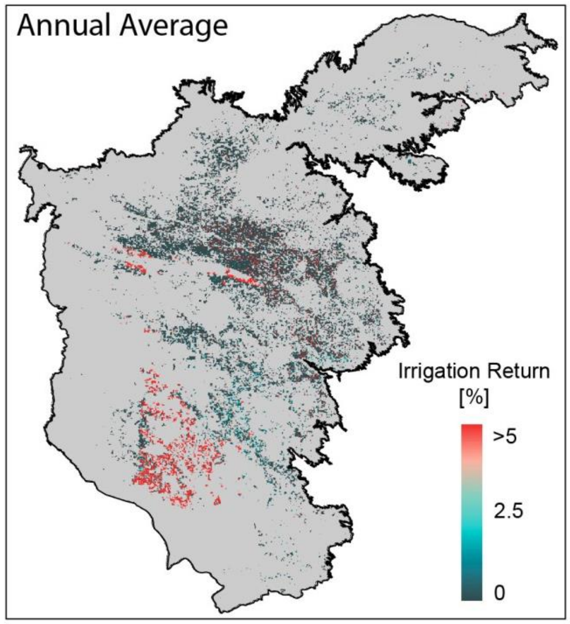

Figure 34. Estimated locations of irrigated return flow across the SHP.

A CDF for each focus period is displayed in Figure 35. For each period, notable irrigation return flow only occurs in ~20% of irrigated cells. Irrigated return flow trends mimic interplaya recharge trends, where a wide variance is observed between 2004 and 2011. Irrigated return flow was greatest in 2004, where 10% of cells had more than 38% returned. In 2011, 86% of cells were below 1% return flow. Average return flow across all irrigated cells is 9% for the annual average, 12% for 2004, and 8% for 2011.

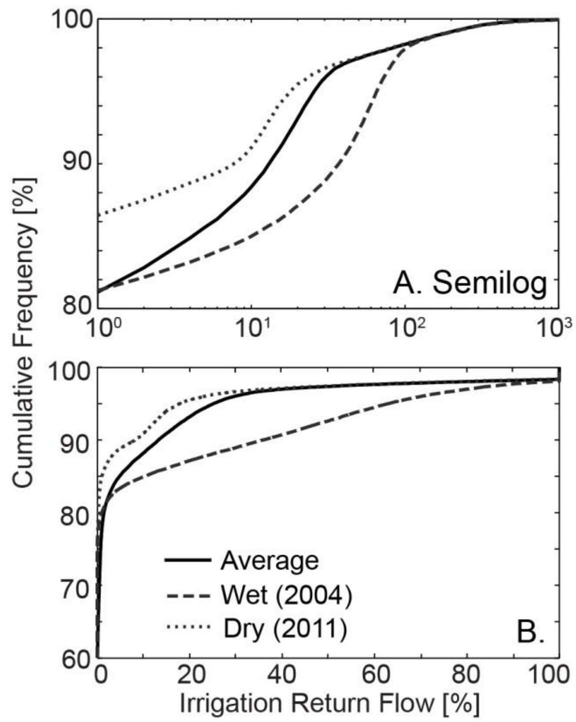

Figure 35. CDF of irrigation return flow across the SHP.

3.4. Validation

We validated our model using a mass-balance approach solving for the annual change in groundwater storage. We first solved for annual change in water storage using simulated water level data from Haacker et al., 2015 (equation 2).

$$\Delta Storage = \Delta Water\ Level * Specific\ Yield \quad \text{(eq. 2)}$$

We then used our simulated recharge to solve equation 3,

$$\Delta Storage = Recharge - Pumping - Groundwater\ Discharge \quad \text{(eq. 3)}$$

where “recharge” was assigned the regional recharge during the irrigated simulation, “pumping” was defined as the applied irrigation at 95% efficiency (i.e., we assumed a 5% reduction in applied water due to wind loss), and we assumed that groundwater “discharge” (i.e. flow out of the region) was negligible (after Blandford et al., 2003). Equations 2 and 3 were then rewritten as equation 4,

$$(Recharge - Pumping - Groundwater\ Discharge) - (\Delta Water\ Level * Specific\ Yield) = 0 \qquad \text{(eq. 4)}$$

and the annual average regional results of this equation are displayed in Figure 36, Panel A. The majority of the region has a predicted water storage difference of less than 50 mm, with the most notable difference occurring in the major irrigated areas where water storage differences are more than 150 mm. Here, the negative value indicates that our model has over-predicted the amount of change in water storage (i.e., estimated too large of reduction in storage), due an overestimation of total pumping. This is to be expected for two reasons: (1) we originally converted partially irrigated areas to entirely irrigated cells to fit within our model limitations, and (2) our metrics for irrigation include ideal efficiency. For example, our simulation starts irrigation exactly at a predictable soil moisture lower threshold and turns off irrigation at an upper moisture threshold. Farmers do not have this level of precision and often react to dry soil conditions, where our model would have already been irrigating. Our model also does not have any consideration for the production cost or policy-regulated limitations to irrigation. In reality, farmers may choose to irrigate less due to the cost or policies governing water use. Given that (1) our simulation is idealized in irrigation conditions, (2) real irrigation applications are often inefficient (Smidt et al., 2016), and (3) we do not simulate partially irrigated cells, the differences between our simulated change in water storage and the validated set are minor enough to establish confidence in our model results.

We also validated our pumping estimates against those reported by the USGS in 2005 and 2010 (USGS, 2016). The USGS has groundwater pumping data at the county level, so our pumping estimates were averaged within each county boundary to generate a county-wide pumping value. The percent difference between our estimates and the data reported by the USGS are displayed in Figure 36, Panel B. Here, a positive difference indicates an overestimation of pumping by our model, which further supports our validation in Panel A. For the majority of SHP counties, our estimations were within 50% of the observed pumping data. The locations of our estimates also mirror the over- and under-estimated areas in Panel A. Again, the differences quantified here are minor enough to establish confidence in our model results, while noting that our pumping system represents optimal conditions and generates more irrigated applications that are observed across the SHP due to the conversion of partially irrigated areas to entirely irrigated cells.

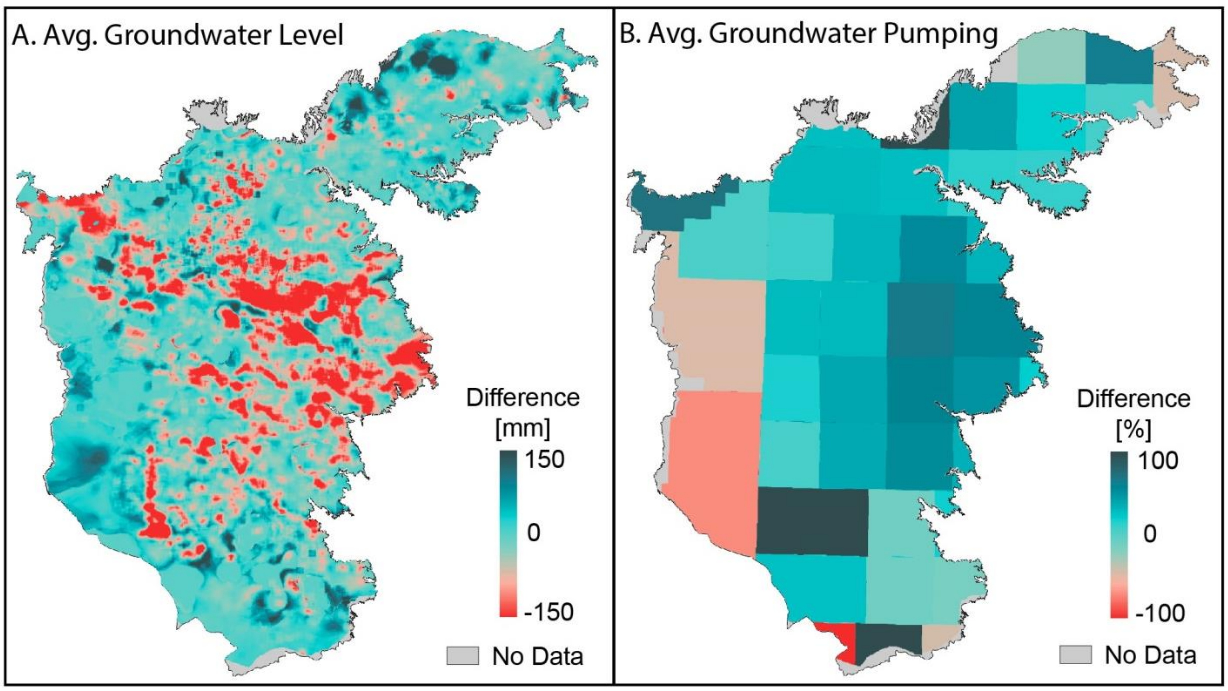

Figure 36. Percent differences in water level using a mass-balance approach (Panel A) and simulated versus observed percent differences for groundwater pumping (Panel B).

4. Discussion

In addition to the strong validation, our recharge estimates within playa lakes fell well within the limits measured by other researchers, as outlined in the introduction (Scanlon et al., 2007; Gurdak and Roe, 2010), further validating our overall recharge estimates. However, our interplaya observations add to this field by highlighting areas of significant recharge. Other researchers have indicated that interplaya recharge is negligible, often citing chloride concentrations in the unsaturated zone (Scanlon et al., 1997; 2007; Blandford et al., 2003), but these chloride concentrations were analyzed in areas also predicted as negligible by our model (Figure 37). For example, our average annual interplaya recharge values in the locations of these chloride studies is 6 mm, with a maximum value of only 23 mm and a minimum of 0.02 mm. Based on our model, interplaya recharge is not negligible when averaged across the region and is significant in areas of coarse-soils or paleochannels. Further site-specific analyses should be conducted in regions predicted as high recharge by our model.

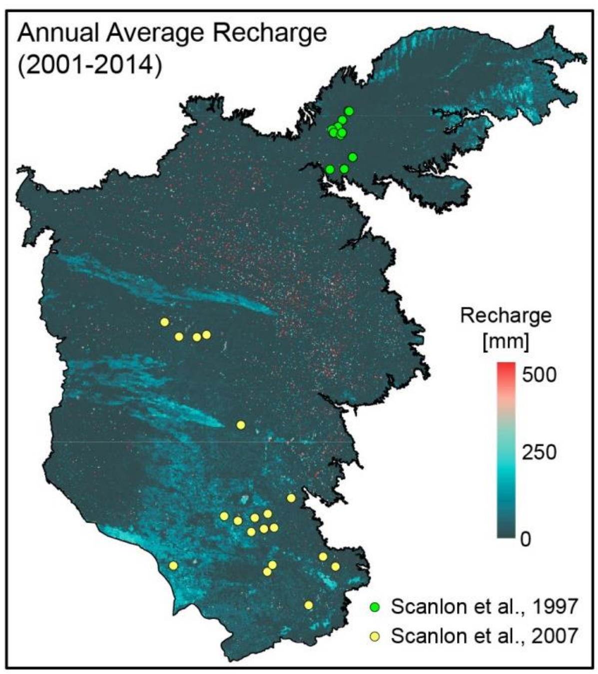

Figure 37. Average annual recharge with locations of field-based experiments quantifying interplaya recharge across the Southern High Plains Aquifer.

Analysis of the 2011 drought can be used as precursor for future climate change scenarios. Temperatures for the SHP region are predicted to increase by 2 to 5°C, and droughts are expected to become more intense and frequent (IPCC, 2007). Simulated recharge significantly declined during 2011, which would further add stress to an already water-stressed region. Estimates from 2011 can be used as a baseline for the amount of recharge to be expected during drought years. Our model also demonstrated that regional irrigation applications drastically increased in response to drought, intensifying groundwater pumping and ultimately leading to increased groundwater decline. This is particularly evident in the central band across the region, where irrigation is the highest and groundwater levels have declined the most (Haacker et al., 2015). Our model can also help estimate irrigation demand, or total water extraction, during periods of low precipitation.

A significant observation in this study is that playa recharge in the central band remains consistently high regardless of precipitation during both the baseline and irrigation simulation. This may be due to factors such as: (1) irrigation return flow or (2) the function residence time and depth of ponded water in a playa basin. Significant irrigation return flow runs off to playa basins, validating the influx of water to playa basins during the dry year. However, the steady simulated playa recharge indicates that the rate at which water is able to percolate beneath playa basins is the controlling factor. Percolation through playa basins thus can only occur at a specific rate, so extra water included in the basin does not drastically increase recharge, as long as ponded water exists in the basin. It is also possible that the high soil moisture beneath the playa lake acts as a moisture buffer for subsequent precipitation events. It is possible that while recharge may not be contributing as recharge, it may be acting as a soil moisture buffer for subsequent precipitation events. If soil moisture is regulated and kept high by irrigation runoff, then less precipitation is lost to matric potential of dry soils. This relationship may allow for a higher percentage of precipitation to percolate below the root and evaporative zones, ultimately becoming recharge to the aquifer.

One limitation to this work is that a uniform irrigation scheduling is applied across the region based on soil moisture and site-specific drivers to irrigation are not captured. While this is appropriate at the large scale and neutralized through the county-level validation, this does not adequately address localized controls such as management district regulations or priority rights. Future analyses focused on small-scale regions will need to capture the localized controls to water use. A second limitation is that the baseline model without irrigation assumes current agricultural land cover, despite the elimination of the artificial water applications to support current land cover. In a future scenario where irrigation may not be widely available, land cover will subsequently be converted to an economically viable dryland commodity or land practice. While this model is effective for current land practices, land cover changes will need altered to simulate scenarios indicative of future dryland regimes. A third limitation is that LHM does not simulate caliche, which is likely an inhibitor to interplaya recharge. It is likely that caliche layers have significant fractures, dissolution paths, or macropores that allow for percolation. Yet, in some areas across the SHP, this may be an important factor to add to the simulations.

5. Conclusion

Despite the economic significance of irrigation water use on the SHP, regional recharge estimates across the region are limited and often too simplified to capture the complex hydrology or the intensive land use across the region. This study overcomes these challenges by applying a fully-coupled landscape hydrology model to capture both the hydrology and the land use-land cover for 2001-2014. Simulations were done both (1) with irrigation and (2) without irrigation, over three focus periods, (a) annual average based on individually simulated years from 2001-2014 , (b) 2004, serving as an abnormally wet year, and (c) 2011, serving as an extremely dry year. Based on the results of this study, we conclude that:

1) approximately 149 mm of annual recharge occurred through playa basins and 21 mm occurred throughout interplaya basins. The magnitude of recharge in interplaya areas is a significant contributor to the overall water budget of the region, accounting for 71% of all recharge. This is a significant contribution to the literature as interplaya recharge has been often considered negligible. Playa recharge accounts for 29% of all recharge, while only occupying 6% of the land surface.

2) Regional recharge in interplaya areas responded directly to fluctuations in precipitation, particularly in areas of high interplaya recharge. On the other hand, regional recharge in playa basins remained relatively steady regardless of annual precipitation, likely due to rate-limiting infiltration. Based on the results, there are significant areas of interplaya recharge in areas of coarse soil, often referred to as paleochannels.

3) Simulated irrigation return flow across the SHP was an annual average of 8% across irrigated cells, although 81% of irrigated area had simulated return flow of less than 1%. Irrigation return flow did correlate with fluctuations in precipitation, increasing in wet years and decreasing with in dry years. This is likely due to the increased soil moisture acting as a buffer and allowing for more irrigation runoff to enter playa basins without becoming trapped by matric potentials in dry soils.

4) Total regional recharge increased when irrigation was included in the simulation compared to a baseline dryland scenario. However, irrigation return flow remained fairly constant between both simulations, indicating that irrigation excess is not the main cause of

change in recharge. Instead, increased soil moisture due to irrigation applications may act as a critical buffer to recharge, allowing for precipitation to percolate to greater depths than when interacting with dry soils.

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