Underutilized Spaces and Marginal Lands for Sustainable Land Use:

A Multi-Scale Analysis

by

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ABSTRACT

Drawn from a trio of manuscripts, this dissertation evaluates the sustainability contributions and implications of deploying underutilized spaces for alternative uses at multiple scales: urban, regional and continental. The first paper considers the use of underutilized spaces at the urban scale for urban agriculture (UA) to meet local sustainability goals in Phoenix, Arizona. Through a data-driven analysis, it demonstrates UA can meet 90% of annual demand for fresh produce, supply local produce in all food deserts, reduce areas underserved by public parks by 60%, and displace >50,000 tons of carbon-dioxide emissions from buildings.

The second paper considers marginal agricultural land use for bioenergy crop cultivation to meet future liquid fuels demand from cellulosic biofuels sustainably and profitably. At a wholesale fuel price of $4 gallons-of-gasoline-equivalent, 30 to 90.7 billion gallons of cellulosic biofuels can be supplied by converting 22 to 79.3 million hectares of marginal lands in the Eastern United States (U.S.). Displacing marginal croplands (9.4-13.7 million hectares) reduces stress on water resources by preserving soil moisture. This displacement is comparable to existing land use for first-generation biofuels, limiting food supply impacts. Coupled modeling reveals positive hydroclimate feedback on bioenergy crop yields that moderates the land footprint.

The third paper examines the sustainability implications of expanding use of marginal lands for corn cultivation in the Western Corn Belt, a commercially important and environmentally sensitive U.S. region. Corn cultivation on lower quality lands, which tend to overlap with marginal agricultural lands, is shown to be nearly three times more

sensitive to changes in crop prices. Therefore, corn cultivation disproportionately expanded into these lands following price spikes.

Underutilized spaces can contribute towards sustainability at small and large scales in a complementary fashion. While supplying fresh produce locally and delivering other benefits in terms of energy use and public health, UA can also reduce pressures on croplands and complement non-urban food production. This complementarity can help diversify agricultural land use for meeting other goals, like supplying biofuels. However, understanding the role of market forces and economic linkages is critical to anticipate any unintended consequences due to such re-organization of land use.

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I dedicate this dissertation to baby Henry.

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CHAPTER 1

INTRODUCTION

1.1 Land use and sustainability challenges

People currently use about a quarter of the land-based net primary production for food, feed, fiber, timber and energy (Krausmann et al 2013). Land also provides biodiversity and many other essential ecosystem services, like climate regulation.

Land use / land cover change is one of the most pervasive stressors of the Earth's physical resources and natural systems (Turner II et al 2007). Deforestation, urbanization, and the expansion and intensification of agriculture have altered the planet's surface. As a result of human activities, over three quarters of the Earth's ice-free land area can no longer be considered wild (Ellis and Ramankutty 2008), and even remaining wild areas are likely influenced by human beings (Sanderson et al 2002). Land use / land cover change is also linked to almost all major environmental problems, like land degradation, water pollution and habitat destruction, as well as climate change, because land acts as a source as well as a sink for greenhouse gas (GHG) emissions.

For example, soils in croplands are estimated to be lost at a much faster rate than they are formed (10 to 20 times under no-till systems to more than 100 times under conventional tillage) (Montgomery 2007; IPCC 2019). Extensive and persistent seasonal hypoxia in the Gulf of Mexico has been attributed to intensification of agriculture in the U.S. Midwest (Donner and Kucharik 2008 and Van Meter et al 2018), and existence of similar "dead zones" have been documented in coastal waters worldwide (Diaz and Rosenberg, 2008). Faster rates of urbanization near protected areas fragment wildlife corridors, reducing the effectiveness of efforts to prevent habitat loss (Radeloff et al 2010). During

2007-2016, an estimated 23% of total anthropogenic greenhouse gas (GHG) emissions were attributable to Agriculture, Forestry and Other Land Use (AFOLU) (IPCC 2019). Agriculture is the single biggest transformer of land. Land use / land cover change and land use intensification associated with agriculture remains an important source of economic growth and development. Worldwide, the total production of cereal crops rose by 240% (1961-2017) and cotton output rose by 130% (1961-2013) due to expansion of croplands and yield improvements, primarily as a result of increased use of agro-chemicals (IPCC 2019). Currently, more than one-third of global land use is for agricultural purposes (FAOStat 2015): cropping (12%) and permanent pasture (25%). For the majority of the world's countries (78%), the share of agriculture in total land area is more than 14%, and for a quarter, it is at least 50% (Figure 1.1). Irrigated agriculture is also responsible for nearly 70% of freshwater consumption worldwide (IPCC 2019).

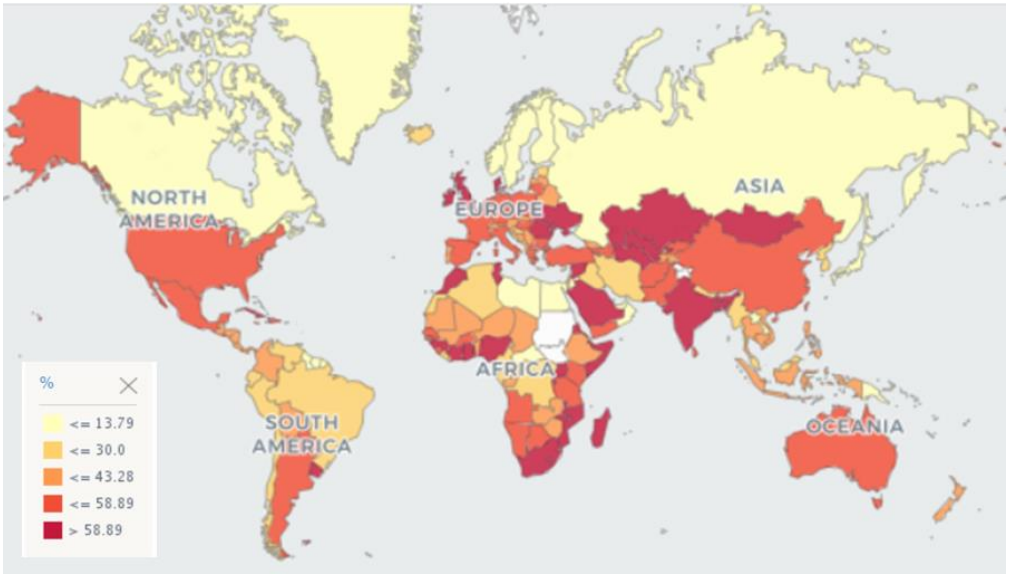


Figure 1.1. Share of agricultural land use by country. Share of agricultural land as percent of total land area by country (2015). Source: FAOStat.

The following trends highlight the sustainability challenges associated with the use of agricultural lands: (i) the shifting of agricultural lands to the tropics; (ii) population growth, rising incomes, and increasing demand for food, particularly animal protein; (iii) losses and waste along food supply chains; (iv) expansion of crop-based biofuels; (v) urbanization and (vi) teleconnections in land use as a result of globalization.

The share of agricultural land to total land (also cropland to total land) appears to be relatively stable worldwide. However, most developed countries have been experiencing a decline in agricultural lands since the 1960s, while in the developing and the least-developed countries, agricultural lands have been expanding (Fig. 1.2). In particular, significant expansion of agricultural lands has been documented in tropical regions, with implications for food supply and the environment (Foley et al 2011). Crop yields in the tropics tend be lower (Gibbs et al 2008, Monfreda et al 2008). Also, in the twenty years between 1980–2000, forest clearing was the source of over 80% of the new agricultural land created in these regions, adding to the planet's carbon debt, endangering habitats and adversely affecting many critical ecosystem services provided by these intact ecosystems (Gibbs et al 2010).

The world's population will reach 9.7 billion by 2050 (UN 2019), along with an anticipated doubling of food demand (Tilman et al 2011). This is accompanied by rising incomes, which generally translates to higher consumption of animal protein (Godfray et al 2010, Tilman et al 2011), even though it is an inefficient means of delivering food calories. If current production and consumption trends persist, nearly 1 billion hectares of land could be cleared globally by 2050 to meet this demand (Tilman et al 2011).

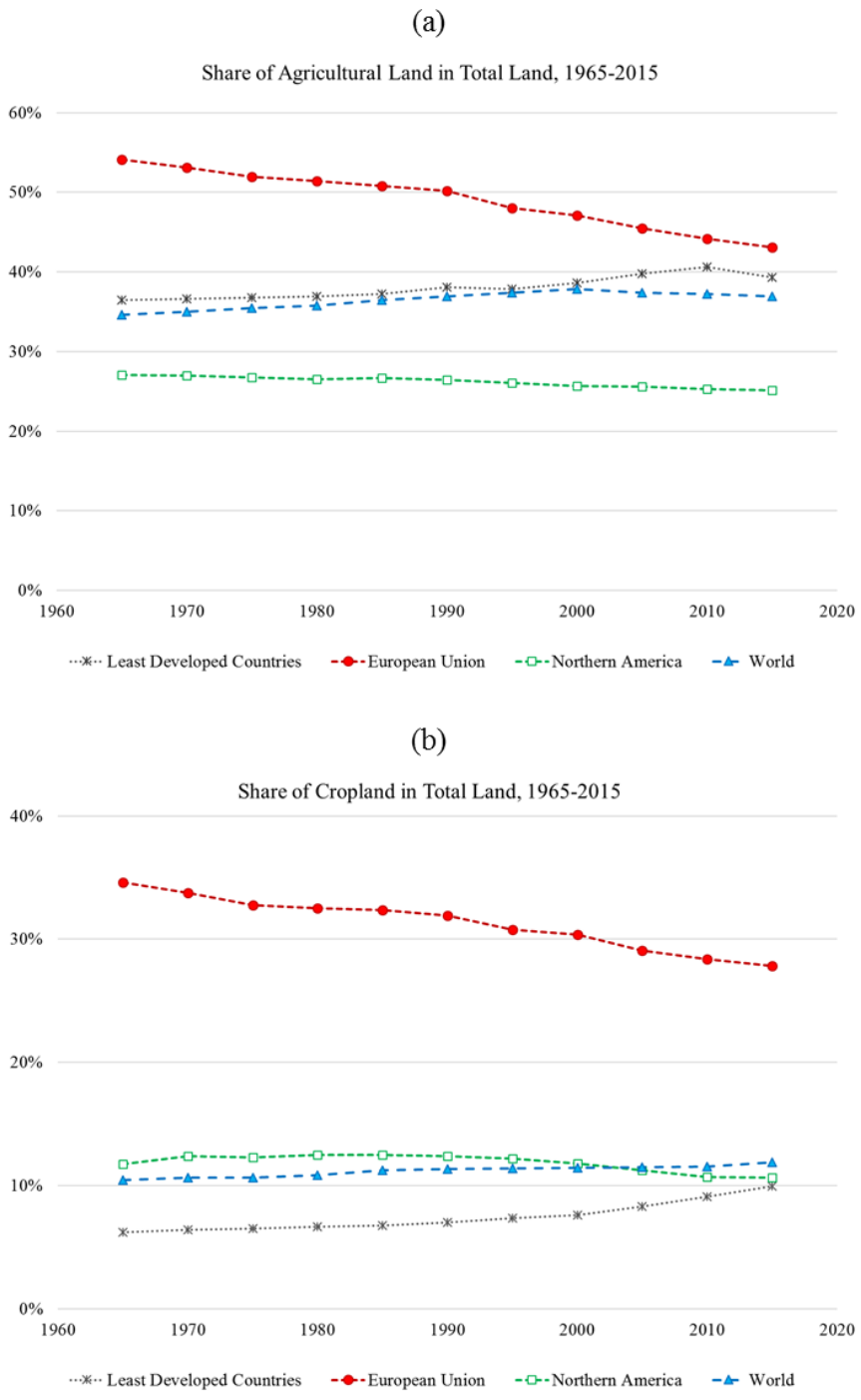


Figure 1.2. Global trends agricultural land use. Change in the share of (a) agricultural land and (b) cropland in total land area for selected regions (European Union, Northern America, Least-Developed Countries, and World), in 5-yearly increments from 1965 to 2015. Source: FAOStat.

Production and distribution inefficiencies at all stages of food production--from harvest to household consumption--indicate approximately one-third of all food produced worldwide is lost or wasted. The corresponding land footprint is 1.4 billion hectares (2007), or 28% of all agricultural land used worldwide. Dairy and meat products are responsible for nearly 80% of this land impact (FAO 2013).

The use of edible crops as biomass for liquid fuels production is a relatively new agricultural use of land (Cassidy et al 2013, Rulli et al 2016). These first-generation biofuels fuels can be less-carbon intensive substitutes for petroleum-based liquid fuels, which depends on several factors, including the underlying land used to grow the biomass (Fargione et al 2008). A period of rapid expansion of first-generation biofuels buoyed by policy support in the EU and the U.S. brought about unintended consequences associated with food supply and land use (Golub et al 2013, Khanna and Crago 2012, Chen and Khanna 2013). Diversion of productive croplands for biofuels effectively reduces net calories available as food, but it could also shift cropping to other locations, which may generate a carbon debt if previously uncropped lands are cleared, among other negative environmental consequences (Searchinger et al 2008). One of the largest such displacements has taken place in the U.S., where corn-based ethanol production uses about third of the country's corn supply annually, corresponding to an estimated 7% of its cropland (9 million hectares in 2011 per Mumm et al 2014).

Meanwhile, more than half of the planet's population now live in urban areas (2018), a proportion slated to increase to 68% by 2050 (UN 2018). The world's economic output, energy use and emissions from energy use are concentrated in cities (80% and 70%, respectively; Seto et al 2017). Even though cities themselves represent a small portion

(1%-3%) of global land use (Lambin and Meyfroidt 2011 and Grimm et al 2008), continued urban expansion in Asia and Africa will result in the loss of highly productive cropland with negative implications for food supply (Bren d'Amour et al 2017).

These trends have increasingly global consequences due to growing international trade in food and agricultural commodities (Lambin and Meyfroidt 2011) and global supply chains that connect emerging export-oriented, land-intensive industries in developing countries with buyers in developed economies (Lenzen et al 2012). Similarly, the land footprint of cities goes beyond their areal extent. The supply chains for the resources that support urban life are increasingly complex and often traced to places outside of the immediate regions and even countries where the cities are located. This has created "teleconnections" between between urban functions and rural land uses in distant localities (Seto et al 2012).

1.2 Sustainable land management

Sustainable land management (SLM) strategies are needed to mitigate the negative consequence of these stressors on land use and food supply. The goal–and the challenge—of SLM is to ensure food security for a growing population with as little new land clearing for croplands as possible, efficient use of existing agricultural lands for food, feed, fiber, timber and energy, while maintaining the long-term productivity of land and its capability to provide ecosystem services (FAO undated, Foley et al 2011).

On the supply side, closing yield gaps and improving the resource (input use) efficiency of existing agricultural lands are important SLM strategies. Both can be accomplished, via precision agriculture methods for example, without completely resorting to the conventional intensification approaches that has led to the overuse of agrochemicals and

overdrawing of freshwater resources for irrigation. On the demand side, improving food distribution and access would reduce post-harvest crop losses and food waste. Shifting diets away from grain-fed dairy and meat sources would free up valuable plant-based calories to be directly consumed as food. Similarly, limiting the diversion of food crops for non-food purposes (bioenergy) would increase available food supply. All of these strategies are ultimately aimed at minimizing the future expansion of agricultural land use, particularly clearing of forests, and limiting the environmental impacts of existing agricultural land use.

Considering alternative productive uses for underutilized spaces (at the urban scale) and marginal agricultural lands (at larger –e.g. regional or continental--scales) can complement these SLM strategies, which is the focus of this dissertation.

Urban agriculture (UA) can address food security issues at a city-scale by increasing local food supply (mainly from fresh fruits and vegetables) and by improving food distribution and access. UA can also cut the rate of food losses and waste—which is substantial for highly perishable fruits and vegetables—by shortening supply chains. When deployed on underutilized urban spaces, UA can also improve land use efficiency within cities.

At the regional or continental scale, emphasizing the cultivation of dedicated energy crops on marginal lands for bioenergy purposes accomplishes two sustainability goals: One, the substitution of bioenergy from food crops with that from non-food crops. Two, shifting energy production away from the most productive of croplands.

However, this dissertation does not advocate for either as one-size-fits-all SLM solutions, or suggest these are the best possible uses of such underutilized spaces or marginal lands.

Rather, the objective is to demonstrate the applicability and viability of both in the U.S. context, while also developing analytical frameworks to evaluate them in other geographic settings.

1.3 Underutilized spaces and marginal lands

Underutilization suggests a resource— in this case, land—that is not used to its full capacity or potential. However, the notion of underutilization in the context of land can be hard to describe, and mean different things to different people. Broadly, the difficulty lies in the implication that there is an optimal use (or mix of uses) for a given piece of land that maximizes this potential, and with respect to that benchmark the existing land use can be improved upon or the land can be put under an alternative use (or mix of uses) to deliver larger net benefits to its owner. The range of possibilities also includes the optimal use being non-use by humans (e.g. wilderness).

There are inherent challenges in estimating these net benefits from current versus alternative uses: most crucially, this is because not all benefits and costs from a given land use can be readily measured in monetary or other common unit (e.g., biophysical, utility) terms. Additionally, costs and benefits can accrue to near as well as distant stakeholders who are not the owners of the land. From a sustainability standpoint, the numerous and multi-scale environmental functions of land resources can be difficult to value even when they are categorized into "ecosystem services" (Saunders and Luck 2016, Schröter et al 2014). On the other hand, determining the value of land based on its productive (economic) uses is comparatively straightforward, especially in a market economy as the U.S.

Thus, without a full accounting of benefits and costs under the full spectrum of alternative uses it is difficult to establish the benchmark use against which the claim of underutilization is made.

The term marginal land is used widely and conveniently to describe land from both economic and biophysical productivity perspectives, but it is mainly economic in its implications. In other words, the economic prospects of land make it marginal along with its physical characteristics. On marginal lands, "the returns to production are either meager or precarious" (Peterson and Galbraith 1932). More importantly, what are marginal lands -or more precisely land resources at the margin- can change with time, and in response to economic growth, demographic shifts, technology and policy changes. This is exemplified by the gradual shifting margin of U.S. agricultural production westward and abandonment of farmlands in the East over the past 150 years (Peterson and Galbraith 1932, Campbell et al 2008).

Not all underutilized spaces / lands are economically marginal (e.g. fallowed croplands). Some lands conveniently labeled as marginal because of poor physical characteristics (e.g. poor drainage, rocky, erodable, etc.) might be in productive economic uses. Sometimes, urban lots remain vacant and underutilized in the middle of booming city centers with high rents; clearly there could be a zoning or remedial issue with the land, but it is often the case that owners simply are waiting for rents to rise even further (Titman 1985). Clarifying these distinctions is outside the scope of this dissertation, but would be important for designing and implementing successful SLM strategies.

Research presented in this dissertation also does not address any conceptual ambiguity associated with the definition of marginal land or underutilized spaces, and does not

claim to resolve whether or not the land or spaces are truly underutilized. For its purposes, it focuses narrowly on several types of spaces that are generally accepted as underutilized with respect a dominant use type.

For example, cities exhibit a density of settlement and development that is more compact than their outlying areas or rural areas. In this setting, settlement and development is the de-facto benchmark land use and underutilized spaces can extend from urban beaches to wetlands, include community gardens, parks, and peri-urban farmlands, as well as parking lots, abandoned buildings, empty lots, and inactive industrial sites. Traditionally, however, underutilized spaces in an urban context have been associated with vacant and/or abandoned lots or buildings (Pagano and Bowman 2000). Even then, precise definitions vary from one locality to another (Mori 2004).

Focusing specifically on vacant lands and buildings in the U.S., Newman et al (2016, in an update of Pagano and Bowman 2000) survey 124 U.S. cities with populations >100,000. They report, on average, 16.7% of total urban area is made up of vacant land. The estimates vary regionally (9.3% in the Northeast versus 23.5% in the South). At the city-scale, estimates reveal even wider discrepancy, with proportion of vacant land ranging from <1% to more than 70% of total land. Using the authors' estimate for the Northeast region (which is the most urbanized region of the U.S.), the amount of vacant lands in U.S. urban areas is about 2.6 million ha.

In this dissertation, the definition of underutilized spaces in an urban setting is extended to include rooftops and building façades in addition to the underutilized vacant parcels. Rooftops and building façades have been considered as potential suitable spaces for UA applications (e.g., Clinton et al 2018 and Thomaier et al 2015), but not necessarily

described as underutilized. There are few country- or global scale estimates of urban spaces associated with building exteriors: one example is Clinton et al (2018), which estimates approximately 1.4 million ha of total available space globally from these sources. The same study, aggregating spaces from building exteriors with vacant areas, estimates the global potential area available for UA to range from 7 million ha to 11 million ha worldwide. According to the authors, the U.S. is the largest source of (underutilized) urban spaces for UA with 2 million ha of available space that represents about 7% of total urban area in the U.S. (based on Bigelow and Borchers 2017).

Other country- or global-scale studies on UA consider "land" requirements, but not necessarily the availability of (underutilized or other) urban spaces for UA. For instance, Badami and Ramankutty (2014) estimate what proportion of urban land would need to be used for UA for select countries to improve food security for the urban poor. (They report 1.3% of urban land area in the U.S. is needed). Martellozzo et al (2015) estimate one-third of global urban area would need to be used for UA to meet the existing vegetable demand of urban residents (>21 million ha). Thebo et al (2014) estimate existing agricultural land use within urban and peri-urban areas and report as much as 67 million ha of urban land is currently used for cropping.

Descriptions of underutilized lands are variable in non-urban settings too. For example, croplands can be considered underutilized if observed crop yields fall short of the physical/climatic potential (i.e., the "yield gap"). Also, some lands are either not suitable for agriculture--due to low soil productivity or other concerns that make it risky for growing crops--or prone to degradation (Bouwman 2007), and therefore deemed marginal. Marginal lands may also refer to abandoned or degraded lands, wastelands, and

otherwise idle lands. Abandoned lands were at some point used for agriculture or pasture, but no longer in productive use (Field et al 2008). Degraded lands due to soil erosion, desertification, salinization, acidification can no longer provide ecosystem services without aid or intervention. The physical and ecological characteristics of wastelands make them unfit for agriculture. Lastly, idle lands can include all of the above in addition to undeveloped lands and conservation areas (Farrel et al 2006). Independent of their physical state, marginal lands could also arise in response to changing economic conditions, public policy, or land management practices (Wiegmann 2008). For example, the process of urbanization creates underutilized agricultural lands at the urban-rural fringe (margin) as farming activity declines in anticipation of development.

Since around 2006, literature has commented on the potential of marginal lands to support biofuels production using biomass from non-crop feedstocks (e.g. Tilman et al 2006). Based on large-scale resource assessments estimate the size and distribution of such lands, the scale of marginal lands is at least several hundred million hectares, and could be as high as a billion hectares worldwide. For the U.S., comparable estimates indicate a range of 43 million ha to >100 million ha.

Independent of their potential for biomass energy agriculture, Houghton (1991) estimated 500 million ha of degraded lands globally, 60% of which was in Africa, and the balance distributed equally in Asia and Latin America. The source for these degraded lands were areas identified as formerly tropical forests that remained undeveloped and not used for agriculture. Using these degraded land estimates as a starting point, Tilman et al (2006) calculate bioenergy could meet 25% to 35% of global energy needs.

Field et al (2008) focus on abandoned agricultural lands whose size they estimate to be 386 million ha globally (with ±50% uncertainty). In their analysis, the authors use HYDE database of historical land cover change from 1700 to 2000 to determine areas that were once used for agriculture (cropping and pasture), but not any longer. An initial estimate (>700 million ha) is adjusted downwards by removing lands that have since become urbanized, reforested, or went back into agricultural use based on more recent land use /land cover data from MODIS. They estimate the potential of biomass energy (content) to be about 5% of global primary energy consumption.

Campbell et al (2008) refine the area estimates of abandoned agricultural lands of Field et al (2008), while also employing a global ecosystem model (CASA) for determining biomass yields. In this study, the available area is 474-579 million ha range (and excludes transitions between crop and pasture). The energy content of dry biomass harvested from this land base is about 7-8% of world primary energy demand, although sufficient to meet energy needs in certain regions (e.g. Africa).

Cai et al (2011) use a range of definitions for marginal agricultural lands to estimate the potential for second-generation biofuel production in Africa, Europe, South America, and China, India, and the continental U.S. The low-end of estimated land available is 320 million ha of land based on the narrowest definition of marginal lands (limited to abandoned and degraded cropland and mixed crop and vegetation land). Under the most extensive marginal land designation, existing pasturelands, grassland, savanna, and shrubland with marginal productivity are also included. Then, the total available land exceeds 1,100 million ha globally. (This is approximately 25% of existing agricultural lands in the world). The authors estimate such land use could meet as much as 58% of the

[then] current world liquid fuel consumption. Based on Cai et al (2011) analysis, the available land base in the U.S. varies from 43 million ha to 128 million ha. Zumkehr and Campbell (2013) also emphasize the use of abandoned croplands for bioenergy purposes, but focus on the U.S. They create historical agricultural land use map for the continental U.S. combining recent census data on agricultural land use with a simple land use change model applied backwards to estimate historical croplands at the county-scale. This study estimates 45 million ha of abandoned croplands, whose bioenergy potential can meet 4% to 30% of U.S. liquid fuel demand, under a range of biomass yield assumptions.

1.4 Description of the three papers and the organization of the dissertation

The three papers that constitute this dissertation are:

1. Urban Agriculture's Bounty: Contributions to Phoenix's Sustainability Goals (Paper 1; Chapter 2 of the dissertation),
2. Can Environmentally Sustainable Land Utilization for Bioenergy Crops Support a Cellulosic Biofuels Industry? (Paper 2; Chapter 3 of the dissertation), and
3. Role of land quality in corn acreage response to price and policy changes: evidence from the Western Corn Belt (Paper 3; chapter 4 of the dissertation).

The study areas for all three papers are in the U.S., a highly urbanized country with a large agricultural sector. In 2012, U.S. agricultural land use of approximately 422 million ha represented more than 50% the total land use, despite a steady decline in croplands since 1950s (by 18%; Bigelow and Borchers 2017). The U.S. also makes up 10% of world agricultural land use, and is second only to China (2012 data from FAOStat).

More than 82% of U.S. population live in urban areas (compared to 55.3% for the world; UN 2018). Urban land use in the country is about 28.3 million ha (3% of the total land use). It has nearly tripled since 1949 in response to economic and population growth (Bigelow and Borchers 2017).

Below, I briefly describe the research objectives and summarize key features of each paper. The papers are then presented in the order above as Chapters 2-4. Chapter 5 provides a brief conclusion and outlines possible avenues of future research.

The first paper conducts a city-scale analysis to quantify benefits from UA for the City of Phoenix, Arizona. The paper adopts the data-driven approach of Clinton et al (2018) in quantifying these benefits. It also develops a "desired outcomes" framework which uses metrics to assess the effectiveness of UA in meeting the city's 2050 sustainability goals in terms of local food supply, reducing energy and emissions from buildings, and increasing green open spaces. The paper asks two key research questions:

- Can UA be a tool to advance sustainability goals in arid cities like Phoenix and what tradeoffs exist with use of scarce water resources?
- Can UA help meet urban sustainability goals in addition to food production?

The second paper zooms out with a study area that extends to the entire CONUS. The objective of the paper is to demonstrate how the future demand for low-carbon liquid fuels in the U.S. can be met with biofuels from sustainably grown energy crops, while ensuring an economically viable biofuel industry. The analysis in this paper focuses on the final (economic optimization) stage of a coupled hydroclimate-ecosystem-economic modeling approach. Sustainably grown energy crops are perennial grasses (miscanthus or switchgrass) grown on marginal lands (based on gridded data from Cai et al, 2011) to

avoid competing with food crops grown on prime farmlands. The land base for biomass supply is further restricted to eastern CONUS where rain-fed agriculture is feasible (specifically, East of the 104th Meridian). The analysis also assesses the supply and economic implications of placing an additional constraint on the land-base with regards to soil moisture (over a 10-year period). Avoiding regions with soil moisture loss prevents overuse of surface and groundwater resources. A novel feature of the integrated modeling framework involves the use of modeled bioenergy crop yields that reflect the hydroclimatic feedback resulting from large-scale land conversion to bioenergy crops. The principal questions asked by Paper 2 are:

- What is the resource (land base) and fuel price scenarios under which sufficient biofuels are produced to meet the future, non-electrifiable portion of U.S. liquid fuels demand?
- What are the output impacts of land- and water-use related constraints imposed to ensure sustainability of these biofuels?

The third paper investigates the expansion of corn cultivation in the Western Corn Belt (WCB) region in response to changing crop prices over a three-decade period (1986-2015). The WCB is a five-state region where concerns have been raised about conversion of previously uncultivated (native) grasslands into corn cultivation following changes to U.S. biofuel policy (e.g., Wright and Wimberly 2013 and Lark et al 2015). It contains a large share of the marginal agricultural lands identified by Cai et al (2011) and is also home to ecologically sensitive areas, like the Prairie Pothole Region (PPR). The main research questions of this paper are:

- How has corn cultivation expanded on lower versus higher quality lands in response to high corn prices?
- Has this price response varied before and after the change in US biofuel policy in 2005?

If lower quality--more marginal, less productive--lands move in and out of corn cultivation more easily in response to changing crop prices, this has implications for alternative uses of marginal lands (like bioenergy). For environmentally vulnerable marginal lands, high prices would generate larger environmental impacts, but small output benefits.

The first two papers in this dissertation are assessments under fixed economic conditions representing snapshots in time, whereas the third paper allows (crop) prices to vary by taking a historical view and conducting an analysis using panel data.

The research in this dissertation is empirical in nature and utilizes secondary data, most of which is publicly available. The land use data are both GIS-based (e.g. USDA Crop Data Layer, gridded marginal land map from Cai et al (2011), LiDAR, NAIP imagery) and survey or census-based data collected by the federal government (USDA) as well as parcel-level cadastral data from local government. These data are complemented with other spatially explicit data sources (administrative boundaries, soil maps, climate and weather data, maps of industrial infrastructure, roads, and railways). Socioeconomic and other data is sourced primarily from the U.S. Census Bureau's American Community Survey.

Paper 1 uses parcel and building-level data for the assessment of underutilized urban spaces, and then aggregates them to the census block group level to permit neighborhood

level analysis. Paper 2 uses gridded marginal land map from Cai et al (2011) (1km), map of existing croplands from USDA Crop Data Layer (30m), and gridded data of simulated biomass yields (10km) generated by an ecosystem model (AGRO-IBIS). The optimization analysis is done at the 10-km scale to fully represent spatial variation in simulated biomass yields. Paper 3 primarily uses county-level land use data for harvested corn acreage from the USDA, but groups the counties in the sample with respect to land quality using 90-m resolution data on soils. In this case, the advantage of county-level data is the longer temporal coverage necessary for the analysis despite it being coarser in spatial resolution than satellite-based sources.

The three papers also employ different methodological approaches and utilize different datasets. Paper 1 uses a data-driven framework that combines fine resolution GIS data with socioeconomic and physical indicators. Paper 2 uses a spatially explicit optimization model by Parker (2011) that simulates a profit-maximizing cellulosic biofuel industry based on simulated biomass yields, product and input prices, transportation costs, and other relevant constraints (such as technology parameters). Paper 3 uses a multivariate regression model to estimate the price response of corn acreage at county-scale. Table 1.1 summarizes the key features of the three papers.

1.5 Collaborations and publication status

Two of the papers in this dissertation are collaborative efforts, led by the author of this dissertation.

Paper 1 is work joint with Michelle Stuhlmacher, Jordan P. Smith, Matei Georgescu (School of Geographical Sciences and Urban Planning, Arizona State University) and Nicholas Clinton (Google, Inc.). It was published in Environmental Research Letters in

September 2019 (https://doi.org/10.1088/1748-9326/ab428f). I had a leadership role at all stages of this paper, from study design to the publication. Specifically, I, M. Stuhlmacher and M. Georgescu were responsible for study design and developing the city-scale framework. I substantially performed the literature review with contributions from J.P. Smith. The 2017 update to Phoenix, Arizona vacant lots assessment was led by J.P. Smith, with M. Stuhlmacher and I performing the GIS aggregation of parcel level data to to block group level. M. Stuhlmacher and I worked jointly on the assessment of available building spaces (rooftops and façades) for urban agriculture using LiDAR data. M. Stuhlmacher and I also jointly undertook the energy flux calculations involving rooftops with and without urban agriculture substrate. I collected the data on suitable crops, calories, food losses, water use and other relevant data, and conducted the crop ranking and food supply analysis, including the sensitivities. I also collected and analyzed the socioeconomic data (e.g. on food deserts, energy use, demographics, income) used in the discussion. I performed all analysis and manuscript revisions in response to feedback from anonymous reviewers. I substantially wrote the manuscript, with all coauthors taking part in refining the finished product initially for journal submission and later for final publication purposes.

Paper 2 is collaborative work with Nathan Parker (School of Sustainability, Arizona State University), Meng Wang, and Matei Georgescu (School of Geographical Sciences and Urban Planning, Arizona State University) Dr. Andrew VanLoocke, Iowa State University, and Dr. Justin Bagley (Carbon Lighthouse). It is slated for submission for publication during Spring 2020. This paper constitutes the final stage of a larger coupled hydroclimate-ecosystem-economic modeling effort concerning a future cellulosic biofuel

industry in the U.S. It summarizes the economic modeling results, which are obtained using a spatially-explicit optimization model. In this paper, M. Georgescu, N. Parker and I worked on study design. I and N. Parker developed scenarios for modeling. I collected, analyzed, and processed all of the spatially-explicit input data used for the economic optimization model, except for the simulated bioenergy crop yields. These were provided by A. VanLoocke and J. Bagley. Summary results from hydroclimate simulations relevant to the economic modeling were provided by M. Wang. I performed the spatially-explicit biomass assessment that is at the core of the optimization model. I assisted N. Parker in customizing the other sub-models that feed into the economic optimization, such as the network model of biomass transportation and fuel distribution costs. N. Parker conducted optimization runs, and I processed and analyzed all output from these runs. As in Paper 1, I substantially wrote the manuscript, with valuable comments and suggestions from M. Georgescu and N. Parker to improve exposition.

Paper 3, which is a sole-authored paper, was published in Environmental Research Communications in July 2019 (https://doi.org/10.1088/2515-7620/ab2c3f).

Table 1.1

Key features of the three papers included in the dissertation

	Paper 1	Paper 2	Paper 3
Title	Urban Agriculture's Bounty: Contributions to Phoenix's Sustainability Goals	Can Environmentally Sustainable Land Use for Bioenergy Crops Support a Cellulosic Biofuels Industry?	Role of land quality in corn acreage response to price and policy changes: evidence from the Western Corn Belt
Method	Data-driven assessment	Economic optimization as part of larger integrated hydroclimate-ecosystem-economic modeling	GIS and multivariate regression analysis
Geographic scope	City of Phoenix, Arizona	CONUS	Western Corn Belt (398 counties)
Spatial scale	Census block group	"Pseudo-farms" at 10 km scale	County
Temporal scale	Snapshot circa 2016/2017	Single year (2012)	1986-2015
Land use data	Cadastral data, NAIP imagery, and LiDAR	Map of suitable marginal lands for bioenergy crops from [8] and USDA Crop Data Layer (2012) for competing land uses (aggregated to 10 km pixel)	USDA NASS survey data on land use and acreages (county-year)
Dependent variable	Metrics based on the desired outcomes associated with local sustainability goals for local food systems, energy and buildings, and green open spaces	Aggregate profits to cellulosic biofuel industry	Acreage planted for corn
Key independent variables	Local data on suitable crop yields, municipal water use, energy use and emissions, weather, land values, population, and other socioeconomic variables	Crop yields, input and transportation costs, technology parameters, product prices	Crop and input prices, CRP rents; weather/climate and soil quality
Outcome	Proportion / percentage delivered for the metrics described above (dep. var.)	Supply curves for cellulosic biofuels and configuration of biofuel industry	Average acreage response to crop prices for typical WCB county
Is outcome spatially explicit?	Yes	Yes	No

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CHAPTER 2

URBAN AGRICULTURE'S BOUNTY: CONTRIBUTIONS TO PHOENIX'S SUSTAINABILITY GOALS

ABSTRACT

With over half of the world's population living in cities, there is mounting evidence indicating that investments in urban sustainability can deliver high returns on socioeconomic and environmental fronts. Current scholarship on urban agriculture (UA) reports a wide range of benefits which have been shown to vary with the scale and type of benefit examined. Notably, most city-scale studies do not align benefits of UA with locally meaningful goals. We fill this gap by conducting a city-scale analysis for Phoenix, the fifth largest city in the USA by population and evaluate these benefits based on their ability to contribute to select desired outcomes specified in Phoenix's 2050 Sustainability Goals: the elimination of food deserts, provision of green open space, and energy and CO_2 emissions savings from buildings. We consider three types of surfaces for UA deployment—undeveloped vacant lots, flat rooftops, and building façades—and find that the existing building stock provides 71% of available UA space in the study area. The estimated total food supply from UA is 183,000 tons per year, providing local produce in all existing food deserts of Phoenix, and meeting 90% of current annual consumption of fresh produce based on national per capita consumption patterns. UA would also add green open space and reduce by 60% the number of block groups underserved by public parks. Rooftop deployment of UA could reduce energy use in buildings and has the potential to displace more than 50,000 tons of CO_2 per year. Our work highlights the importance of

combining a data-driven framework with local information to address place-based sustainability goals and can be used as a template for city-scale evaluations of UA in alternate settings.

2.1 Introduction

Globally, the proportion of urban dwellers is expected to increase from 55% to 68% by 2050 (UN 2018). Cities occupy a small portion of the world's land area (1%-3%), but are responsible for most of the energy and resources used and emissions generated (Seto et al 2017; Grimm et al 2008). Today, large-scale urbanization is a change process occurring simultaneously in many parts of the world (Seto et al 2017). However, the nature of these urban transitions and patterns of urbanization varies from place to place, each with their particular challenges and consequences (Cohen 2006 and Seto et al 2017). Although the growth of cities in Asia and Africa involves loss of highly productive cropland with implications for global food production (Bren d'Amour et al 2017), developed cities in North America and Europe have completed their agrarian transition. There, retrofitting existing building stock is a sustainability priority that will lead to substantial reductions in energy use (Georgescu et al 2015; Guneralp et al 2017). Therefore, it is important to identify the appropriate city-scale response for achieving sustainability, which then needs to be complemented by an understanding of feedbacks and interactions across scales.

Urban agriculture (UA) is one tool in a portfolio of measures for achieving urban sustainability and making cities more resilient (Dubbeling et al 2009; de Zeeuw et al 2011; de Zeeuw and Dreschel 2015). It contributes to resilience by providing locally produced food and diversifying existing food supply, creating alternative earning opportunities for residents (de Zeeuw et al

2011 and Zezza and Tasciotti 2010). It serves as an opportunity to productively use increasingly scarce urban water resources by reusing greywater, and redirecting household organic waste (de Zeeuw et al 2011). Commodities supplied through UA are distributed via short marketing chains, potentially reducing transportation costs and associated emissions (Weber and Matthews 2008). UA landscapes can act as urban habitats for wildlife (Goddard et al 2010).

There are risks and drawbacks associated with UA that may reduce the magnitude of benefits, though these can be managed by testing, treatment, and use of appropriate management practices. Food production can be inefficient with respect to large-scale commercial agriculture (Desrochers and Shimizu 2012). Plants grown on contaminated soils (Beniston and Lal 2012; Meharg 2016) or in areas of high air pollution (Bell et al, 2011) reduce yields and pose health and safety concerns. Other risks associated with UA involve the application of herbicides and pesticides, and use of untreated manure or wastewater, and introduction of new pests and diseases (Smit et al 2001).

Global studies (Clinton et al 2018, Martellozzo et al 2014) emphasize the scale of the UA opportunity. In addition, quantifying the potential for co-benefits that extend beyond merely addressing food insecurity concerns (Clinton et al 2018) require place-based assessments to tailor how UA can be used to achieve sustainability and contribute to resiliency in specific urban contexts. In the process, it is essential to focus on solutions that are consistent with place-based desired outcomes—"a sustainability wish list"— of urban policymakers and residents. These can accelerate the transition towards urban resilience and reduce risks due to a lack of understanding of the socio-political infrastructure (Eakin et al 2017). This makes sustainability science relevant

to city needs, consistent with its goal to be "use-inspired" and "action-oriented" (Turner et al 2007; Clark, 2007 and Seto et al 2017).

Here we quantify UA benefits for the City of Phoenix, AZ, through a "desired outcomes" framework by connecting UA with Phoenix's locally defined sustainability goals. Phoenix is the fifth largest and second fastest growing city in the U.S. by population (1.63 million people in 2017 with 1.5% increase over 2016-2017; U.S. Census Bureau, 2018a). The region's climate allows for year-round crop production with the aid of irrigation. Not unlike other Sunbelt cities, however, urban population growth and climate change exacerbate existing water-resource concerns for Phoenix (Georgescu et al 2013; Gober and Kirkwood, 2010). The overarching question we address here is to what extent can UA be a tool to advance sustainability goals in arid cities like Phoenix and what tradeoffs exist with use of scarce water resources?

We adopt the data-driven approach introduced by Clinton et al (2018), which performed the first global assessment of ecosystem benefits owing to UA deployment. Our work builds upon prior city-scale assessments of UA (e.g., Ackerman et al 2014; CoDyre et al 2015; Haberman et al 2014; Mack et al 2017; McClintock et al 2013; Mendes et al 2008; Orsini et al 2014; Patel and MacRae 2012; and Saha and Eckelman 2017) by extending analysis beyond food production. We inquire whether a broader range of locally-defined sustainability outcomes, in addition to food production, could be attained, and if so to what extent, through UA adoption. The developed framework is translatable to other cities and can be used as a template for similar city-scale evaluations of UA as well as other urban sustainability solutions.

2.2 Materials and methods

2.2.1 Study area

Phoenix is the seat of Maricopa County and capitol of the state of Arizona, in southwestern USA (Fig. 2.1A). Our study area includes all census block groups with LiDAR building footprint data that intersect the administrative boundaries of Phoenix (Fig. 2.1B).

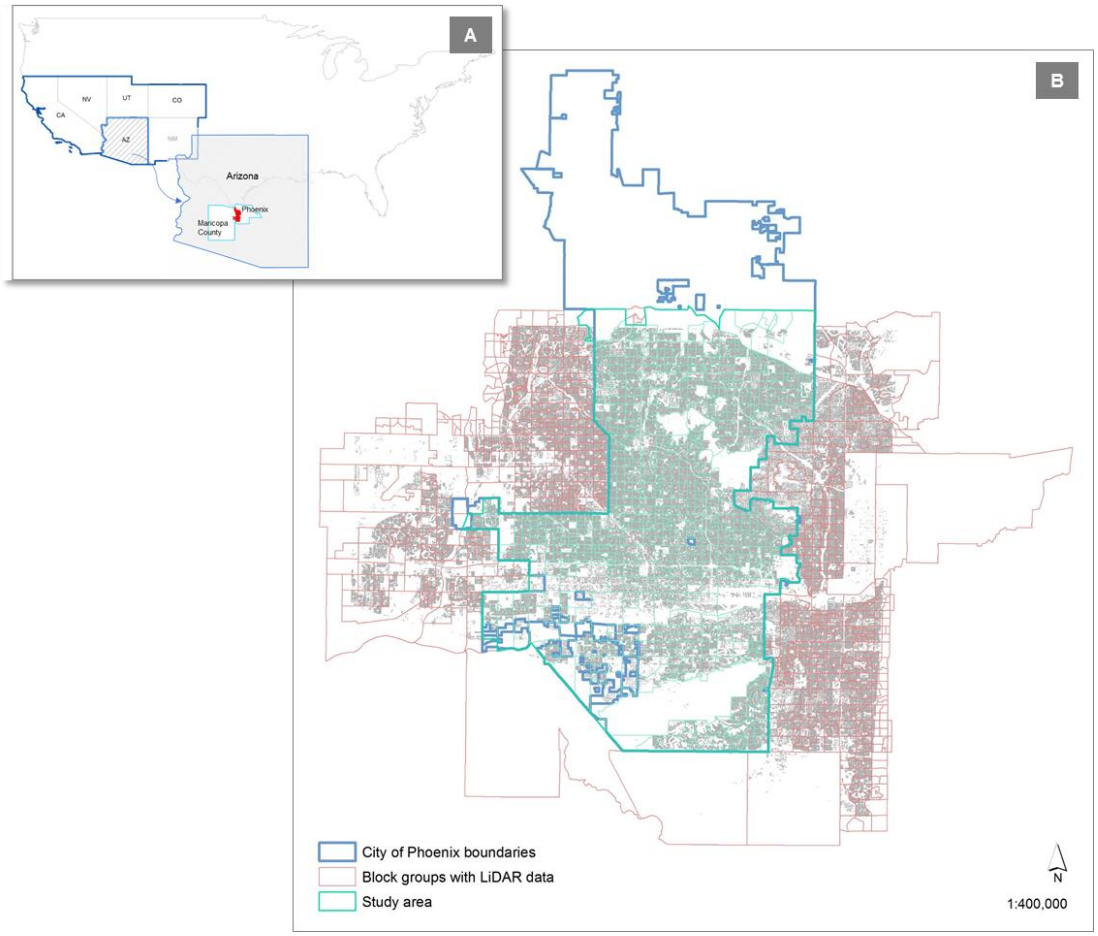


Figure 2.1. Study area. (A) The location of City of Phoenix in relation to Maricopa County, state of Arizona, Arizona's neighboring states within the USA. (B) The administrative boundaries of the city (blue); existing LiDAR buildings data coverage in metro Phoenix by block group (red); and the census block groups constituting the study area (green).

Arizona is the sixth largest state in the U.S. by land area. Urban areas constitute 2% of land in Arizona (compared to 3% across the U.S.), with Phoenix being the largest urban area in the state. Situated in the Salt River Valley in the Sonoran Desert, Phoenix is spread over nearly 1,344 square kilometers (519 square miles). It has a subtropical desert climate with abundant sunshine (85% of daytime hours) and receives about 203.2 mm of rainfall per year.

Agriculture has a long history in Phoenix. Evidence of irrigation canals and seasonal farming predate modern settlement by over a thousand years. Modern Phoenix also has roots as an agrarian community. The city originally consisted of a small urban core surrounded by farmland. Since the 1940s, it has transitioned into a commercial and industrial center, resulting in the current land use pattern observed: loss of farmland to development and vacant lots in the urban core and along the periphery.

2.2.2 Desired outcomes and metrics

In 2016, the City of Phoenix adopted the 2050 Environmental Sustainability Goals towards becoming a "Sustainable Desert City" as envisioned in the city's General Plan (City of Phoenix, 2017a). The city's current goals were developed with widespread input from the community and encompass buildings and energy use, clean air, water use, parks and open spaces, local food systems, transportation, and waste management. Along with each goal are associated desired long-term outcomes.

We focus on three of these sustainability goals for which UA can help deliver the associated desired outcomes. These are (1) local food systems, (2) open space provision, and (3) building and energy use. We identify the delivery mechanism and associated metrics to evaluate the effectiveness of UA in meeting these desired outcomes (Table 1). We then quantify these metrics

on an aggregate city (study area) scale and analyze and report them at the block group level to pinpoint areas of the city that could benefit from prioritized deployment of UA. This method considers the variations in available area for UA as well as resource and income disparities in different parts of the city.

Table 2.1

How UA can contribute to Phoenix's sustainability goals

City of Phoenix 2050 Sustainability Goals Category*	Desired Outcomes*	UA Delivery Mechanisms	Metrics

Local Food Systems 	"Eliminating food deserts" (p.13)	Help reduce food deserts via local production of food	- Local food supply from UA
			  - Total tons
			  - Tons/census block
			  - Tons/person

			- In block groups overlapping known food deserts (½ mile LILA tracts)
			  - Total tons
			  - Tons per food desert
			  - Tons/person

Parks, Preserves and Open Spaces 	"Having all residents within a five-minute walk of a park or open space by adding new parks or open space in underserved areas." (p.10)	Create green open spaces by repurposing vacant lands as urban farms or community gardens	- Increase in green space area (%)
			- Number of census blocks with public green space that formerly had none (#)
			- Increase in the 5-minute green open space access zones (% area and population served)
Energy: Buildings and Land Use 	"Reduce carbon pollution from vehicles, buildings, and waste by 80%-90%" (p.3)	Create rooftop gardens that insulate buildings	- Avoided electricity use in buildings from added insulation provided by rooftop deployment of UA (MWh)
			- Avoided CO_2 emissions as a result of reduced building electricity use (metric tons)

Connects the City of Phoenix's sustainability goals, associated desired outcomes through deployment of UA by identifying the specific delivery mechanism and metrics. (*) City of Phoenix (2017a).

2.2.3 Available area estimation for UA

We focus on underutilized exterior spaces suitable for UA. Following Clinton et al (2018), we consider three types of urban space available for UA: undeveloped (unpaved) vacant lots, flat

rooftops, and building façades (vertical growing spaces). Rooftop UA infrastructure involves a substrate made up of specialized membranes and drainage barriers for growing vegetation on top of buildings. The façade format grows vegetation alongside exterior walls of buildings, using infrastructure like trellises or cages to support plants. Both of these formats have been employed in UA applications (Thomaier et al 2015). We exclude other forms of vertical UA within buildings such as controlled environment agriculture (CEA; Goodman and Minner 2019) or building-integrated agriculture (BIA; Gould and Caplow 2012).

Available area from vacant lots is based on a detailed inventory of all vacant property --built, paved lots, or undeveloped lots-- in Phoenix metro area compiled for 2010 and 2017 following Smith et al (2017). The available area from rooftops and vertical surfaces are derived using the 2014 (most recently available) LiDAR building footprint data for Phoenix. We do not consider alternative land use patterns (beyond what current data reflects) for assessing food supply. (See Supplemental Information sections 2.7.1 and 2.7.5 for details.)

We derive total potential area (A_t) available for UA by aggregating areas for the three types of spaces (i) at the block group level and then summing across block groups (1).

$$A_t = \sum_{j=1}^{910} \sum_{i=1}^{3} A_{ij} \quad (1)$$

where j denotes block groups (N=910) in our study area and i denotes the types of spaces considered for UA with 1=vacant lots, 2=rooftops, and 3=building façades.

2.2.4 Local fresh produce supply from UA

We identify 34 suitable crops consisting mostly of vegetables and fruits grown in Maricopa County (2012 Census of Agriculture). Field crops (e.g., alfalfa, cotton) or tree crops (e.g., citrus)

are excluded because of potential space and time (crop growth) limitations under UA. We rank the selected crops using three criteria on an equal-weighted basis: estimated water use (mm/m^2), historical yields (ton/ha), and supermarket retail prices from 2016 ($/ton).

Crops with higher yields, lower water use, and higher prices are preferred, and therefore ranked higher. The combined preference for high yields, low water use (highest cost input) with high retail prices in ranking crops is our proxy for profitability.

We next sort the 34 crops and form terciles of suitability: good, average, and poor. Planting area is allocated proportional to suitability: e.g., crops in the "good" group are allocated more area than those in the "poor" group. All UA sites get the same crop-area allocation, except for façades, which are limited to a subset of these crops (22 out of 34) since some crops are less suitable for vertical cultivation. (See Supplemental Information, section 2.7.1 for details.)

Total annual food supply from UA is the sum of crop production at block group level (2):

$$\text{Annual food supply} = \sum_{c=1}^{34} A_{ij} * \text{Weight}_c * \text{Yield}_c \quad (2)$$

where A_{ij} is the area in m^2 by type of UA space i (e.g. vacant lots) as described previously, and each crop's weight (weight_c) is proportional to the total rank of all crops suitable for UA (3, 3.1).

$$\text{Weight}_c = \text{Rank}_c / \sum_{c=1}^{34} \text{Rank}_c \quad \text{(for vacant and rooftops)} \quad (3)$$

$$\text{Weight}_c = \text{Rank}_c / \sum_{c=1}^{22} \text{Rank}_c \quad \text{(for building facades)} \quad (3.1)$$

2.2.5 Open green space provision

We consider any open green space provision by UA to be associated with the use of undeveloped (unpaved) vacant lots (excluding rooftop/façade applications), and incremental to existing open green spaces. In our analysis, existing green open spaces are limited to public parks. Block

groups that have no public parks are considered underserved. When UA deployed on vacant lots adds open green space to a previously underserved block group, we consider that block group no longer underserved.

We compute vacant lot area that will be counted towards open green space provision in two ways. First, we consider the impact of deploying all available vacant lots for UA, regardless of lot size. Second, we place a lower limit on vacant lot size (5,000 m^2) and prioritize UA deployment on fewer but larger lots that could more easily incorporate park-like features. We refer to these larger vacant lots (either on an individual or aggregated basis) as "vacant sites". Using this sub-sample of vacant sites, we further estimate how open green space provided by UA can improve walkability using buffer analysis (See Supplemental Information, section 2.7.1 for details).

2.2.6 Energy savings and avoided emissions from buildings

Rooftop applications of UA can reduce the energy usage of and emissions from buildings via improved insulation. The UA substrate added on rooftops increases thermal resistance (R-value) of roofs and reduces building energy use. We follow Clinton et al (2018), modifying it with locally available data for Phoenix.

The difference between total kWh electricity use before UA and after UA provides an estimate of avoided building energy use. Using CO_2 emission rates from power generation in Arizona, we also derive corresponding estimates for avoided CO_2 due to UA. Further methodological details are provided in (See Supplemental Information, section 2.7.1 and 2.7.4 for details.)

2.3 Results and discussion

2.3.1 Potential area available for UA

In our study area, using 2017 data, we estimate a total of 7,230 hectares is available for UA, representing 7.1% of the land area, and 5% of total surface area (including surface area of buildings). More than half (53%) of this total area is provided by rooftops, 29% by vacant lots, and the remaining 18% from building facades. Thus, 71% of the total available area for UA is supplied by existing building stock (Table 2.2).

Table 2.2

Potential area available for UA by category

Category available area for UA	Total potential area (ha)	Share (%)	Count of units*	Average area (m^2) per unit*	Count of block groups**	Average units by block group**	Average area (m^2) by block group**
(1) Vacant lots	2,096.3	28.9%	10,708	1,957.7	565	19	37,103
(2) Rooftops	3,821.9	52.8%	20,114	1,900.1	843	24	45,337
(3) Building façades	1,311.8	18.1%	432,493	30.4	907	477	14,463
	7,230.0	100.0%					

Summary statistics for available UA area in our study area by category: vacant lots, rooftops, and building façades (vertical). (*) *Unit* corresponds to individual vacant lots for vacant lots; flat rooftop buildings with roof area >464.5 m^2 (5,000 ft^2) for rooftops. (**) Out of a total of 910 block groups in study area. The averages exclude block groups with no reported area for a given category.

Grouped by type of space available for UA, our study area contains 10,708 individual undeveloped vacant lots, over 20,000 suitable rooftops with roof areas >464.5 m^2 (5,000 ft2), which represents 35% of all flat rooftops by area, and more than 432,000 individual buildings.

Not all block groups have all types of space available for UA: of the 910 block groups in our study area, we document vacant lots in 565, suitable rooftops in 843, and buildings in 907, and only one block group has no available UA area of any type.

The area of block groups ranges from 85.5 to over 6,500 hectares in Phoenix and affects the total available area for UA. Controlling for size, the block groups with largest available area from vacant lots are concentrated in southwestern Phoenix (Fig. 2.2A). Block groups with the greatest rooftop area are found along the main commercial-industrial corridors of the city (Fig. 2.2B). The block-group distribution of UA area from façades (Fig. 2.2C) is correlated with the number of individual buildings, and therefore favors more densely settled residential districts with smaller lot sizes.

Compared to Clinton et al (2018), which completed a similar area assessment for UA at the global scale, the contribution to total UA area from vacant lots is substantially lower (29% in Phoenix vs. 80% globally), meaning that area from rooftops and façades is higher (71% in Phoenix vs. 20% globally). This is partly driven by the fine-scale inputs used in area estimation, highlighting the need for city-scale assessment.

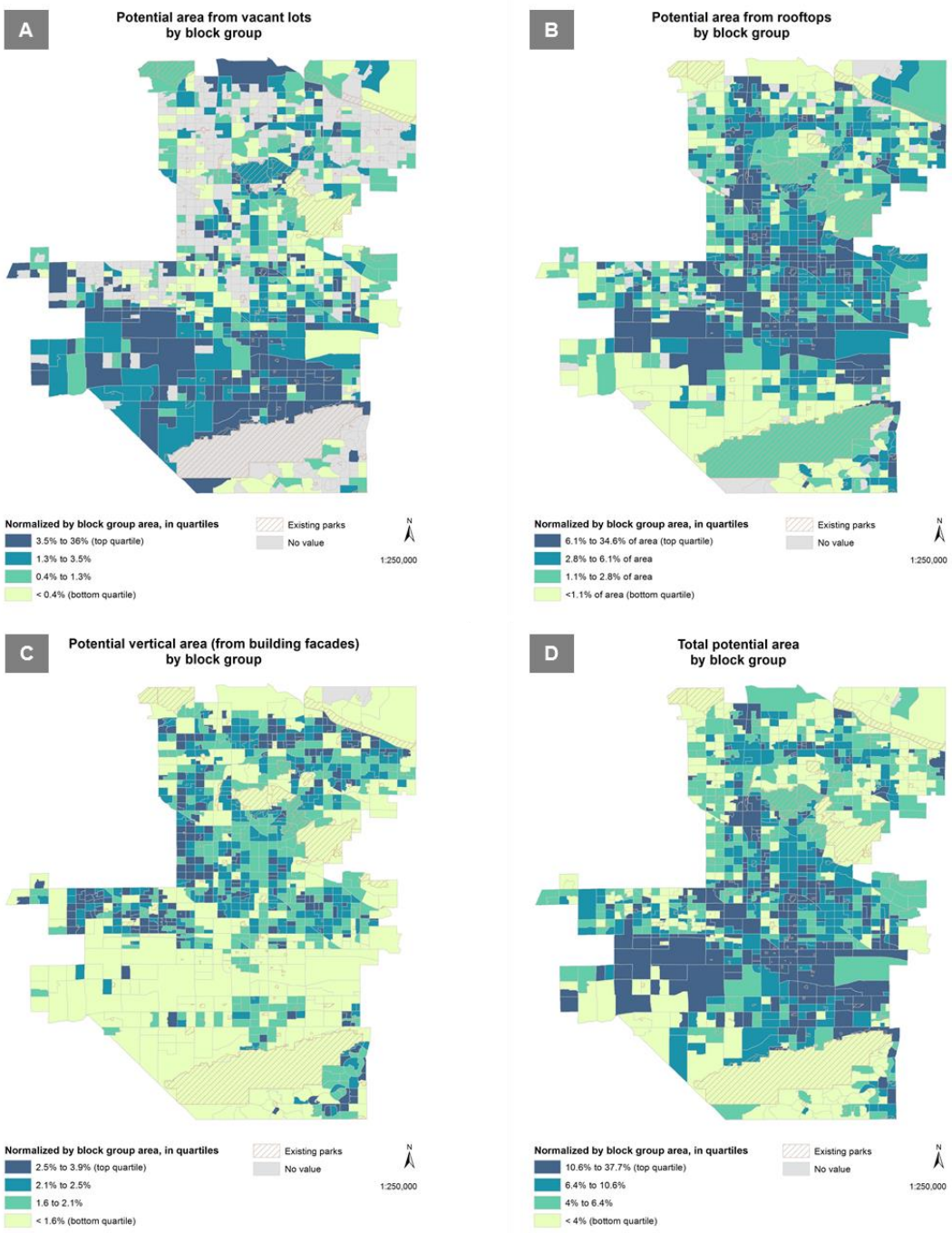


Figure 2.2. Total potential area available for UA. Each panel depicts potential area by type available for UA by quartiles of block groups normalized by block group area adjusted by regional parks that are excluded from development. (A) Potential area from vacant lots; (B) Potential area from rooftops; (C) Potential area from building facades (vertical); and (D) Total potential area that is the sum of areas in A, B, and C.

The total area for undeveloped vacant lots in Phoenix decreased from 2010 to 2017 by 23% following the economic recovery. Since we apply the same crop mix to all vacant lots available for UA, this means the corresponding food supply would have been 23% higher in 2010. Notably, the reduction in vacant lot area over this period is not uniformly observed across the city. While the majority of block groups (N=345) experienced a reduction in undeveloped vacant lot area, others increased (N=161) (see Supplemental Information, section 2.7.5 for details).

2.3.2 Local fresh produce supply from UA

The total estimated food supply from UA by primary weight (raw weight inclusive of the weight of peel, skin, pit, seed, and stem, depending on the item) is 182,983 tons per year, 89% of which comes from vegetables and the remainder from fruits. Food supply varies widely on a block group basis, ranging from under 1 ton to more than 6,654 tons. The amount of produce correlates with block group area, where larger block groups have, in general, more available area for UA and therefore supply more output (Fig. 2.3A). Normalizing food supply by the available area for UA produces a more compact distribution, ranging from 21.8 to 26.1 tons per ha of available area (Fig. 2.3B), with a block group average of 24.4 ton/ha. Higher and lower output block groups are clustered both in terms of absolute tons and on a tons/ha of available area basis (with Moran's I z-score = 13.1 $p<0.00$ and z-score = 12.0 and $p<0.00$, respectively). Higher output block groups are clustered in south Phoenix, and lower output ones tend to be in northern parts of the city.

The average output estimated in our analysis (24.4 ton/ha) is consistent with recent field data from urban gardens with broadly comparable food baskets (ranging from 10 ton/ha in Paris from Pourais et al 2015 to as much as 36.6 ton/ha in San Jose, California from Algert et al 2014). A

survey of aggregate yields from community or home food gardens in New Jersey reported 24.4 ton/ha to be the mode (Rabin et al 2012).

On a per capita basis, the food supply obtainable from UA is 125 kg/person. The median supply at block group level is 68 kg/person, with block group means ranging from 0.25 kg/person/year to over 1,000 kg/person/year. In densely populated residential neighborhoods there is generally less area available for UA, which generates smaller per capita estimates. Conversely, some of the larger block groups are less densely populated, and thus have more area available. Not every block group can produce meaningful amounts of fruits and vegetables from UA per person, and a few block groups produce very large amounts (Fig. 2.3C).

Nationally, the per capita intake of non-citrus fresh fruits and fresh vegetables is 138.7 kg/person/year (2016 figures on primary weight basis from USDA (2018a)). The estimated output from UA can meet 90% of this demand for fresh produce.

More than 85% of Arizonans do not consume the recommended amount of fruits and vegetables (Lee-Kwan et al 2017). Therefore, UA can be an important pathway for provision of fresh produce in areas where there is limited local supply (e.g. food deserts), and could supplement consumption in other areas. UA can also satisfy the increased consumer interest in local foods, based on economic evidence (willingness-to-pay that falls with the transport distance for food; Grebitus et al 2013).

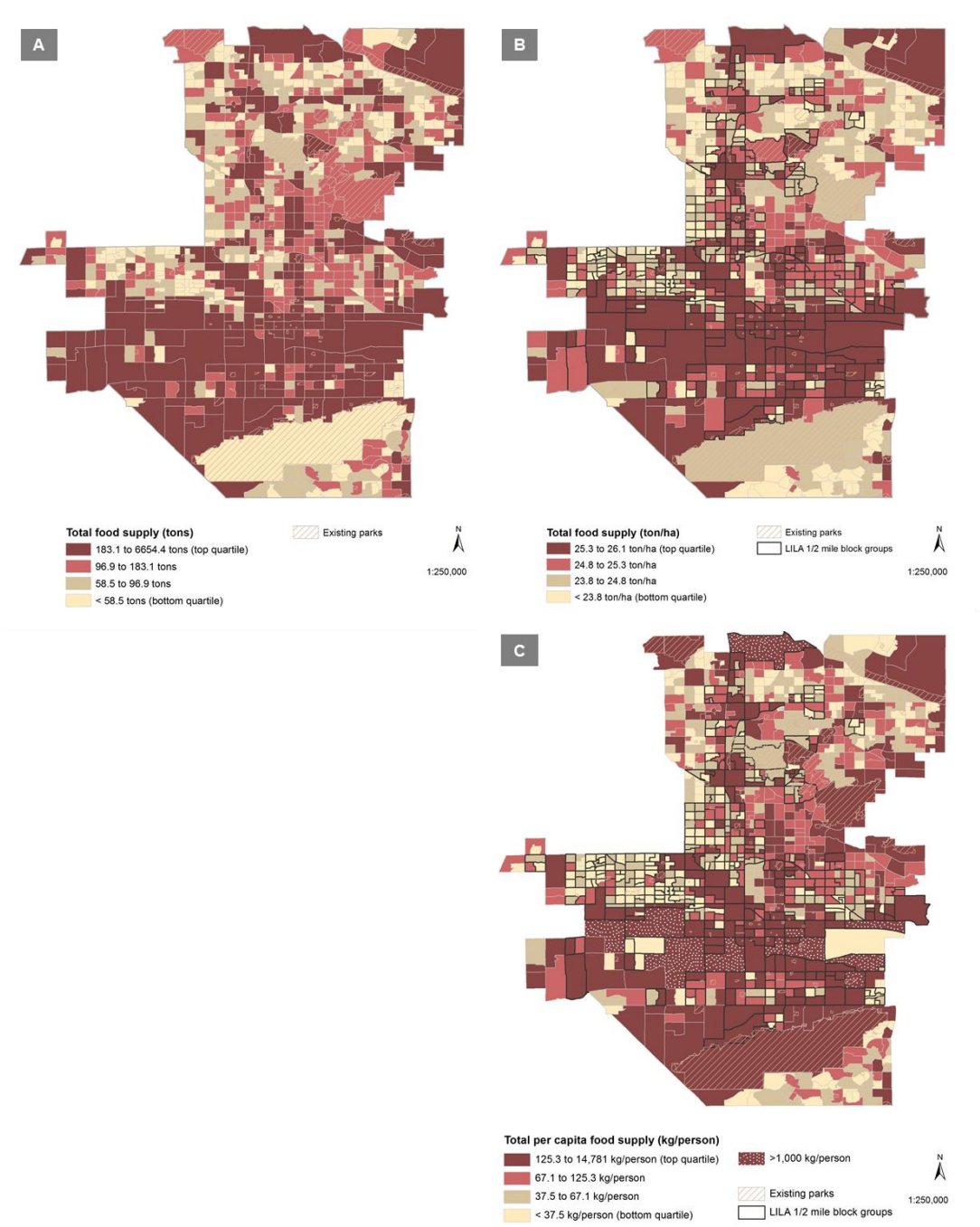


Figure 2.3. Total annual primary food supply from UA and existing food deserts. (A) By block group quartiles in tons. (B) Same as (A) but quantities are normalized by hectare of total potential area available for UA. (C) Per capita food supply from UA in kg/person terms. Outlier block groups where output exceeds 1,000 kg per person are highlighted. Known food deserts in Phoenix identified as half-mile LILA (low-income, low-access) census tracts are outlined in black in (B) and (C).

This is noteworthy because over half (51%) of the study area population lives in the 439 block groups which overlap with known food deserts (see Supplemental Information, section 2.7.4). We estimate food supply from UA in all 439 of these block groups, constituting 55% (101,008 tons) of the total UA food supply (Fig.3B). Notably, the per hectare food supply is also higher in these block groups by 0.4 ton (significant based on median test with Pearson's X^2 statistic = 18, P<0.00). Average food supply from UA is higher on a per capita basis, but this is driven by a few large and sparsely populated block groups. Statistically, the group medians are not different (66 kg/person/year in food deserts vs. 69 kg/person/year in non-food desert block groups). As Fig. 3C shows, some of the outlier block groups are food deserts.

The food desert block groups represent a segment of Phoenicians with lower median household incomes and more reliance on SNAP benefits compared to non-food deserts. Residents also tend to have lower rates of health insurance coverage, and experience higher rates unemployment. Median home values are also lower. (All differences significant per median tests with Pearson's X^2 statistics of P<0.00). Prioritizing UA deployment in these block groups can therefore be an important strategy to improve sustainability not just from a local food supply but also from a community resilience standpoint. However, this requires a clearer understanding of socioeconomic processes that might be at work. For example, the fact that we observe larger vacant lots and lower home values in food deserts suggests an underlying dynamic where vacancies might be related to lower home values (i.e., low returns for developers), while lower home values might be related to other factors (such as higher crime rates), which in turn might be influenced by high vacancy rates.

UA and sale of UA produce in farmers' markets may increase fruit and vegetable intake (McCormack et al 2010). However, the direct health and other beneficial outcomes (such as lower cost of purchased food) from eliminating food deserts are less clear because these are areas where access and affordability of healthy food options both play a role. Low income residents in food deserts either travel longer to supermarkets, or pay more for food at nearby convenience stores (Ver Ploeg et al 2009). For UA to be impactful, the cost of fresh fruits and vegetables supplied by UA in food desert areas needs to be cheaper than traveling to the nearest supermarket. Allowing UA-based food supply and UA-vendors to be SNAP-eligible could increase the opportunities for low-income residents in food deserts to benefit from UA.

Two sensitivities to this baseline analysis are also considered. One ranks our list of suitable crops based on productivity alone, and the second with respect to national consumption patterns. While food supply increases (by 16% and 6% respectively, for the high productivity and national consumption pattern scenarios), so does water use (see Supplemental Information section 2.7.2 for details). Thus, our baseline approach is more sustainable, albeit more conservative from a supply standpoint.

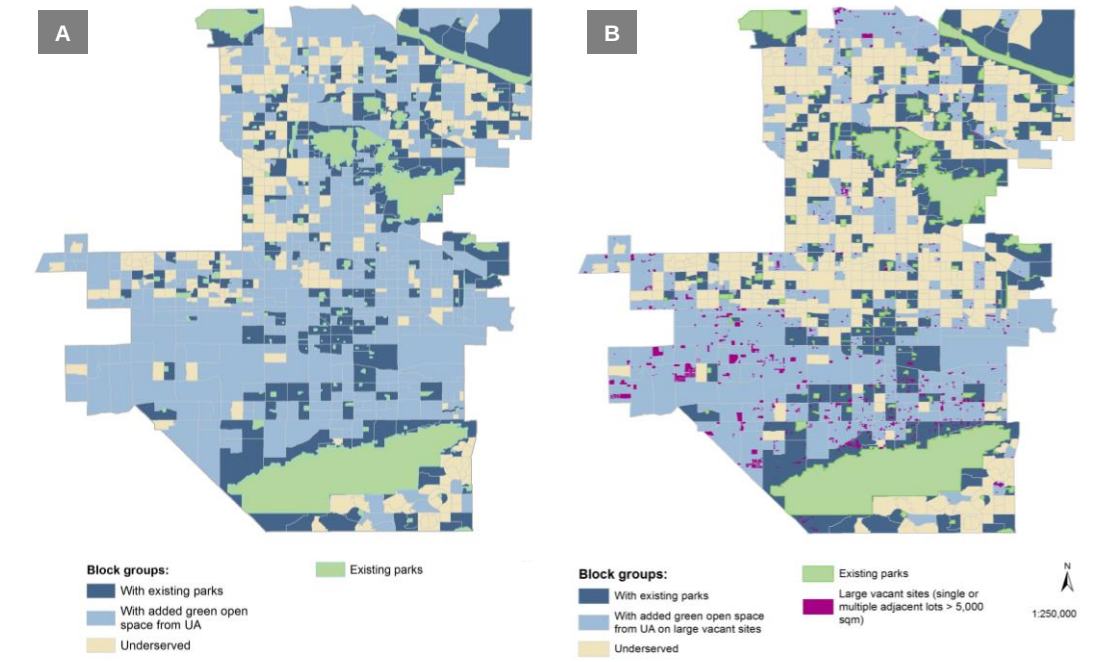
We also convert our weight-based analysis of food supply into calories. Under UA, less of the highly consumed, calorie-dense crops like potatoes are produced, compared to national consumption patterns. As a result, UA food supply provides 46% fewer calories from fruits and vegetables than the U.S. average (42 kcal/person/day vs. 92 kcal/person/day). However, on a cup-equivalent basis, which is the metric for determining recommended daily intake levels, UA food supply (1.40 cups/day) surpasses the national average (1.38 cups/day) (U.S. Department of

Health and Human Services and U.S. Department of Agriculture, 2015; also see Supplemental Information section 2.7.3 for details).

2.3.3 Open green space provision

There are 154 existing public parks in Phoenix, covering about 12% of our study area (12,467 ha). Most are local community parks (N=144) and a few are large regional parks (e.g., the South Mountain Preserve; N=10). The regional parks constitute the bulk of the public park area in the city (90%), but are not within daily reach of most residents. The block groups adjacent to these regional parks represent only 6% of the city population.

Most block groups in our study area (78%, N=713) do not have any public parks that residents can access to on a day-to-day basis and are thus underserved. If all suitable vacant lots in our inventory are developed as urban gardens, the number of block groups without any prior green open space can be reduced by 433, decreasing the proportion of underserved block groups to 31% (Fig. 2.4A), which will also expand total green open space to 14% of the study area. Vacant sites (N=764), consisting of either individual or aggregated adjacent vacant parcels with an area greater than 5,000 m2, constitute 85% of vacant lot area available in Phoenix. If the deployment of UA is strategically prioritized for these larger vacant sites, the reduction in number of underserved block groups is lower, though still substantial (N=216, Fig.2.4B). UA applications in these larger vacant sites can emphasize features like pathways that would encourage interaction with residents. Note that if UA deployment is restricted to these vacant sites alone, this would lower food supply from UA by 50%, concentrate benefits to fewer block groups, and reduce UA's coverage of food deserts (from 439 to 122 block groups).



Impact at block group level	Without UA	With UA	
		(A) Full deployment	(B) Prioritized deployment
Block groups with green open space	197	630	413
Underserved block groups	713	280	497
Previously underserved block groups with added green open space from UA	0	433	216
Total green open space area (ha)	12,467	14,564	14,254
Green open space share in total study area	12.2%	14.3%	14.0%

Figure 2.4. Green open space provision through vacant lot deployment for UA. (A) Block group availability of green open space after full deployment of all available vacant lots. (B) Block group availability of green open space based on prioritized deployment of UA on vacant sites >5,000 m2. Vacant sites highlighted in red. Both panels show block groups with existing parks; block groups that gain green open space from UA; and underserved block groups using the same color scheme.

Additionally, the walkability (to nearby parks or green open spaces) benefits from UA will accrue mainly from the 764 vacant sites identified for strategic deployment (even if UA is deployed fully throughout the study area). Based on existing parks, we determine 30% of our study area is either green open space or within 5-minute walking distance of such spaces using a

402-meter (0.25 mile) buffer zone. Under the assumption that the larger vacant sites are converted into UA, the proportion of such walkability zones increase by 25% to 55% (Fig. 2.5).

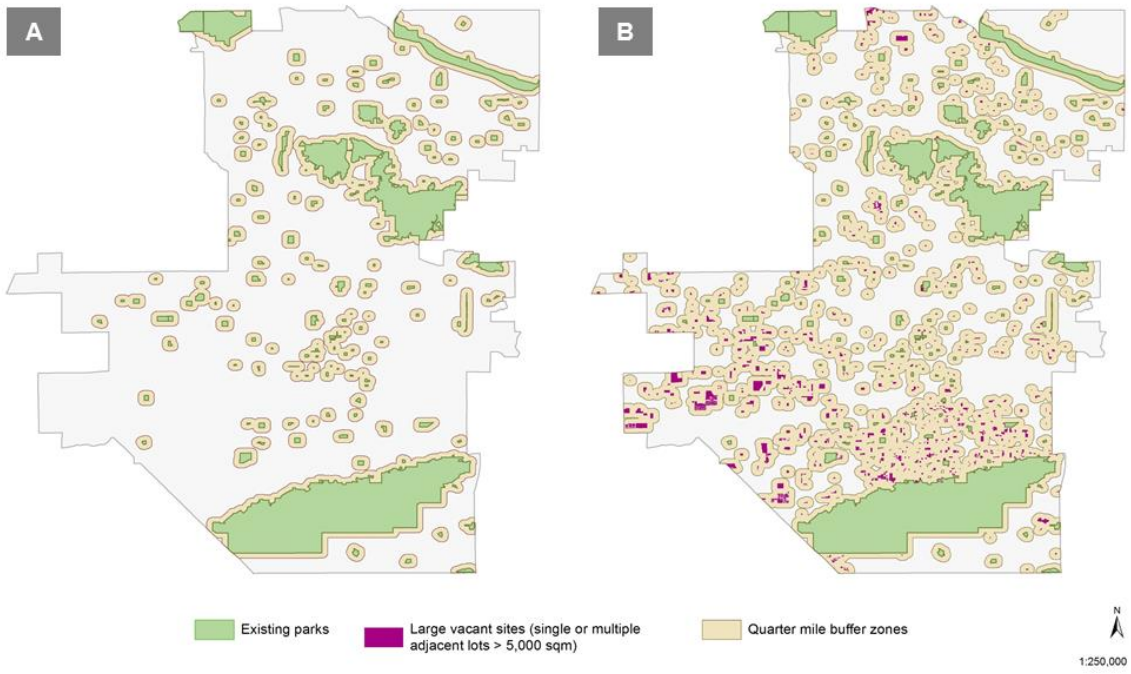


Figure 2.5. Walkability benefits from UA. (A) 5-minute walkability zones around existing public parks; (B) Comparable (non-overlapping) zones after UA conversion of vacant sites >5,000 m^2.

In our study area, existing parks overlap with block groups with a combined population of 306,773 (20% of the total population). The added green open space under prioritized deployment increases the number of residents served by 391,153, raising it to 48% of the total population. Not surprisingly, large vacant sites utilized for prioritized deployment tend to be in block groups with lower population density and fewer households (compared to the Phoenix median). To improve walkability in denser block groups, additional strategies to UA may need to be considered.

2.3.4 Energy savings and avoided emissions from buildings

In our analysis, the avoided energy (electricity) use and associated emissions reductions are sensitive to and vary non-linearly with the existing thermal insulation on rooftops (Ro values), since the insulative benefits from added UA substrate (Ra=0.25) are assumed to be constant. We consider three levels of insulation for existing roofs: well-insulated (high Ro = 3.66); average insulation, which is our baseline estimate (using average Ro = 1.84), and poorly insulated (low Ro = 0.25).

Typical Ro values likely vary across buildings and neighborhoods. To establish a range of values, we first consider all rooftops suitable for UA to have the same starting thermal insulative properties (Fig. 2.6A). Under these three scenarios, the annual avoided electricity consumption due to rooftop UA ranges from 33,712 MWh (using high Ro) to 3,859,475 MWh (low Ro), with a baseline value of 125,451 MWh (average Ro) (Fig. 2.6A). The lower and upper bounds represent 0.1% to 13% of electricity sales to commercial end-users in Arizona, and about 0.2% to 23% of total estimated electricity consumption in our study area. Corresponding CO_2 emissions avoided range from 13,778 to 1,577,315 tons, with a baseline estimate of 51,270 tons. The baseline represents 1.7% of commercial building GHG emissions in Phoenix in CO_2-equivalent terms (see Supplemental Information, sections 2.7.1 and 2.7.4 for details). At building-level, the baseline electricity savings we estimate are 6,237 kWh/year. This is about 3% of the annual site consumption of an average U.S. commercial building (U.S. EIA 2016a).

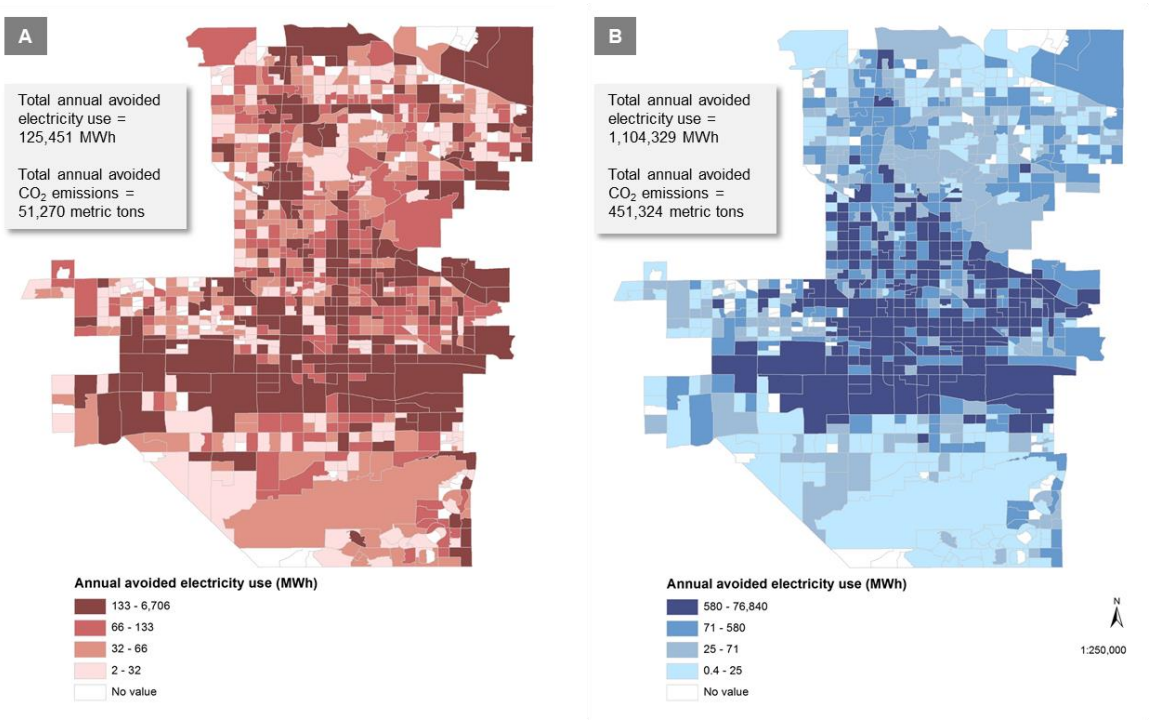


Figure 2.6. Avoided building electricity use from rooftop deployment of UA. Avoided building electricity use by block group quartiles in MWh per year. (A) Assumes same thermal resistivity (R-value) for all suitable rooftops at 1.84 (average). (B) Assumes thermal resistivity is inversely related to building age and assigns low (0.25), average (1.84), and high (3.66) R-values according to median building age per block group. In both (A) and (B) no value block groups have no suitable rooftop area for UA.

Poorly-insulated buildings are a driver of high energy burdens (share of energy expenditure in total spending). The median Phoenix household has a higher energy burden (4.2%) than average among a sample of 48 large U.S. metro areas (3.6%) (Drehobl and Ross 2016). On the other hand, Phoenix had the nation's highest share of EnergyStar certified new single-family homes in 2017 (59%; Energystar.gov). This suggests Phoenix likely has a mix of buildings with different insulative properties. Therefore, treating the savings from UA based on the average Ro as baseline is realistic.

Alternatively, we assign the high, average, low R_o values to block groups based on median year housing structures built in that block group (Fig.2.6B). The underlying assumption is that older buildings will have poorer insulation. This yields total avoided electricity use of 1,104,329 MWh and avoided emissions more than 451,324 tons (see Supplemental Information, section 2.7.7 for details). Accuracy of the energy and emissions savings estimation would improve with higher spatio-temporal resolution weather data: e.g. by computing hourly energy flux at a finer spatial scale (<1km).

2.3.5 Meeting Phoenix's desired outcomes

We summarize the contribution of UA to Phoenix's sustainability goals by revisiting Table 1 and adding our findings (Table 2.3, Results column).

UA can supply nearly 183,000 tons of fresh produce per year in Phoenix. This amount meets 90% of the current per capita fresh vegetable and fruit consumption in our study area. There is potential for locally-produced food from UA in all block groups that overlap with food deserts, constituting 55% of total UA food supply. Rooftops provide the largest area for UA in Phoenix. As we do not know the proportion of selected rooftops that are structurally suited to carry the incremental UA loads (substrate plus applied water), there is some uncertainty about the UA food supply from rooftops. Also, specific design and substrate choices could influence the magnitude of food supply and environmental benefits (Sanyé-Mengual et al 2015a; Sanyé-Mengual et al 2015b; and Dorr et al 2017), though we expect rooftop UA applications in Phoenix to use the current dominant technology (open-air and soil-based; Thomaier et al 2015).

Table 2.3

Contribution of UA to Phoenix's sustainability goals

City of Phoenix 2050 Sustainability Goals Category	Desired Outcomes	UA Delivery Mechanisms	Metrics	Results
Local Food Systems 	"Eliminating food deserts" (p.13)	Help reduce food deserts via local production of food	- Local food supply from UA - Total tons - Tons/census block - Tons/person	182,983 tons 201 tons/block group 0.15 tons/person
			- In block groups overlapping known food deserts - Total tons - Tons per food desert - Tons/person	101,008 tons (55% of total) 230 tons/block group 0.19 tons/person
Parks, Preserves and Open Spaces 	"Having all residents within a five-minute walk of a park or open space by adding new parks or open space in underserved areas." (p.10)	Create green open spaces by repurposing vacant lands as urban farms or community gardens	- Increase in green space area (%)	17%
			- Number of census blocks with public green space that formerly had none (#)	433 block groups
			- Increase in the 5-minute green open space access zones (% area and population served)	25% increase in area; 391.1 thousand additional residents served (28% increase)
Energy: Buildings and Land Use 	"Reduce carbon pollution from vehicles, buildings, and waste by 80%-90%" (p.3)	Create rooftop gardens that insulate buildings	- Avoided electricity use in buildings from added insulation provided by rooftop deployment of UA (MWh)	125,451 MWh; 0.4% of commercial electricity use in Arizona*
			- Avoided CO_2 emissions as a result of reduced building electricity use (metric tons)	51,270 tons of CO_2; 1.7% of reduction in 2012 Phoenix commercial building GHG emissions*

Reports the results for the metrics from Table 2.1. (*) Under baseline roof insulation assumptions.

Another desired outcome specified in Phoenix's 2050 Sustainability Goals is to have all residents within 5-minute walking distance of a park or open space by adding open spaces, especially in underserved areas. Through the use of vacant lots for UA, the number of underserved block groups is reduced by 433, and green open space area is increased by 17%. Using larger vacant sites to develop park-like UA spaces, walkability zones are expanded by 25% to cover 55% of the study area. The incremental green open space access zones provided by UA more than doubles the number of residents with access (391,153 in addition to 306,773 existing), representing 48% of population in our study area. For UA to contribute to improved walkability,

access rights to private property need to be negotiated between landowners and UA operators. Neither the UA operators, nor the landowners might be willing to seek or grant unlimited access to the general public. Landowners would be concerned with establishment of public rights over their land where there were none, and UA operators might be concerned with issues like food contamination and safety.

UA can also deliver energy savings and associated reductions in emissions from buildings. The desired outcome is an 80% to 90% reduction in emissions from transport, buildings and waste (from 2012 levels). The baseline savings through UA represents 0.4% of commercial sector electricity consumption for Arizona, and avoided emissions are 1.7% of GHG emissions from commercial buildings in Phoenix. Our baseline savings in energy use are within range of similar building-scale estimates (e.g. Wong *et al* 2003). The reductions are highly sensitive to the existing rooftop insulation, with substantial benefits for poorly insulated buildings. UA operations integrated with existing building systems have the potential to deliver larger savings in energy and emissions (Specht *et al* 2014). Other rooftop sustainability interventions, such as cool roofs can deliver higher reductions in cooling energy demand (e.g., up to 14% city-wide when all roofs are completely light-colored as in Salamanca et al 2016, and ranging from 14% to 30% at the building-scale based on Xu et al 2012 and Synnefa et al 2007). Yet, net benefits of cool roofs over the year tend to be lower because of increased heating needs during winter (Karl et al 2009, Georgescu et al 2014).

To some extent, savings are also limited by our assumptions. We consider flat rooftops greater than 464.5 m^2 (5,000 ft^2) that would be most suitable for commercial-scale UA, and not all flat rooftops. Likewise, possible variation in micro-scale temperatures is not captured in our analysis,

because of a lack of high spatio-temporal resolution weather data. We also do not account for additional building energy use from rooftop UA applications, which may reduce net benefits, thereby yielding lower desired outcomes than those specified here.

Even if the estimated benefit appears small in proportion to the desired outcome, it grounds expectations from the intervention considered (e.g., the scale of energy and emission savings achievable through UA), and underscores the need for alternative measures to achieve that goal.

2.3.6 Economic and policy considerations

Deploying UA as a sustainability strategy also requires an understanding of costs and trade-offs. For most urban areas, the availability and cost of suitable spaces for UA is the primary consideration and a binding constraint. Land tenure affects the level of investment made by a farmer. In a survey, urban farmers in the U.S. reported a preference for land ownership, but due to high land values, they often lease or acquire temporary user permits (Pressman 2016). Residential land prices can be used as a lower bound for the value of undeveloped or underutilized spaces allocated to UA. In our study area, they range from $217,000/ha to over $3.02 million/ha ($21.7/m2 to $302/m2) by census tract (Davis et al 2019). The average price of residential land in the Phoenix metro area ($720,419/ha) is lower than the average for the largest urbanized areas (population≥500,000) in the U.S. ($1,270,065/ha) (Davis et al 2019 and U.S. Census Bureau 2018b). Combining our block group-level UA area estimates with the census tract-level land prices, the weighted average cost of "space" for UA in our study area is $536,760/ha ($53.7/m^2). This can be treated as a one-time land acquisition cost, or converted into an annual rent of about $27,000/ha ($2.7/m^2), assuming a 5% capitalization rate (see Supplemental Information, section 2.7.4 for details).

In Phoenix, water is the critical non-land input for UA. We estimate that the cost of municipal water (at 0.64 $/m^2) would be about seven times the cost of fertilizer per square-meter of cultivated area. Water use in Phoenix also has a high social cost owing to limited fresh water resources available. The full deployment of UA in our study area would require 35,630,953 m^3 of water per year, adding about 7.2% to municipal water consumption in Phoenix. Yet, this is still less than 0.6% of total state-level water use, 74% of which is consumed by irrigated agriculture. Nonetheless, opportunities to recycle and reuse water for UA (i.e., incorporating UA into green infrastructure and vice versa) could limit further stress on scarce water resources. Changing residential water use patterns, such as eliminating turf grass and backyard pools, can improve water sustainability in Phoenix (Gober and Kirkwood 2010). Our estimate of UA water use is comparable to the amount needed to maintain mesic landscapes and backyard pools in Phoenix (see Supplemental Information, section 2.7.6).

Installation and operating costs for UA can vary highly with the scale and design of the actual UA operation and crop selection. Representative cost estimates come from detailed crop budgets for fruits and vegetables (e.g., https://coststudies.ucdavis.edu/en/current/) for the U.S. and surveys of rooftop garden applications. The one-time installation costs (excluding land) can range from under $10 (on vacant lots) to $30-$200 per square meter (for rooftop operations) and annual operating costs can range from $1/m^2 to $14/m^2 (Conner and Rangarajan, 2009; Acks et al 2006; Carter and Keeler 2008; and Blackhurst et al 2010). Use of permanent or semi-permanent structures (a hoop-house, fixed planters, fencing, cooler, or storage sheds) would lead to higher installation costs. Similarly, site preparation can raise upfront costs if any contamination remediation is needed, or the site needs re-zoning. Automated water and nutrient

delivery systems would also increase installation costs, but might reduce operating (labor and material) costs.

Local governments and residents also need to consider the opportunity costs of city-scale UA with regard to tax tradeoffs, provision of local food versus affordable housing, rooftop solar energy, cool roofs and other development strategies.

Removing barriers to UA could include strategies such as clarifying zoning for intra-city farming activities that are conducted for profit versus not-for-profit (subsistence farming, educational farms) and expanding building codes to codify structural requirements for adding live substrate on rooftops.

Some jurisdictions also consider policies that create explicit incentives for commercial UA. These may target property taxes. For example, California's AB551 (2014) legislation allows (private) landowners in metropolitan areas to receive tax incentives for placing their land in agricultural use. In Utah, SB122 (2012) allows qualified parcels in Salt Lake County to be assessed at lower property rates provided the land is used as for-profit farming. Purchase requirements that support local farmers may also benefit UA. For example, in Illinois HB3990 (2009) calls for 20% of all food products purchased by state agencies and state-owned facilities to be local farm or food products by 2020.

2.4 Conclusions

We combine fine scale data on available urban surfaces (undeveloped vacant lots, façades and rooftops) with Phoenix- or Arizona-specific inputs to determine the contribution of UA towards meeting Phoenix's desired sustainability outcomes. Our findings show that the total UA food

supply in the fifth largest city in the U.S. meets the annual consumption of fresh produce based on national per capita consumption patterns. UA can also supply local produce in all food deserts of Phoenix, and more than half of all output would be produced in the half-mile LILA census tracts identified by the USDA's widely-used national assessment of food deserts. This assessment likely overestimates food access in poor neighborhoods because of spatial aggregation (Bao and Tong, 2017), meaning locally produced food from UA might prove more impactful than we estimate here.

UA would also add green open space for Phoenix residents. Full deployment of all available vacant lots can reduce number of underserved block groups by 60%. Larger vacant sites converted into UA could include park-like features, which would bring 55% of the study area within 5-minute walking distance to a green open space (up from 30%). Rooftop deployment of UA could reduce electricity use and associated CO_2 emissions from buildings and displace more than 50,000 tons of CO_2 based on average roof insulation.

An advantage of UA as a sustainability solution is its potential to improve and encourage multi-purpose utilization of existing capital stock (infrastructure, buildings) within an urban area. In addition, there are several co-benefits from UA that are not explicitly evaluated here. These include public health benefits (possible reduction in obesity rates and diet-related disease) to the extent such potential health and nutritional benefits are a function of improved food access; benefits from shortened supply chains such as lower emissions from transport of produce (i.e., food miles); and local dust mitigation and runoff control from cultivation of otherwise underutilized vacant lots.

Today, the fastest growing cities in the world are those with fewer than 1 million residents, rather than megacities (>10 million residents). By 2030, more urban residents will live in small to mid-size urban areas (500 thousand to 5 million residents), similar to Phoenix. In the U.S. alone, the number of cities with 1 to 5 million residents will grow 25% by 2030 (UN 2018). It is therefore important to examine sustainability solutions in small and mid-size city settings.

Yet, UA cannot be a sustainability panacea for all cities, and we are not advocating that urban areas should be used for growing crops. Our goal is to demonstrate the value of a data-driven approach that involves using fine-scale, local data to evaluate UA. Ultimately a database of comparative city-scale assessments is needed to show why UA can be a sustainability solution in one context and not others (Webster 2018).

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2.7 Supplemental information

2.7.1 Methods

Available area estimation for UA: Our approach for determining suitable vacant lots has two stages. First, we identify privately-owned vacant properties based on cadastral data. We then refine this initial inventory using high-resolution imagery (NAIP) and a digital elevation model using GIS and spatial statistics (Smith et al 2017). We exclude built vacant properties or paved vacant lots to avoid the additional cost of converting them into UA sites and count only undeveloped vacant lots. The 2017 inventory informs our baseline estimate and the 2010 assessment is used as a sensitivity.

Prior to computing available rooftop and vertical area using building footprint data from LiDAR, redundant features associated with building footprints (like awnings, overhangs, chimneys, etc.) are eliminated to prevent an overestimation of available area.

For rooftop surfaces, we consider only flat rooftop buildings with base roof area greater than 464.5 m^2 (5,000 ft^2). Existing commercial rooftop UA operations in Europe and the U.S. point towards the use of larger rooftops to facilitate food production at scale (Buehler and Junge, 2016 and Berger, 2013). Larger rooftops are also more likely to have the structural stability for supporting incremental loads from UA substrate, although we do not treat this as a binding constraint in our analysis. Our sample therefore consists of medium- and large-size commercial / industrial buildings, and excludes most single-family homes, producing a conservative estimate for the available rooftop UA area.

To estimate potential area from building façades, we multiply the average of length and width for each building with an assumed vertical cultivable surface of 1.83 meters (6 feet). Cultivation is

limited to one building side, based on the location of services (e.g. utility hookup) and appropriate sun exposure. This is an agnostic approach, as we do not restrict UA deployment to a specific side, assuming that would be the most advantageous for a façade application. However, we also considered an alternative estimation that is exposure-specific. We estimated the same area using building sides with either south or east exposure, although in Phoenix the south exposure may be undesirable even in the cooler months. Available area is higher by 10% using a south-facing exposure, and by 4% using an east-facing exposure. However, due to excessive radiation loading for south-facing exposure in our study area, the added area will likely not increase crop yield or provide higher benefits. We therefore view our base case assumption as more realistic.

Building height does not factor into our estimation of available vertical and rooftop areas. This means we have not eliminated few very short or very tall structures (that meet other criteria) where height might adversely affect crop viability (e.g. on rooftops due to high winds).

Crop selection and area allocation: There are three steps to crop selection and area allocation. The first step is an initial screening of the crops. We start with a list of crops that are grown in Maricopa County commercially (for which data is available). This excludes crops grown on hobby or educational farms which may be experimenting with different crops/varietals.

We then eliminate from contention some types of crops due to their physical attributes and potential space and time (crop growth) limitations in a UA setting. Specifically, we exclude perennial tree crops (citrus, other fruit trees), because they would be difficult to grow in an UA setting either due to the medium chosen which may limit rooting depth (e.g. rooftop, vertical) or due to land tenure concerns. We also exclude grains like corn, process-intensive oil crops like

sunflowers, feed crops like alfalfa, and non-edible fiber crops like cotton, as well as ornamental crops, since the purpose of UA is to supply food, and mainly fruits and vegetables. We end up with 34 crops that are mostly annual vegetable crops, and a few fruits.

The second step is to rank these 34 crops to determine area allocation. The decision to change land use and/or utilize underutilized spaces associated with buildings (façades, rooftops) will be based on economics. The ranking approach is used to approximate this decision. In absence of detailed estimates for cost, we use water consumption of crops as a proxy for operating (input) costs. Combined with (higher) yields and (higher) prices, the preference lower water use is meant to be a proxy for profitability. We are not suggesting these are the only preferred qualities, but considered the combination of the three to be a practical benchmark. In the case of Phoenix, a preference for crops with lower water use is a sustainability indicator as well.

We then group the 34 crops into terciles based on their rankings. The highest ranked crops are in the top (3^{rd}) tercile, and lowest ranked crops are in the bottom (1^{st}) tercile. Finally, we allocate area proportionally: i.e., larger proportion of area is allocated to crops in the top tercile, and smaller proportion is allocated to crops in the bottom tercile. For example, for a 0.6 ha-sized UA lot, 0.3 ha would be allocated to top tercile crops, 0.2 ha to the second or middle tercile, and 0.1 ha to the bottom tercile.

Below is the list of crops assumed suitable for UA in Phoenix (N=34). The underlined crops (N=22) were assumed suitable for cultivation along building façades. The suitability categories of good, average and poor correspond to the terciles 3, 2, and 1 described above.

Suitability for urban agriculture in Phoenix	Good	Average	Poor
	Brussels Sprouts	Artichokes	Broccoli
	Herbs	Beans, Snap	Cabbage
	Lettuce, Head	Beets	Cauliflower
	Lettuce, Romaine	Carrots	Cucumbers
	Onions, Dry	Celery	Eggplant
	Onions, Spring	Greens, Collard	Melons, Cantaloup
	Parsley	Greens, Kale	Melons, Watermelon
	Spinach	Greens, Mustard	Peppers, Chile
	Squash	Lettuce, Leaf	Potatoes
	Strawberries	Melons, Honeydew	Pumpkins
	Tomatoes	Peppers, Bell	Radishes
			Turnips

Figure 2.7. UA crops used in analysis by suitability category. Underlined crops represent the subset assigned for UA applications along building façades.

Figure 2.7 shows the dispersion in crop yields (tons/ha), water use (mm/m^2) and retail prices ($/ton) for the crops selected for UA in Phoenix by suitability category (described above). Across the three suitability categories, we observe less dispersion in yields, and more variation with respect to prices and water use. Overall, crops in the good category have higher median yields; lower median water use, and retail for higher median prices. The converse is true for the crops in the poor category.

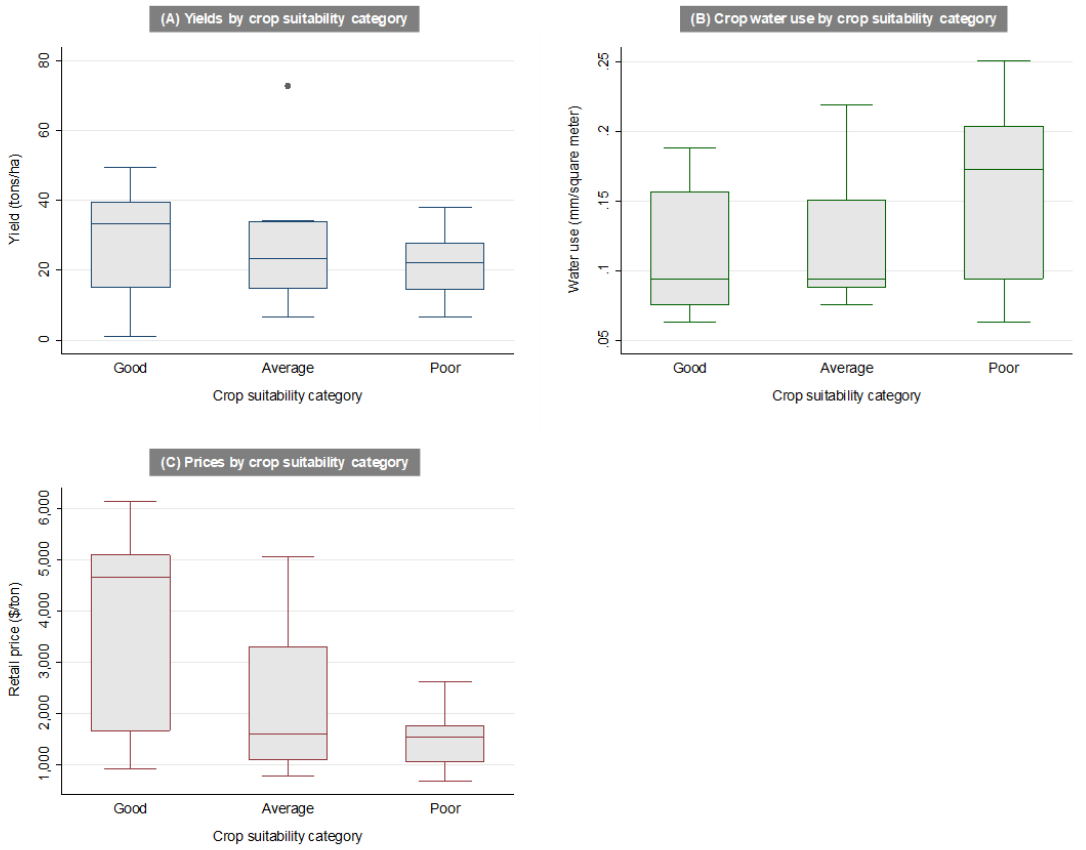


Figure 2.8. Box plots of unit yields (A), water use (B), and prices (C) for UA crops by crop suitability category. A box plot, or a box-whisker plot is used to show dispersion in data. The box portion bounds the values that lie between the 25th and 75th percentiles for each group (also known as the interquartile range, or IQR). The dividing line in each box shows the median for that group. The whiskers show the upper and lower adjacent values which are calculated as (1.5*IQR)+75th percentile and 25th percentile-(1.5*IQR), respectively. Any values outside these ranges are indicated by a marker, indicating the presence of outlier(s).

We use the following data sources for crop-specific yields, water use, and retail prices. Historical county-level yields are from the USDA annual surveys (2000-2016) for most crops, or the last two observations reported in the Census of Agriculture (2007 and 2012), if annual data is not available. We substitute with comparable county-level data from Southern California, and in absence of that, national yields when yields for Maricopa county are not reported. If a crop (e.g. cucumber) is grown both for fresh market sale or processing, we use fresh market yields.

Crop-specific water needs are obtained from the UN FAO (1986) and two university cooperative extensions (Masabni 2011 at Texas A&M and the UC Davis Vegetable Research Information Center).

Urban farmers can have a wide range of farming skill and experience (Waldman *et al* 2010). Here, we conservatively assume the average urban farmer to be less experienced in farming than their commercial counterparts. Therefore, when calculating output, we use the minimum of historically reported yields. For the same reason, we use the higher estimates for crop-specific water use (if more than one estimate is available).

Crop retail prices are the 2016 averages from regional (in our case, the Southwest) supermarket surveys conducted weekly (USDA AMS 2016).

Open green space provision: First, we determine existing open green spaces, which include all public parks within or adjacent to the study area. Golf courses and school yards are excluded for limited access reasons. We then compute total park area by block group. Where large regional parks overlap or are adjacent to multiple block groups, park area is allocated proportional to block group area.

The size threshold we use for the prioritized deployment case (5,000 m^2) is based on the distribution of existing park sizes in Phoenix. More than 95% of all parks in our study area are larger than 5,000 m^2. However, this cutoff would exclude many proximate (adjacent) small vacant parcels if evaluated individually. Therefore, if such parcels are located within 10m of each other (which is narrower than representative street canyons in Phoenix; see Hedquist and Brazel, 2014) and their aggregation is greater than 5,000 m^2, they are counted towards the sub-sample of vacant lots used for prioritized deployment.

We refer to these larger vacant lots (either on an individual or aggregated basis) as "vacant sites".

We utilize these vacant sites (which are sub-sample of all available vacant lots for UA) in a walkability analysis. First, we define walkability zones as green open spaces plus a 402-meter (0.25 mile) surrounding buffer. This buffer corresponds to 5-minute walking distance at approximately 4.83 kilometer/hour (3 mile/hour) average walking speed for a typical adult (Browning *et al* 2006). We then compute these buffers for existing parks and then again for the vacant sites. Finally, we measure the change in total area at the study area-scale before and after UA deployment.

Energy savings and avoided emissions from buildings: We assume electricity is used for both heating and cooling to maintain indoor temperatures between 21°C (70F) and 25°C (77F). Across the U.S. and in Arizona, electricity is the dominant fuel for cooling. For space heating, 3 out of 5 Arizona households rely on electricity, and so do nearly half of commercial buildings in the Western U.S. (U.S. EIA 2016b and 2019).

We then derive temperature gradients between these target indoor temperatures and air temperature data for 2017 using NASA's GLDAS-2.1 dataset that has a 3-hour temporal and 0.25 arc degrees spatial resolution (Rodell 2004). Summing the 3-hourly gradients over a year produces cumulative ΔT (heating and cooling needs) in Kelvins (K).

Thermal resistance for rooftops is represented by R-values in m^2 K/W. Three different R-values are used to characterize existing insulation on rooftops (R_o), before any UA substrate is added. These are 1.84 (average, for average roof insulation; representing the baseline), 0.25 (low, for poorly insulated roofs) and 3.66 (high, for well insulated roofs).

These values are based on observed values reported in the literature (e.g., Castleton et al 2010; Clark et al 2008; Niachou et al 2001; Vacek et al 2017; Van Hooff et al 2014 and Wong et al 2003). For UA, a 10cm thick, high moisture substrate with an R-value (R_a) of 0.25 is added onto roofs (Sailor *et al* 2008). Since thermal resistivity of layers are additive, roofs with UA have higher R-values ($R_o + R_a$).

Avoided energy use due to UA (kWh/block group/year) =

$$[A_{2j}*(\sum_{t=1}^{2920} \Delta T /R_0)*3/1000] – [A_{2j} * (\sum_{t=1}^{2920} \Delta T/(R_0 + R_a))*3/1000] =$$
$$A_{2j} * \sum_{t=1}^{2920} \Delta T * R_a/(R_0(R_0+R_a))*3/1000 \quad (4)$$

Where:

A_{2j} = Available rooftop area per block group j (m^2)
Unit flux (W, watts) = $A*\Delta T/R$
ΔT = Difference between 3-hourly (t) outdoor and indoor target temperature (K)
t = period in GLADS-2.1 = [1,2920]
3/1000 = factor to convert to kWh per year
Indoor temperature targets for Phoenix = 21°C (70F) for heating, 25°C (77F) for cooling
R is thermal resistance of roof (m^2K/W):
$R_0 = R_{existing\ roof}$ = 1.84 (average); 0.25 (low); or 3.66 (high)
$R_a = R_{UA\ substrate}$ = 0.25 (10cm thick, high moisture substrate)

Lastly, we convert resulting quantities to kWh/m^2 and multiply by block group area (m^2) to obtain electricity used for heating and cooling in kWh per block group.

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2.7.2 Local fresh produce supply from UA: sensitivity analysis

In our baseline analysis, we use a set of crops that are already grown in Maricopa county (where Phoenix is situated), excluding from that list crops we deem not suitable for UA applications (such as tree crops, fiber crops, etc.). In the sensitivity analysis, we do not deviate from this crop list and maintain the same approach -i.e. ranking according to one or more criteria and then grouping in terciles- so that the sensitivities are comparable to the base case.

We consider two scenarios that alter the ranking and therefore the area allocation for these crops in a UA setting:

(1) The "high productivity" scenario: Rank and then group into terciles our selected crops according to yield alone (productivity).

(2) The "national consumption pattern" scenario: Rank and group into terciles our selected crops based on each crop's consumption (in primary weight per capita terms) using the 2016 national per capita food availability data from the USDA. If a crop that we selected as suitable for UA in the City of Phoenix does not appear in this summary, which focuses on the most sold individual crops, we assign them the lowest ranking.

Table 2.4 shows how the suitability category for the crops we selected for UA application vary by scenario. For example, in the base case, potato is ranked 1 (poor) because it is among the low-value / high-water use crops in our list. In the high productivity sensitivity, this crop moves up a rank (2) to the average category, because it is in the middle tercile purely on a kg/ha basis. In the national consumption pattern sensitivity, it moves up to the good category (rank 3) because it is the most consumed vegetable in the country on a (primary) weight per capita basis. (Recall that rank 3 = good, rank 2 =

average, rank 1 = poor; and crops ranked 3 get 3 times the area allocated to them than those ranked 1.)

Table 2.4

UA crops by suitability category: base case and two sensitivities

Selected crops		Base case	High yield sensitivity	National consumption pattern sensitivity
1	Artichokes	Average	Poor	Average
2	Beans, Snap	Average	Poor	Average
3	Beets	Average	Good	Poor
4	Broccoli	Poor	Poor	Good
5	Brussels Sprouts	Good	Average	Poor
6	Cabbage	Poor	Good	Average
7	Carrots	Average	Good	Good
8	Cauliflower	Poor	Average	Average
9	Celery	Average	Good	Average
10	Cucumbers	Poor	Average	Good
11	Eggplant	Poor	Average	Poor
12	Greens, Collard	Average	Poor	Average
13	Greens, Kale	Average	Average	Poor
14	Greens, Mustard	Average	Poor	Poor
15	Herbs	Good	Poor	Poor
16	Lettuce, Head	Good	Good	Good
17	Lettuce, Leaf	Average	Average	Good
18	Lettuce, Romaine	Good	Good	Good
19	Melons, Cantaloup	Poor	Average	Average
20	Melons, Honeydew	Average	Average	Average
21	Melons, Watermelon	Poor	Average	Good
22	Onions, Dry, Summer, Non-Storage	Good	Good	Good
23	Onions, Spring	Good	Good	Poor
24	Parsley	Good	Poor	Poor
25	Peppers, Bell	Average	Good	Good
26	Peppers, Chile	Poor	Poor	Poor
27	Potatoes	Poor	Average	Good
28	Pumpkins	Poor	Average	Average
29	Radishes	Poor	Poor	Poor
30	Spinach	Good	Poor	Average
31	Squash	Good	Average	Average
32	Strawberries	Good	Good	Good
33	Tomatoes	Good	Good	Good
34	Turnips	Poor	Poor	Poor

The food supply from UA is higher under both sensitivities (Table 2.5). This is not surprising because we relax the constraint on water use and crop value (price), and sort and group crops along a single dimension (yield in one case, and national per capita consumption in the other). Moreover, both sensitivities result in higher water use, but the high yield sensitivity exhibits a smaller increase in unit water use –i.e., response—per each incremental ton of output. On the other hand, the national consumption pattern results in water use response of over 100%. This means for each ton of incremental output in that scenario requires a higher unit use of water.

Table 2.5

Summary results for the food supply sensitivity analyses

Total annual food supply from UA by scenario	Food supply				Water use		
	Tons	Tons per ha	Kg per person	Increase over base case (%)	Water use ('000 m^3)	Increase over base case (%)	Water use response (%)
Baseline	182,983	25.3	124.9		35,631		
High productivity	211,625	29.3	144.5	16%	39,741	12%	74%
National consumption pattern	194,633	26.9	132.9	6%	38,628	8%	132%

Food supply by scenario in terms primary weight. Water use response is the percentage change (increase) in water use divided by percentage change (increase) in food supply over the baseline.

2.7.3 Local fresh produce supply from UA: calorie analysis

We also add a measure of calories to the original measure of weight to describe food supply from UA. We start with the existing primary weight measure (tons/year), which is the harvested amount for the selected set of fruits and vegetables under UA. We then account for food losses along the supply chain and determine the edible weight per crop. These losses occur when going from producer to retailer and from retailer to consumer. The consumers then separate food items into edible and non-edible parts (discarded). The last stage is cooking loss and uneaten food (waste).

Crop-specific loss factors (for year 2016) and edible shares are obtained from the USDA's Food Availability (Per Capita) Data System (USDA 2018a). After accounting for losses and determining the edible weight by crop, we combine this with the calorie and cup-equivalency data from the USDA's Nutrient Database for Standard Reference Release (USDA 2018b) for each crop (in terms of kilocalories per 100 grams of edible portion). For six crops in our list (out of 34), there were no corresponding loss factor or edible share estimates in the USDA data. For those, we used the average of existing factors. Also, calories can vary by how each food item is prepared. We make no assumptions concerning that, and use calorie data for the fresh/raw fruit and vegetable. We then derive per capita/day calories based on study area population.

We estimate daily calories for the base case and the two food supply sensitivities. We also consider that retail losses might be lower in the case of UA due to shortened supply chain, and conduct a sensitivity where we assume no retail losses (i.e., from retailer to consumer). In the U.S., total losses accumulated along the conventional supply chain

average 51% for vegetables and 57% for fruits (share of primary weight, 2016 figures from USDA 2018a). For our crop mix, the comparable mean loss rate is 60%; ranging from a low of 38% (tomatoes) to a high of 83% (pumpkins). If we assume away the retail losses, the average loss factor decreases to 46%.

Table 2.4 provides summary results from this analysis. Based on primary weight, the base case per capita output from UA represents 90% of the national average consumption of fresh fruits and vegetables. Under the two food supply sensitivities, food supply from UA is higher and closer to existing national per capita consumption of fruits and vegetables. On a loss-adjusted basis, the ratio of per capita UA food supply relative to the national average is slightly lower (86%, base case) because of the higher UA supply of some crops with higher loss rates (e.g. artichokes, melons).

Estimated food supply from UA yields fewer calories compared to the national averages from fruits and vegetables (driven mainly by our overall less use of calorie-dense crops like potatoes). Daily calories from UA food supply range from 42 kcal/person/day under the base case, to 60.5 kcal/person/day under the high productivity sensitivity and assuming no retail losses. Nationally, the average daily calories from the consumption of fresh fruits and vegetables are estimated at 92 kcal/person/day.

However, we also calculate the cup equivalent measures (using conversion factors from USDA 2018b) which form the basis of recommended fruit (2 cups) and vegetable (2.5 cups) daily intake in the Dietary Guidelines for Americans (U.S. Department of Health and Human Services and U.S. Department of Agriculture 2015). On a cup-equivalent basis, the food supply from UA is higher than the national average under the base case

(1.38 vs 1.40), as well as under all the other scenarios we consider (Table 2.6, final column). Yet, even under the high productivity scenario, and assuming no retail losses, the UA food supply would meet only the half of the recommended daily intake amounts on a cup-equivalent basis.

Table 2.6

Per capita availability of fresh produce

	Food supply scenario	Losses	Weight (kg/person/year)		Daily calories (kcal/person/day)	Cups per day (cup equivalent per person/day)
			Primary	Loss-adjusted	Loss-adjusted	Loss-adjusted
Study area averages under UA	Base case	Conventional	124.9	55.6	42.0	1.40
	High productivity		144.5	64.1	48.8	1.55
	National consumption pattern		132.9	58.8	45.4	1.42
	Base case	No retail loss	124.9	71.0	53.0	1.86
	High productivity		144.5	80.6	60.5	2.00
	National consumption pattern		132.9	75.7	57.6	1.89
National average		Conventional	138.7	65.0	92.0	1.38

Summary results from UA and comparison with U.S. averages by primary and loss-adjusted weight (kg/person/day), daily calories and cups equivalent per day. National consumption figures based on per capita food availability data from the USDA (2018a) for fresh vegetables and non-citrus fresh fruit. In the Losses column, “conventional” means all losses along the traditional food supply chain are included; “no retail loss” means we assumed away losses when going from retail to consumer.

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2.7.4 Other data used in the analysis

Income, housing characteristics and demographics: We use data from the American Community Survey (ACS) 2012-2016 5-year estimates (U.S. Census Bureau 2018c) for block group level data on population, income, home values, receipt of supplemental nutrition assistance (SNAP) benefits and age of residential structures.

Food deserts: We use the USDA's latest national assessment of food deserts based on income and access to retail food outlets (ERS 2015 and Rhone et al 2017). All the retail food outlets included in the USDA's assessment of food access sell fruits and vegetables (Ver Ploeg et al 2012).

We use the half-mile measure that qualifies a food desert as a low-income census tract with a significant number (at least 500 people) or share of the population (at least 33%) that is more than 0.5 mile (0.8 kilometer) from the nearest supermarket, supercenter, or large grocery store for an urban area. A low-income census tract has either (i) poverty rate >20%; or (ii) median family income ≤ 80% of the state-wide median family income; or (iii) median family income ≤ 80% of the metropolitan area median family income. The USDA's definition of food deserts is a widely used measure of food access in the U.S. This is relevant to our analysis, because the City of Phoenix's sustainability goal under local food systems is the "elimination of food deserts", and the city also utilizes the same USDA definition of food deserts.

Buildings and energy: As benchmarks for the results from the buildings and energy analysis, we use Arizona-level data on electricity consumption by end-use sector and CO_2 emission rates from electricity generation (U.S. EIA 2018b). We also rely on the City of

Phoenix's 2012 inventory of city-wide GHG emissions (City of Phoenix 2017b). To estimate total electricity consumption in our study area, we calculate per capita electricity consumption for the entire state and then scale this up by the study area population. These are summarized in Table 2.7.

Table 2.7

Reference figures for contextualizing avoided energy and emissions estimates from rooftop deployment of UA

Sales to Ultimate Customers (MWh) by Total Electric Industry by Sector, Arizona		
Residential	33,690,783	a
Commercial	29,564,157	b
Industrial	14,975,656	c
Transportation	7,232	d
Total	**78,237,828**	e=a+b+c+d
Net electricity generation, Arizona (MWh)	108,763,449	f
Total CO_2 emissions from power generation, Arizona (thousand metric tons)	44,531	g
CO_2 emissions rate (tons/MWh)	0.4094	h=(f*1000)/e
CO_2 emissions rate (lbs/MWh)	901	h*2204.62
GHG emissions in Phoenix (2012) from commercial buildings, thousand metric tons of CO_2eq (*)	3076	
Arizona population, 2016 (**)	6,909,000	k
Per capita electricity consumption (kWh)	11,324	l=(e/k)*1000
Total population in study area (**)	1,464,694	m
Estimated electricity consumption in study area (MWh)	16,586,261	n=(m*l)/1000

All energy and emissions figures are for year 2016 and from the EIA (2018b), except (*) from the City of Phoenix (2017b) and (**) from the U.S. Census Bureau (2018c).

Land prices: We use census-tract level residential land price data from Davis et al (2019) to derive indicative cost of space for UA. Davis et al (2019) interpolate residential land prices in the U.S. over a period from 2012 to 2017 using data on 16.4 million home mortgage appraisals at the county, zip code, and census tract level. The study covers 85%

of the population and 83% of all single-family homes in the country. This data overlaps with 756 (out of 910) block groups in our study area. The summary statistics cited in the main text are based on this subset of block groups but also for the Phoenix-Mesa, AZ metro area for comparison with other urban areas (Table 2.8). For that comparison, we pair Davis et al (2019) data with data on urban areas from the Census Bureau (2018c) to calculate average land prices for urban areas with populations are larger than 500,000.

Table 2.8

Average residential land price estimates for various areas of interest

	Mean residential land price	
Coverage	**$/acre**	**$/ha**
Study area	305,015	753,707
Phoenix-Mesa, AZ metropolitan area	291,543	720,419
All Arizona	127,025	313,885
All U.S. urban areas with population >= 500,000	513,977	1,270,065

Summarizes average residential land price estimates from Davis *et al* (2019) using 2012-2017 pooled cross-section data. The averages are calculated over census tract-level estimates.

We do not know which vacant lots we identify would be candidates for residential or commercial (including industrial) development (in absence of UA). Commercial land values can diverge from residential land values for economic reasons. Residential property values reflect locational amenities valued by households (good schools, availability of employment opportunities, recreation and transportation options; low crime, etc.). Commercial property values also depend on location, but are more sensitive

to the proximity to roads, airports, and railroads; nearby complementary businesses; and the income and demographics of the local population (Nichols *et al* 2010).

Albouy *et al* (2018) estimates a combined average for residential and commercial land prices for the Phoenix metro area at $1.1 million/hectare. This is higher than the residential-only average land price based on Davis *et al* (2019). Hence, the cost of land for UA we calculate can be treated as a lower bound.

Table 2.9 shows the urban land value estimates for a select number of US metro areas for 2005-2010, adapted from Table 2 of Albouy et al (2018). Values for Phoenix are highlighted in yellow. Average urban land values for Phoenix metro area is estimated to be $452,000 per acre, or $1.1 million per hectare. For the quarter mile zone around the downtown areas, the land values jump to $3.5 million per acre, or $8.7 million per hectare. This study finds New York City to have the most expensive land, and San Francisco ranks fourth. The study reports land values over $300 million per hectare in the heart of New York City, and $13 million per hectare for the metro area. For San Francisco, values in the city center average over $40 million per hectare, and average over $8 million for the entire metro area.

Average residential land value from Davis et al (2019) for the comparable area (Phoenix metro) is $720,748/ha. Note that this average is based on census tract level estimates; covers a later period than Albouy et al (2018); and is estimated using a different method, a different data source, and with a larger sample size. With those caveats in mind, we note that the average estimate for bundled land values from Albouy *et al* (2018) is higher than the estimate for residential lands alone. If we assume the two studies are

comparable, then we can conclude commercial land values are higher than residential land values. In that case, the weighted average cost of land for UA that we calculate in the main text can be considered a lower bound.

Table 2.9

Selected U.S. metro-area level land values

				Land Values ($000s/Acre)			*Land Values ($000s/hectare)*		
Rank	Metro area name	Total urban area (sq miles)	Number of land sales	Simple average	Estimated average value for the entire urban area	Estimated value for the city center	Simple average	Estimated average value for the entire urban area	Estimated value for the city center
1	New York, NY	749	1,603	26,139	5,264	123,335	64,591	13,008	304,767
2	Jersey City, NJ	47	43	7,667	3,305	9,554	18,946	8,167	23,608
3	Honolulu, HI	198	56	4,357	3,290	16,256	10,766	8,130	40,169
4	San Francisco, CA	300	152	8,722	3,239	25,446	21,553	8,004	62,878
5	Los Angeles–Long Beach, CA	1,359	1,760	3,709	2,675	16,801	9,165	6,610	41,516
6	Orange County, CA	494	233	3,163	2,595	3,208	7,816	6,412	7,927
7	San Jose, CA	305	217	2,580	2,347	3,552	6,375	5,800	8,777
8	Miami, FL	372	1,233	3,052	1,794	4,478	7,542	4,433	11,065
9	Stamford-Norwalk, CT	179	19	2,753	1,505	2,740	6,803	3,719	6,771
10	Bergen-Passaic, NJ	316	79	1,957	1,423	4,145	4,836	3,516	10,243
16	Washington, DC-MD-VA-WV	1,458	1,840	3,548	1,214	36,913	8,767	3,000	91,214
22	Las Vegas, NV-AZ	317	2,553	1,193	849	1,841	2,948	2,098	4,549
26	Chicago, IL	2,035	3,511	1,455	663	37,632	3,595	1,638	92,991
27	Boston, MA-NH	1,295	122	1,243	600	8,457	3,072	1,483	20,898
32	Denver, CO	536	2,015	828	539	7,586	2,046	1,332	18,745
52	Phoenix-Mesa, AZ	897	5,946	370	452	3,529	914	1,117	8,720
99	Dallas, TX	1,057	811	454	305	2,774	1,122	754	6,855
118	Houston, TX	1,341	1,143	423	272	2,813	1,045	672	6,951
120	Detroit, MI	1,426	679	456	270	2,321	1,127	667	5,735
130	Atlanta, GA	2,105	5,229	402	251	1,750	993	620	4,324
227	Pittsburgh, PA	1,003	240	433	156	1,772	1,070	385	4,379

Shows metro areas are ranked by average (estimated) urban land values. Here, the top ten based on Albouy *et al* (2018)'s analysis and a selection of other large metro areas for comparison are included. Land value data from CoStar COMPS database for years 2005 to 2010. Simple average is based on raw, observed prices per acre. The two estimated value columns are Albouy *et al* (2018) model results. The first is the estimated average land value for the entire urban area. The second reports estimated land values within ½ mile radius from downtown (or multiple downtowns). The data is pooled for all types of land sales. Dollars per hectare ($/ha) conversion involves multiplying $ per acre values with 2.47105, and are done by us. Values for Phoenix metro area are highlighted.

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2.7.5 Change in available vacant lot area between 2010 and 2017

Change in undeveloped vacant lot area between 2010 and 2017 was calculated for 562 block groups in the study area. We record more block groups with a decline in vacant lot area (N=345, red) than those with an increase (N=161, blue) (Figure 2.9). For 56 block groups (yellow), the absolute change in vacant lot area was less than $50m^2$ -- an amount small enough to constitute measurement error. Block groups that had no undeveloped vacant lots inventoried in either year (N=280; grey) were excluded from this analysis, as well as a subset of block groups (N=68; striped) that had data comparability issues.

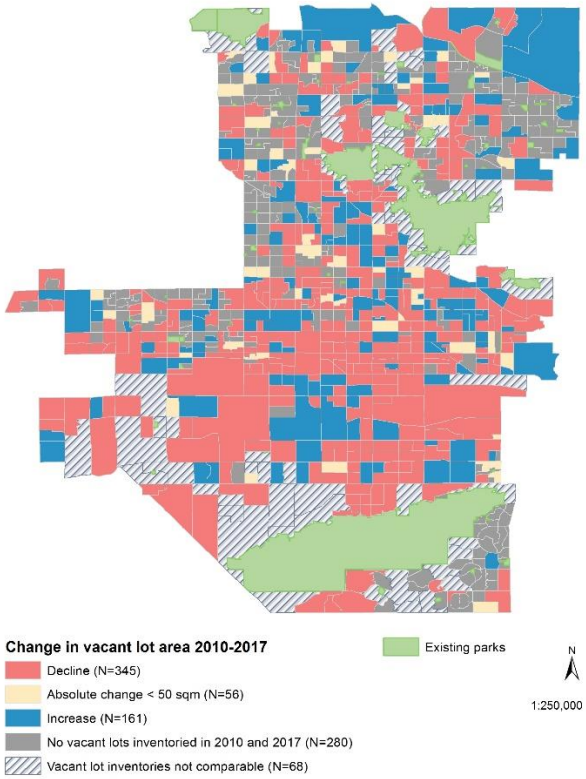


Figure 2.9. Change in available vacant lot area for UA between 2010 and 2017

There is some evidence of clustering in the changes by block group (using Moran's I spatial autocorrelation measure, $p<0.05$). Targeting areas where vacant lots persist--determination of which requires more than one change period and thus beyond the scope of this paper--might be a strategy to achieve sustainable contributions from UA, even if this means the area deployed for UA would be lower than the maximum possible.

2.7.6 Water use and cost

Water use by UA: We assume UA applications use municipal (potable) water to take advantage of existing urban infrastructure. We are not converting land or building surfaces with existing irrigation water use into UA, therefore, estimated UA water use is incremental to current municipal water use in the City of Phoenix.

Total estimated annual crop water use by UA across all three types of surfaces is 35,630,953 m^3 (or 9,412,706 thousand gallons). This is calculated as follows:

Total water use (gallons) = Total acres by crop (acres) * Water use by crop (inches / acre) * 27,160 (gallons / inch)

Here, the total area planted to each crop is estimated as part of our analysis. We use independent estimates of water use by each crop during growing season (see section 2.7.1). The last term is a constant representing the volume of water that would cover an area of 1-acre at 1-inch deep.

We then convert gallons of water use by crop into cubic meters by crop (where 1,000 U.S. gallons are equivalent to 3.78541 cubic meters). Lastly, we sum water use across all 34 crops. The amount of water used is specific to the baseline crop selection and area allocation, and would change under alternative configurations (e.g., the sensitivities reported in Table 2.3).

Figure 2.10 shows the annual water use (sorted) by crop and indicates crops that are above versus below average water users.

According to the City of Phoenix, municipal water use was about 400,000 acre-feet in 2014. An acre-foot is a unit of volume equal to a sheet of water to cover one acre (=0.405 hectare) to the depth of one foot (=30.48 cm). It is equal to 1,233.5 m^3 (or 325,851

gallons). Thus, on an aggregate basis, the annual water use by UA crops is 7.2% of municipal water use in Phoenix. The state's annual water consumption is about 7 million acre-feet, 74% of which is used for irrigated agriculture (2017 figures from ADWR, undated-1). Our assessment of UA for Phoenix would add 0.6% to state water usage.

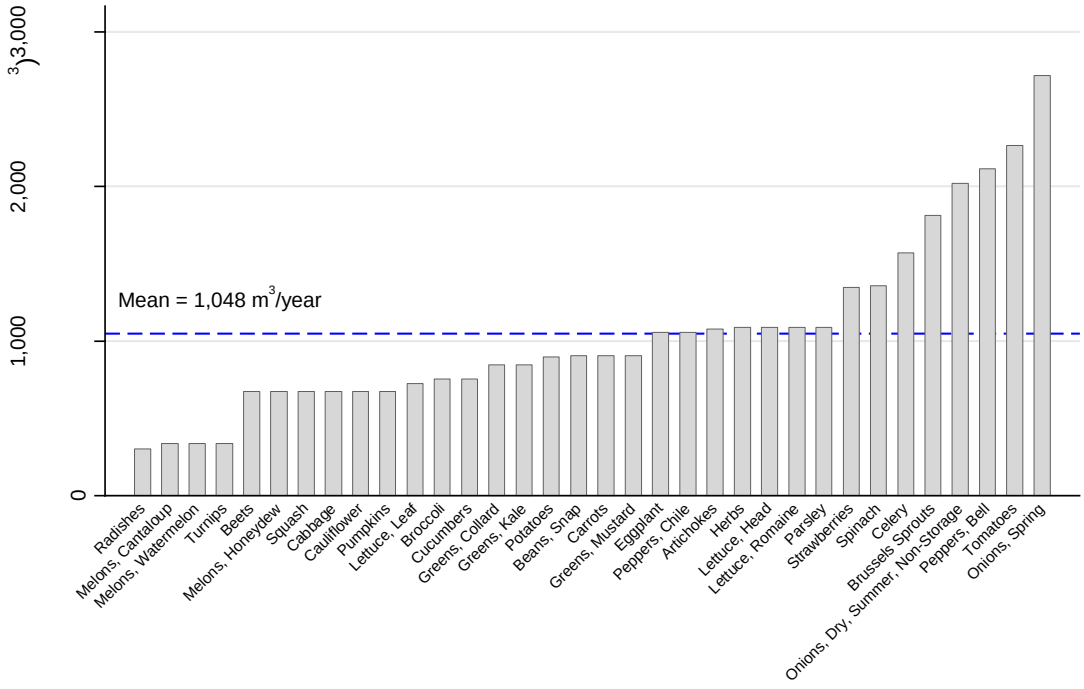


Figure 2.10. Annual water use by selected UA crops. Water use is in cubic meters and crops are sorted from lowest to highest by water use.

Indicative cost of water for UA applications in study area: We calculate indicative water cost to UA operators based on current monthly municipal water rates for City of Phoenix (2019). The rate structure has a fixed and a variable (volumetric) component. The fixed component varies by meter size; location (inside vs. outside the city); and season (low-

medium-high season; where high season is June-September). During the low/medium seasons, 6 units of water (1 water unit = 748 gallons), is included in the fixed charge. For locations inside the city, the lowest fixed charge is $5.50 and the highest is $24.60 per month. The volumetric component also varies seasonally and by location. For locations inside the city, the volumetric rates are $3.05 (low season); $3.56 (medium season) and $3.90 (high season) per month. Effective March 2019, there is also an environmental charge levied on all users. For users inside the city, this environmental charge is $0.56 per water unit.

For UA applications, we assume an indicative $10 per month fixed charge per site, and a volumetric charge of $3.31 per water unit per month over an 8-month growing season from October to May (outside of the June-September high water rate season). The $3.31 volumetric charge is the average of low and medium season rates. We add the environmental charge of $0.56 per water unit to the volumetric charge.

We treat each vacant lot, rooftop or façade UA application identified in our analysis as a unique metered site for water consumption. We then estimate total water use equal to 16.3 gallons/m^2 per month, based on crop water use; the area of each UA site; and area allocated to each crop per site. In our study area, the typical (average) vacant lot UA operation is estimated to consume 120.6 m^3 (31,859 gallons) per month; typical rooftop UA application 117.1 m^3 (30,922 gallons) per month and building façade applications about 1.87 m^3 (494 gallons) per month (based on unit areas in Table 2.2).

We then apply the assumed water rate structure for the UA sites. Note that even though most of the actual water use would take place over an 8-month growing season, users

would pay the monthly fixed charge for the entire year (12 months). The estimated annual average water cost is comparable for vacant lot and rooftop UA sites at about \$0.64/m^2 and nearly \$4/m^2 for building façades. Applications along building façades have a higher cost of water, because they are smaller in size (area) and need to pay the entire fixed charge even though monthly water use could be much smaller than the 6 water units included as part of the fixed charge.

Finally, we compare this \$/m^2 water cost to that of fertilizer, which is generally considered to be an expensive (and necessary) input for agriculture. We assume the application of a complete fertilizer mix suitable for vegetable gardens (containing twice as much phosphorus (P) than nitrogen (N) or potassium (K), e.g. 10-20-10 N-P-K) at a rate of 1,221 kg/ha per year (or 2.5 lbs per 100 ft^2 – an average rate for vegetable gardens). Using national fertilizer prices from the ERS (2015) over 2007-2014, we estimate the average cost of a complete fertilizer is about \$705 per ton. Multiplying this with the application rate (e.g. 705 \$/ton * (1,221 kg/ha / 10^7), we calculate fertilizer cost at 0.086 \$/m^2.

We focus on local impacts in this paper. However, both water (irrigation) and synthetic fertilizers could have potentially large off-site CO_2 footprints associated with their use. Use of reclaimed water instead of potable water where feasible, and minimal application of commercial fertilizers supplemented with compost are sustainable practices that can mitigate some of these lifecycle impacts.

Estimated water use by mesic landscapes and backyard pools in Phoenix: To provide context for the estimated water use by UA, we compare it to water use for maintaining existing mesic landscapes (e.g. turfgrass) and backyard pools in Phoenix.

We assume about 30% of residential structures in our study area have a pool (N=547,000*30%; based on LiDAR data) and the typical backyard pool to contain 45.4 m^3 (12,000 gallons) of water with a surface area of 20 m^2 (0.005 acres). Based on a water evaporation rate of 1.83 m (6 feet) per year (ADWR, undated-2), we estimate total annual water consumption of pools at 4,865 acre-feet in our study area. This is equivalent to 6,002,102 m^3 (1,585,588 thousand gallons).

We also assume a similar proportion (30%) of all properties—but this time including commercial structures as well (N=800,000*30%)—maintain year-round turf grass of about 80 m^2 (0.02 acres). Acceptable quality turf grass is assumed to use water at a rate of 4.2 feet per year in the Phoenix area (University of Arizona Cooperative Extension). Total water consumption of turfgrass in our study area is therefore estimated at 19,927 acre-feet, equivalent to 24,578,990 m^3(6,493,085 thousand gallons). The combined water use by residential backyard pools and turfgrass is 30,581,092 m^3 (8,078,674 thousand gallons). This amount corresponds to 86% of estimated UA water use of 35,630,953 m^3 (9,412,706 thousand gallons).

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2.7.7 Reduced building electricity use and CO_2 emissions savings from rooftop UA deployment

Calculation of avoided emissions. The avoided CO_2 emissions associated with building energy savings are calculated using the actual CO_2 emissions rate for power generation in Arizona (U.S. EIA 2018b). This rate is 901 lbs/MWh for 2016, equivalent to 0.409 tons/MWh. The city of Phoenix is part of the grid in Arizona, which is in turn part of the larger Western Interconnection. Because of this interconnectedness, it is not physically possible to determine the underlying supply mix for electricity consumed in Phoenix. Therefore, we assume it is the same as the supply mix for Arizona. We also assume all buildings for which we estimate energy and emissions savings are grid-connected.

Alternative analysis based on building age. The underlying spatial distribution of median year built data (U.S. Census Bureau, 2018b) by block group is shown in Fig. 2.11. In our study area, the block groups nearer the city's historic core tend to have the oldest buildings and the outlying block groups have the newer buildings.

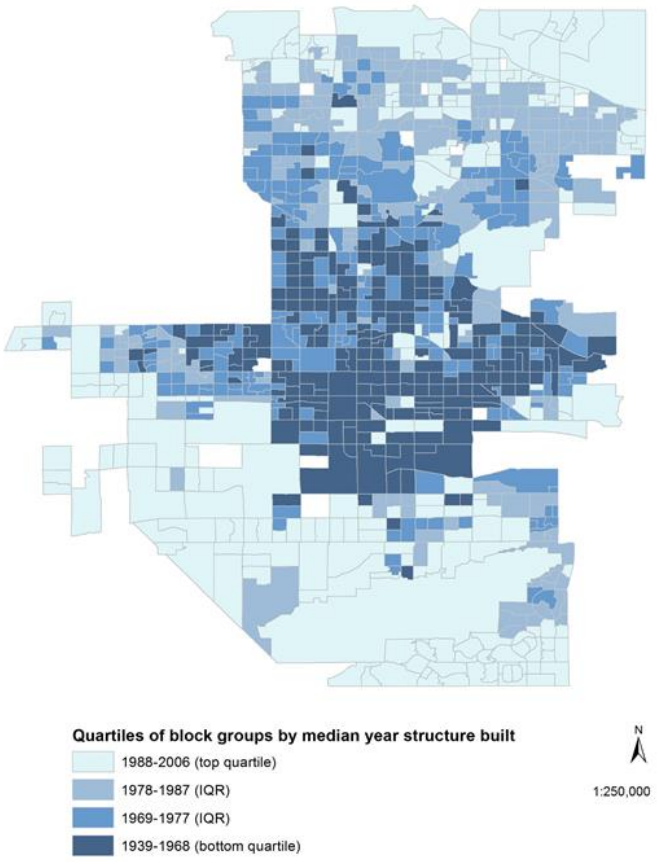


Figure 2.11. Spatial distribution of median year structure built. Median year built for residential structures within each block group. For display purposes, the block groups are grouped into quartiles.

Table 2.10 shows the breakdown of total avoided electricity use and associated CO_2 emissions savings shown in Fig. 2.6B by building age category. The avoided CO_2 emissions are calculated as above. We assign the average (1.84), low (0.25), and high (3.66) R-values to quartiles of block groups based on the median year structure built. The U.S. Department of Energy's commercial reference buildings data provides support for our assumption associating roof insulation with building age (U.S. DOE 2010).

According to this data, which represents approximately 70% of the commercial building stock in the U.S. (Deru *et al* 2011), roof R-values are lower for buildings constructed prior to 1980, and higher for those constructed from 1980 to 2004. The data indicates R-values for the latest vintage of buildings (constructed post-2004) are lower, possibly due to less strict building codes. Because the median year built for 98% of the block groups in our sample precedes 2004, we maintain our assumption that amount of roof insulation is inversely related with building age.

Table 2.10

Total avoided electricity use and CO_2 emissions when R_o-values vary with building age

Quartiles of block groups by median year structure built	Assigned thermal resistivity (R_o)	Avoided electricity use (MWh)	Avoided CO_2 emissions (metric tons)
Top (1988-2006)	High	11,015	4,502
IQR (1969-1987)	Average	50,568	20,666
Bottom (1939-1968)	Low	1,042,747	426,156
Total		1,104,329	451,324

Each row represents the subtotal of avoided electricity use (kWh) and corresponding CO_2 emissions (tons) over block groups that belong to the indicated quartiles (top, bottom, or the interquartile range – labeled IQR). The quartiles are constructed with respect to median building age for block groups. Quantities in the final row is the sum of these three rows.

Block groups in the top quartile have the newest buildings (median year built varying from 1988 to 2006) and are assigned the highest R_o-value. Block groups in the bottom quartile have the oldest buildings (median year built varying from 1939-1968) and are assigned the lowest R_o-value. The block groups in the interquartile range (IQR, combination of the 2nd and 3rd quartiles, with median year built varying from 1967-1987)

are assigned the average R_o-value. Twenty block groups had no reported data on median year structure built, and which were assigned the average R_o-value. We then compute total avoided electricity use and emissions across block groups.

The estimated total reduction in building electricity use of 1,104,329 MWh savings represent 3.7% of total commercial electricity use in Arizona, and 6.7% of the estimated electricity use in our study area. Total avoided emissions of 451,324 tons. account for 14.7% of commercial building GHG emissions in Phoenix (City of Phoenix 2017b). Block groups with the oldest buildings and largest UA rooftop area achieve the largest benefits.

The criteria used for determining suitable rooftops for UA (flat and >464.5 m^2 area) might preferentially select commercial buildings for UA provision. We do not have the same granularity of data on commercial building age in our study area. We know, however that commercial building stock in the region (specifically, the Mountain West census division of the U.S.) is of similar vintage newer (median year built = 1984; U.S. EIA 2016c) to the residential building stock (median year built = 1980-1989; U.S. EIA 2018b). Therefore, the savings shown in Fig. 2.6B are not likely to be sensitive to the underlying building type (residential or commercial).

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CHAPTER 3

CAN ENVIRONMENTALLY SUSTAINABLE LAND UTILIZATION FOR BIOENERGY CROPS SUPPORT A CELLULOSIC BIOFUELS INDUSTRY?

ABSTRACT

Sustainably produced cellulosic biofuels are key to meeting the low-carbon fuel demands of non-electrifiable transportation sectors, like aviation. The U.S. aviation fuel demand isforecast to exceed 30 billion gallons of gasoline equivalent by 2040. We show that it is economically viable to meet this demand with cellulosic biofuels obtained from domestically grown bioenergy crops (miscanthus or switchgrass), even when the cultivation of these crops is constrained to ensure land- and water resource sustainability. Using a coupled hydroclimate-ecosystem-economic modelling framework, we develop scenarios of cellulosic biofuel supply under different land-use and fuel price assumptions. This framework takes into account hydroclimatic impacts from landscape modification that inform land-use scenarios, while also incorporating hydroclimatic feedback on modeled yields. At a wholesale fuel price of $4/gge (gallons-of-gasoline-equivalent), up to 90.7 billion gallons of cellulosic biofuels can be supplied profitably from the conversion of 79.3 million ha of marginal lands in the Eastern U.S. into bioenergy crops. At the same price point, meeting the 30-billion gallon target with the smallest land footprint involves converting 27 million ha. This scenario displaces 10 million ha of existing corn/soy croplands, but in turn reduces stress on surface and ground water resources by preserving valuable soil moisture. This displacement would primarily

tradeoff the cropland (corn and soybean) devoted to supply of conventional biofuels in the U.S., with limited impact on crop production for food purposes. We also find that the hydroclimatic feedback on biomass yields is positive, and leads to higher biofuel output by an average 3.4% across different scenarios.

3.1 Introduction

Decarbonization of the global energy sector is a sustainability imperative. Latest scientific evidence shows rapid cuts are needed in greenhouse gas (GHG) emissions across the globe (IPCC, 2018). This stands in contrast with rising emissions: 2018 was the second year of rising global CO_2 emissions after a period of stagnation during 2014-2016. Today, more than 80% of global energy consumption still comes from fossil fuels (IEA, 2019b).

Aviation sector will represent a growing share of global CO_2 emissions as air travel grows at the expected rates (–at about 4.5% annually, doubling world's passenger miles by 2035.[1] The U.S., as the largest contributor (25%) of CO_2 emissions from aviation, consumed about 22 billion gallons of jet fuel (2018). This is expected to rise to about 27 billion gallons by 2040 (>30 billion gallons of gasoline equivalent) (EIA, 2019). Sustainable biofuels can support sectors of the economy, like aviation, that cannot readily switch from enery-dense liquid fuels (e.g. gasoline) to other forms of low-carbon energy sources (e.g. electricity from wind and solar).

[1] https://www.eesi.org/papers/view/fact-sheet-the-growth-in-greenhouse-gas-emissions-from-commercial-aviation.

The objective of this study is to illustrate how the demand for low carbon liquid fuels in the U.S. can be met with sustainably produced cellulosic biofuels ("sustainable biofuels"), while also ensuring the resulting biofuel industry is economically viable using a coupled hydroclimate-ecosystem-economic modeling approach. Sustainable biofuels (i) use biomass feedstock from perennial grasses (miscanthus or switchgrass) grown on marginal lands in regions where rain-fed agriculture is feasible; (ii) whose cultivation results in minimal or no depletion of soil moisture; while also (iii) generating regional cooling. We also introduce a methodological refinement via the use of modeled yields that reflect the hydroclimatic impacts induced from a large-scale conversion to bioenergy crops. The economic model determines the economically viable (profitable) biofuel industry configuration using this sustainable biomass setup (Parker 2011).

We do not separately consider GHG implications of biofuels derived from perennial grasses: evidence indicates these crops have the potential to completely offset gasoline emissions they displace (Borrion et al 2012, Monti et al 2012, and Wang et al 2011, 2012). They may even act as net carbon sinks when emissions from direct land use changes (LUC) are accounted for (Scown et al 2012, Dwivedi et al 2015, and Hudiburg et al 2016), unless native forests, grasslands, or savannas are converted in the process (Fargione et al 2008).

A large literature on the assessment of second generation biofuels has emerged over the past decade. Some studies empahsized on available land area and the biomass/biofuel output potential (e.g. Field et al 2008, Campbell et al 2008, Cai et al 2011, Zumkehr & Campbell 2013). Economic studies generally assessed the economic magnitude and

impacts of biofuel-related land use change or evaluated various policy scenarios (e.g., Parker et al 2010, Chen et al 2012, Beach et al 2012, Clark et al 2013, Murphy & Parker 2014). Another group of studies examined the role of ecological and biophysical drivers and natural resources impacts (e.g. Dominguez-Faus et al 2009, Robertson et al 2011, Davis et al 2012, VanLoocke et al 2010 and 2012, Gelfand et al 2013, and Georgescu et al 2011).

There is now a need for coupled or integrated approaches to jointly assess a multiple drivers and impacts. Fundamental to such assessments is the combination of environmental and economic sustainability criteria, since without a demonstration of economic feasibility, environmentally beneficial biofuels will remain as academic exercises (Aragon et al 2017).

Such an integrated (coupled) assessment by Hudiburg et al (2016) considers land allocation and emissions implications of a future with biofuel supply target comparable to this study. However, only a portion (16% to 42%) of their estimated biofuel supply is ultimately provided by perennial grasses. This is also the case for the comprehensive Billion Ton Study (U.S. DOE 2016) that includes perennial grasses only as part of a wider biofuel feedstock mix, including food crops. With an emphasis on radical decarbonization, our analysis excludes all conventional (crop-based) biofuel pathways, as well as those from crop residues (like corn stover) whose GHG emissions intensities are generally estimated to be lower than gasoline, but still positive.

The economic models of the agricultural sector have traditionally excluded true spatial variation in biomass supply (and therefore cost). Our approach incorporates a flexible,

fine-resolution (10km) spatial representation of the underlying land use, hydroclimate impacts, biomass yields, biomass supply and costs of biomass aggregation, transportation and bifouel distribution. This allows for more representative real-world constraints and outcomes. (For example, the analysis in Hudiburg et al, 2016, is conducted at the scale of crop reporting districts, which are groups of U.S. counties). Additionally, our analysis includes the actual siting and allocation of biomass to biorefineries. This results in lower, but more realistic assessment of the biofuel potential compared to studies that do not explicitly model the biorefineries and associated logistics, such as the Billion Ton Study (U.S. DOE 2016). In sum, we answer the following research questions:

1. Can we produce sufficient sustainable biofuels to meet the non-electrifiable portion of U.S. liquid fuels demand (from aviation)?
2. What is the required land transformation and what are the displaced land uses?
3. What is the cost gap that needs to be supported by (carbon) policy?

3.2 Coupled hydroclimate-ecosystem–economic modeling

The objective of our framework (Fig 3.1a) is to achieve a sustainable supply of biofuel that respects land, feedstock, water, and profitability constraints (sustainability constraints; left). To accomplish this, we employ a coupled modeling approach that integrates hydroclimate, ecosystem, and economic modeling.

Marginal lands assessment of Cai et al (2011) forms the basis of land use scenarios. Two marginal land use scenarios—the top 25% of suitable marginal lands (top 25% in short), and all marginal lands (all or 100% in short)—are considered to examine the widest

spectrum of outcomes due to large-scale perennial bioenergy crop deployment. These scenarios are used to simulate the near-surface temperature and soil moisture impacts of bioenergy crop-modified landscapes using a process-based hydroclimate model (WRF, Wang et al 2017).

Resulting gridded climate data with and without the bioenergy crop-modified landscapes become inputs to an ecosystem model (Agro-IBIS) to quantify the hydroclimate feedback on bioenergy crop yields. Agro-IBIS has been applied extensively for at multiple temporal and spatial scales in the U.S. Examples relevant to our analysis include comparisons with satellite estimates of vegetation phenology (Twine & Kucharik, 2008), crop yields (Kucharik, 2003), and miscanthus and switchgrass-specific evaluations (VanLoocke et al, 2010, 2012, and 2017). The ecosystem model simulates spatially-explicit 10-year crop yields (from establishment to mature stand) for two types of perennial bioenergy crops: switchgrass and miscanthus.

The switchgrass and miscanthus crop yields are combined with detailed, real-world economic and GIS-based inputs to develop cellulosic ethanol supply estimates using the Geospatial Bioenergy Systems Model ("GBSM", Parker, 2011) under various land use and feedstock yield scenarios (See Methods). GBSM is a flexible, spatially explicit model that considers the optimal behavior of the biofuel industry given fuel demand, biofuel price, and constraints on feedstock supply. Here, the model is tasked with solving for the optimal configuration for a profit-maximizing national cellulosic ethanol industry whose feedstock is sustainably sourced biomass from miscanthus and switchgrass. Iterating the model over range of reasonable wholesale fuel price levels (in $/gallons of gasoline

equivalent terms, hereafter gge) for each scenario, a set of comparable of cellulosic ethanol supply curves are constructed.

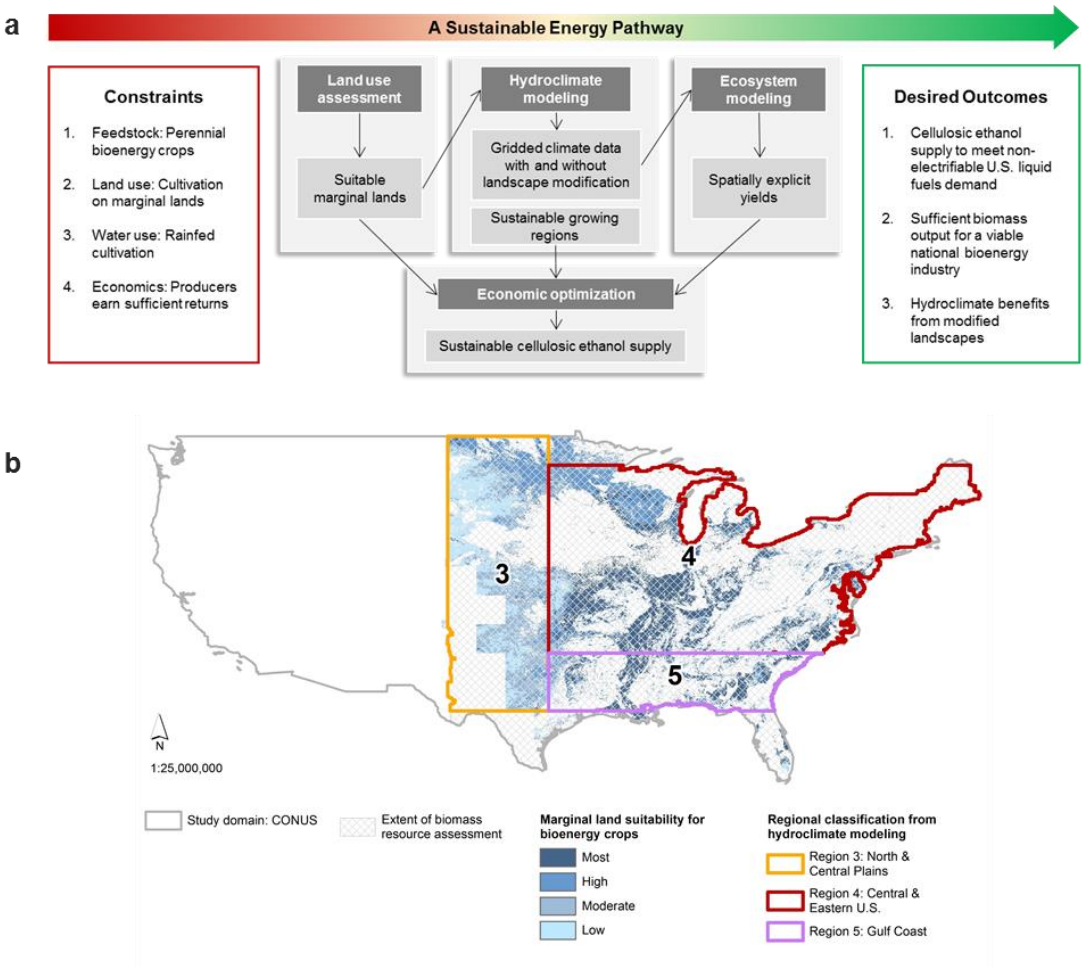


Figure 3.1. The modeling framework and study coverage. (a) Describes the coupled hydroclimate-ecosystem-economic modeling framework that considers the sustainability constraints (left) to accomplish the desired outcomes (right). (b) The study domain is CONUS, with the biomass resource assessment restricted to Eastern CONUS (hatched). Blue shaded areas indicate marginal lands suitable for cultivating perennial bioenergy crops; darker shadings indicate higher suitability. Yellow, red and magenta colored boundaries correspond to the regions 3, 4, and 5 respectively. These regions were identified in hydroclimate modeling and are used to further constrain the available land base.

The study domain (Fig 3.1b) includes all of CONUS, but the extent of the biomass resource assessment is restricted to the rainfed agricultural lands in Eastern CONUS. Its boundaries are defined here as the U.S. counties to the East of 104th Meridian West (hatched). This constraint also applies to the marginal lands that are suitable for bioenergy crop cultivation (blue shaded areas in Fig 1b); meaning we only consider marginal lands within these boundaries. Suitability of marginal lands for perennial bioenergy crops is represented in four categories, based on quartile reclassification of pixel values from Cai et al (2011), where "most" corresponds to the top quartile (highest 25% of pixel values), and "low", the bottom quartile (lowest 25% of pixel values).
The regional classification (3, 4, and 5) captures the three regions where different near-surface temperature and soil moisture change regimes were observed due to bioenergy crop modified landscapes in hydroclimate modeling (Wang et al 2016). We use these regions to test the impact of further constraining the land base available for bioenergy crops. Specifically, the North & Central Plains (region 3) exhibited the largest amount of near-surface cooling (average of 3° to 5°C) due to landscape modification, but also exhibited soil moisture loss (40-cm to 1-m depth) at the end of the 10-year cycle. The Central & Eastern U.S. (region 4) and Gulf Coast (region 5) are further East, and both regions demonstrated a similar near-surface cooling (average of 1° to 2°C). While neither region recorded a permanent soil moisture loss comparable to region 3, region 4 exhibited less variation in soil moisture during the 10-year cycle. In this paper, we treat region 4 as a sustainability hot spot, and consider the combined regions 4-5 as soil moisture retention zones.

3.3 Biofuel output and land allocation

Based on supply curves obtained under a representative set of scenarios, marginal lands in the U.S., even when they are limited to Eastern CONUS, can supply as much as 107.5 billion gallons of gasoline equivalent (gge) of cellulosic ethanol per year. This level of supply is delivered at the highest wholesale fuel price considered here ($5/gge) (see Fig. 2a, the right-most curve for scenario A_M). It involves the transformation of nearly all of the identified marginal lands in Eastern CONUS (234 million out of the 273 million acres total; see SI for underlying data) into managed miscanthus landscapes.

Moving from the least-constrained scenario to the other end of the spectrum, we consider a state of the world where the top 25% of marginal lands are dedicated to cultivating only switchgrass, which is uniformly lower-yielding than miscanthus (Fig. 3.2a, left-most curve, scenario T_S). In this case, the amount of cellulosic ethanol supplied at the same wholesale fuel price of $5/gge is 17.5 billion gallons per year (Fig. 3.2a).

There is a six-fold, or 90 billion gge difference in biofuel output between the least- and most-constrained scenarios (at $5/gge). This quantity could be thought of as the largest aggregate reward in output terms possible for using the maximum available marginal land base, choosing the higher-yielding crop (miscanthus), and ignoring potential soil moisture depletion as the least-constrained scenario (A_M) involves substantial land contribution from region 3.

Six additional scenarios occupy the space between these bookends. For all six, the economical bioenergy crop emerges as miscanthus. Three of these scenarios are not materially different from each other (SM_T_M, T_Mc, and T_M), with an output

difference of less than 5% at each point along the supply curve. The biomass supply shed in the SM_T_M scenario is limited to the "soil moisture retention zones" (i.e., amalgam of regions 4 and 5 from Fig. 1b) within the top 25% of marginal lands. This restriction ends up not being consequential because the top marginal lands mostly overlap with regions 4 and 5, thus SM_T_M and T_M are virtually indistinguishable. The T_Mc scenario uses the same underlying marginal land base T_M (top 25%), but different miscanthus yields. The difference is about simulation skill; while T_M uses yields obtained under the most-skilled simulation of contemporary climate, yields for T_Mc are from the least-skilled alternate (see Methods). The two supply curves closely track one another, leading us to conclude hydroclimate simulation skill is not a major driver of our estimates. We therefore keep T_M and exclude the two redundant scenarios from further analysis.

The scenario HS_T_M lies to the left of T_M. It is a constrained version of T_M to only use the top marginal lands within the hydroclimate hot spot (region 4). The scenario to the right of T_M relaxes the marginal land constraint from top 25% to include all marginal lands, while keeping the biomass supply shed inside region 4. Lastly, SM_A_M considers biomass supply in sourced from all of the available marginal lands that fall within the so-called soil moisture retention zones (regions 4 and 5 combined). It is a regionally restricted version of the least-constrained scenario (A_M) in Fig. 3.2a. Determining the high- and low-price bounds is important in understanding the shape and responsiveness of the supply and under the various constraints imposed. The biofuel supply curves (Fig. 3.2a) are capped at a wholesale fuel price of $5/gge, although we

tested higher price increments (up to $6/gge), which only added a negligible amount to the supply. None of the scenarios we considered can profitably produce a meaningful amount of biofuel from miscanthus below $3.4/gge, and production using switchgrass as feedstock is not feasible until prices reach $3.8/gge. Therefore, these two price points are price floors for sustainable biofuels given current technology and feedstock choices of miscanthus and switchgrass.

Substantial gains in biofuel output emerge as we consider incrementally higher wholesale fuel prices. Going from $3.4/gge to $4/gge (and from $3.8/gge to $4/gge in case of the sole switchgrass scenario), output grows x6.4 on average across the scenarios shown in Fig. 2a. Going from $4/gge to $5/gge, the pace of output growth slows down, with an average rate of output increase of 1.3 times. This suggests the production of sustainable biofuels has a price elasticity (of supply) greater than 1 along the entire supply curve up to $5/gge, with considerable flexibility to scale up when prices are between $3.4/gge and $4/gge. Outcomes between $4 - $5/gge produce the highest amounts of biofuel.

Beyond $5/gge, any increase in total biofuel supply is notional (and price elasticity nearly zero) across most of our scenarios, and the industry configuration does not differ materially.

In terms of meeting our desired demand target of 33 billion gge, only one scenario (the least-constrained, A_M) does so at $3.5/gge. At $4/gge, all but one scenario shown in Fig. 3.2a reach output levels near or above this target amount (94%-275%). The most constrained scenario using switchgrass as feedstock falls short (22%; this scenario could not meet the target even at the highest price we considered $5/gge). Therefore, we use

$4/gge for further evaluation as the lowest wholesale fuel price level that can deliver desired biofuel output levels.

In inflation-adjusted terms[2], wholesale fuel prices in the U.S. have never exceeded $4/gallon. Annual average jet fuel prices over the last forty years (1978-2018) surpassed $3.2/gge only briefly during 2008, primarily a consequence of crude oil prices exceeding $100 per barrel. Delivering sustainable biofuels at prices competitive with this record is not realistic in terms of existing technology and would not support an economically viable biofuel industry. More importantly, these prices are also not representative of a future economy that internalizes the cost of carbon. The cost gap between $3.2/gallon to $4.0/gallon could be traversed with carbon price of approximately $77.5 tCO_2-eq. Carbon prices around $145 tCO_2-eq would make biofuels competitive with crude-oil based liquid fuels today. This assumes sustainable biofuels produced are net carbon sinks; otherwise, they would be subject to the same carbon price, which would widen the gap. While these carbon prices are higher than the effective levels in place in several jurisdictions around the world (OECD 2018; with the exception of California), they are consistent with the lower bound of the required carbon price $135 tCO_2-eq (2010 dollars) to avoid 1.5°C warming based on the IPCC SR15 (Rogelj et al 2018).

At $4/gge, the (5) scenarios that produce sufficient biofuel to meet our demand target are shown in Fig. 3.2b, where the cellulosic ethanol supply ranges from 30.8 (HS_T_M) to 90.7 billion gge/year (A_M) (Fig. 2b, y-axis). The underlying land use ranges from 55.2 (HS_T_M) to 196.1 million acres (A_M) (Fig. 2b, x-axis).

[2] In 2012 dollars, comparable to our assessment.

Under T_M, the maximum biofuel output at $4/gge is 36.6 billion gge per year, using nearly 67 million acres of land. Moving left from T_M to HS_T_M, under which the biofuel supply would be more hydroclimatically sustainable, the output penalty amounts to 5.7 billion gge/year. Moving leftward from scenario A_M, the biomass supply under SM_A_M is 55.6 billion gge/year. This is 63%, lower than the least-constrained outcome. But the corresponding percentage reduction in underlying land use is higher (84%), indicating marginal lands in the more hydroclimatically sustainable regions (4 and 5) are also, on average, more productive.

In the middle of the pack is scenario HS_A_M under which annual biofuel supply is 44.7 billion gge, using 84.3 million acres of land. It can be interpreted in two ways. Approaching from the right, it represents limiting the biomass sources under A_M to only those within region 4. In that respect, the reduction in output of 46 billion gge/year is the cost of sourcing biomass only within the hydroclimatic sustainability hot-spot. Approaching from the left, it represents a relaxing of the marginal land constraint (from top 25% to include all marginal lands), while keeping the biomass supply shed inside region 4. The result is output that is 8.1 billion gge/year higher than T_M. Allowing the model to utilize a larger marginal land base pays off, even if this land base is on average less productive than the top 25% of marginal lands.

Total biomass supply (in dry tons) increases with the total amount of land use by scenario, ranging from 546.6 million to 1,607 million dry tons (Fig 3.2b, 3.2c). The amount of biomass supplied from existing croplands also increases proportionally to underlying land use. However, the proportion of croplands displaced is higher under

scenarios with lower biofuel output and smaller land base. For example, under HS_T_M, where the land use is limited to the top 25% of marginal lands use and further restricted to the hydroclimatic hot spot (region 4), the biomass supply from existing croplands is 52% of total. Biomass supply from corn and soy make up the majority (43%). Under the scenario A_M, which uses the largest possible marginal land base (100%), most of the biomass supply is sourced from non-croplands (66%), followed by corn/soy (20%) and other crops (15%). The economic model, by virtue of its profit-maximizing objective, sources more biomass from more productive lands (where yields are higher). It is thus not surprising that under stricter land use constraints, the relative contribution of biomass from such high-yielding croplands is higher. This is supported by the average effective miscanthus yield that decreases monotonically with land use (blue labels in Fig 2c, going from right to left across the scenarios).

Some of the marginal lands that are suitable for bioenergy cropping (from Cai et al, 2011) are also actively used as croplands. This is because in the U.S., technology can improve productivity of marginal croplands, and crop prices and policies can encourage cropping to opportunistically encroach on otherwise less productive (marginal) lands (Wright & Wimberley 2013, Roberts & Schlenker 2013, Aragon 2019).

This points to a tradeoff in land use for bioenergy cropping. Displacing too much cropland for bioenergy crops may adversely impact food production. Restricting bioenergy cropping only to the marginal lands that are currently not cropped (e.g., most of region 3 in this study), however, could have detrimental impacts on the regional hydroclimate. Additionally, average bioenergy crop yields would be lower over some of

these marginal lands, which would require more land to be used for producing a unit of output, all else equal.

Based on these results, we can further narrow the field down to two "realistic" scenarios: T_M and HS_A_M (Fig. 3.2b, scenarios highlighted in grey box). Both scenarios produce sufficient biofuel to meet our demand target of 33 billion gge/year. They represent choosing biomass supply points from different underlying marginal land bases; the top 25% in case of T_M and all marginal lands in case of the HS_A_M scenario. Both scenarios also source most of their biomass within hydroclimatically sustainable regions 4 and 5. (In fact, HS_A_M is restricted to sourcing within region 4.)

Under the T_M scenario, which selects from the most suitable of marginal lands for bioenergy crop cultivation, total land conversion at $4/gge is 27 million hectares (Fig. 3.3a), which generates an average of 684.4 million tons of biomass annually (Fig. 3.3c). About 10 million hectares of this land comes from existing corn and soy croplands (Fig. 3.3b). The HS_A_M scenario selects from all suitable marginal lands within the sustainability hot-spot (region 4). The corresponding biomass supply at $4/gge is 792.3 million tons, obtained from 34.1 million hectares of land converted into miscanthus (Fig. 3.3d, 3.3f). The amount of corn and soy croplands converted is higher than T_M by 1.1 million ha, but most of the incremental land use comes from non-croplands (6 million ha). The marginal productivity of these lands is markedly lower: the additional 7.1 million acres produces 144 million tons of biomass (20% more over T_M), at an average yield of 20.3 tons/ha (as opposed to 24 tons/ha under T_M).

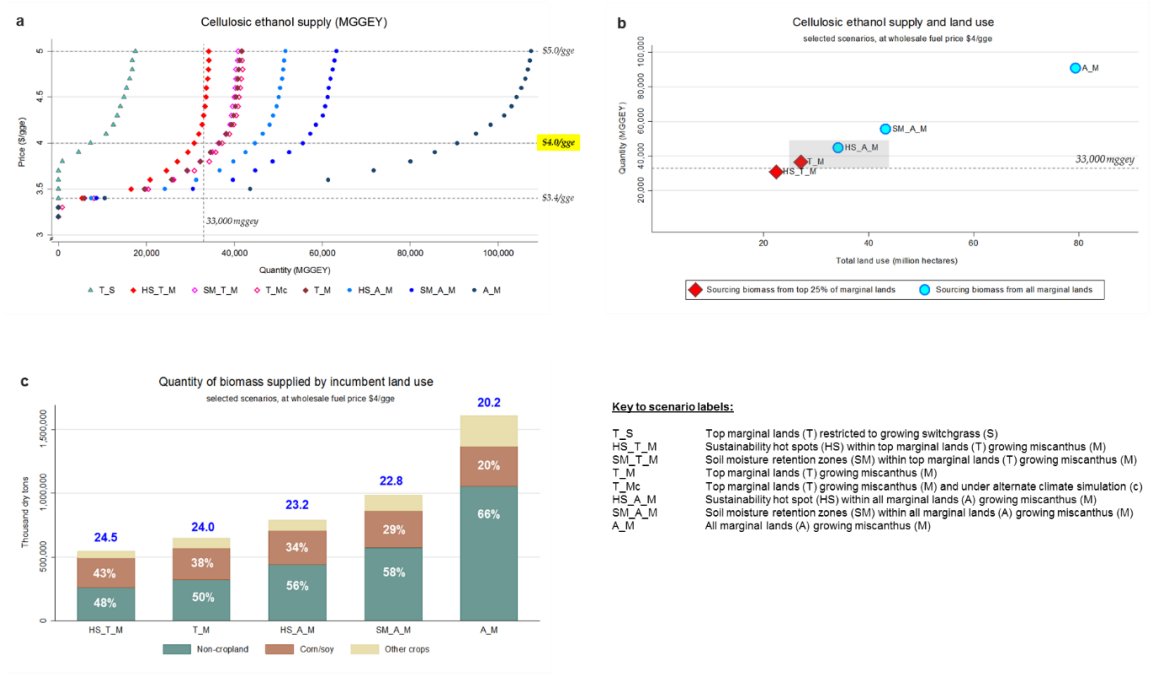


Figure 3.2. Main results from economic modeling. (a) Cellulosic ethanol supply curves by scenario. Shows the quantity of biofuel deliverable (million gallons of gasoline equivalent per year, mggey; x-axis) at the wholesale fuel price in dollars per gallons of gasoline equivalent ($/gge; y-axis). The 33 billion gge amount needed to meet the future non-electrifiable liquid fuels demand is indicated by dashed vertical line. The reference lines at price levels $3.4, $4.0 and $5.0/gge are drawn to highlight their intersection with the supply curves. (b) Total land use (million hectares) corresponding to cellulosic ethanol supply (mggey) from selected scenarios, at wholesale fuel price of $4/gge. This time, the demand target of 33,000 mggey is a dashed horizontal line. (c) Breakdown of biomass supply by underlying incumbent land use type (non-cropland, corn or soy cropland, all other cropland) for the same set of scenarios as in (b), also at wholesale fuel price of $4/gge. The blue labels above the bars indicate effective biomass yield (for miscanthus) in dry tons/ha, by scenario. The labels inside the bars in white text correspond to the proportion of biomass supplied from non-croplands, and from corn and soy croplands. The proportion of biomass supplied from other croplands is not shown, but equals to 100% minus of the sum of two proportions (eg. for A_M, it is 100% - (66%+20%) = 14%).

The biomass-sheds for the two scenarios overlap by a large extent, particularly along the upper sections of the Mississippi River, where we observe the largest land conversions into bioenergy crops. The supply points common to both scenarios provide 83% of all the biomass produced under T_M, and 63% under HS_A_M. Highly productive supply clusters emerge in Missouri, southern Illinois, northeastern Arkansas and western parts of Kentucky and Tennessee.

The economic model favors sourcing biomass from these supply points (grid cells) because of higher expected yields for miscanthus (>25 tons/ha). The very areas that generate higher expected yields for miscanthus, however, are more likely to be already cropped, indicated by the high proportion of croplands within the converted grid cells (Fig. 3.3b, 3.3e). This highlights a feature of marginal lands in the U.S.: those marginal lands that are most suitable for bioenergy cropping tend to be already in cultivation. For the biofuel industry to profitably scale up, displacing some of the existing croplands is required.

For both scenarios, the total land conversion is under 40 million ha (this does not to exceed the largest amount of land use by a single crop in the U.S., which peaked at 38.3 million ha for corn in 2012). Furthermore, more than 50% of the biomass is contributed from non-croplands in these two scenarios. The corn and soy cropland displacement (10 million ha under T_M and 11.1 million ha under HS_A_M), is lower than the acreage dedicated to the production of conventional biofuels--corn ethanol and soy biodiesel—implied by output shares over the recent years.

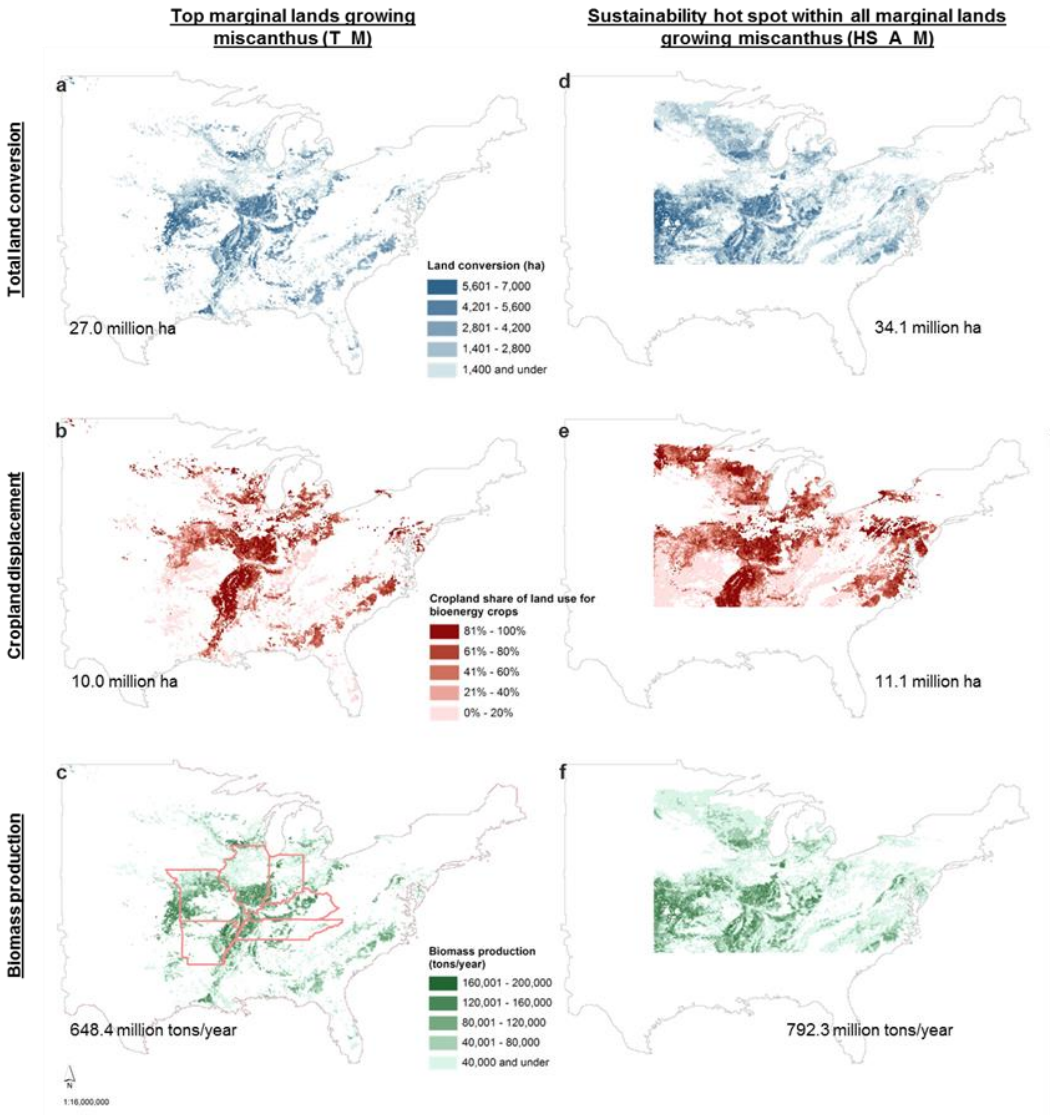


Figure 3.3. Land allocation and biomass production in space by scenario. The two scenarios depicted here are T_M (panels a, b, and c) and HS_A_M scenarios (d, e, and f). Quantities are shown at the modeled grid resolution (10 km, or a maximum area of 10,000 ha per pixel). (a, d) Land conversion into miscanthus in hectares per pixel. Inset labels indicate total land conversion in million hectares. (b, e) Of the amount of land converted, the proportion that displaces existing cropland. Dark shaded areas indicate a higher rate of cropland displacement, and light shaded areas indicate a higher reliance on non-croplands. Inset labels show the amount of corn/soy cropland converted in million hectares. (c, f) Expected biomass quantity (dry matter) from miscanthus in tons per year, per pixel. Inset labels indicate total biomass production in million tons/year. Panel (c) shows the boundaries for the top 5 states in terms of biomass quantity supplied in both scenarios.

However, if we account for the co-products under each biofuel pathway (e.g. distiller's grains from corn ethanol; soymeal and glycerin from soy biodiesel; lignin from bioenergy crop-based ethanol) (Wang et al 2011), the net land devoted to the biofuel portion of each pathway would be lower. The actual net land use would depend on underlying yields and relative conversion efficiencies. For example, Mumm et al (2014) estimated that, in 2011, the share of total corn acreage used for ethanol production at 27%, even though 40.5% of corn output was used for that purpose. Therefore, cropland displacement in favor of miscanthus could impact food production.

3.4 Investment and the size of industry

Total annual costs of the resulting biofuel industry range from 114.3 (HS_T_M), consisting of 112 new biofuel refineries, to 337.5 billion dollars (A_M) for 1,246 new biofuel refineries. Total costs include feedstock procurement, feedstock transport, fuel distribution, operating and maintenance expenditures and an annualized capital cost for these plants. Conversion costs, defined as the sum of annualized capital costs and annual operating & maintenance expenditures, are the largest contributor to total costs (57% average by scenario). Feedstock procurement costs are the second largest cost category (37%, average by scenario) (Fig. 3.4). Total costs rise with the amount of biomass and biofuel output; the rate of increase (about 22%) is roughly proportional to the increase in underlying area going from scenario HS_T_M to SM_A_M. While total costs under the final–least-constrained–scenario (A_M) increase by a much larger margin (63%) over the previous one, average costs of production for all scenarios are very similar (all below

$4/gge). We find average costs per year by dividing total costs by the biofuel output (mgey) in $/gge terms (Fig.3.4, right axis). This is the market price we assume for the final biofuel product (before taxes and distribution to fuel terminals). This indicates the industry, on the whole, is profitable. Though average costs are very similar in magnitude, the scenarios with the smallest and largest land base have the highest unit costs, and the 2nd and 3rd scenarios from left (highlighted in Fig. 3.3) exhibit the lowest unit costs.

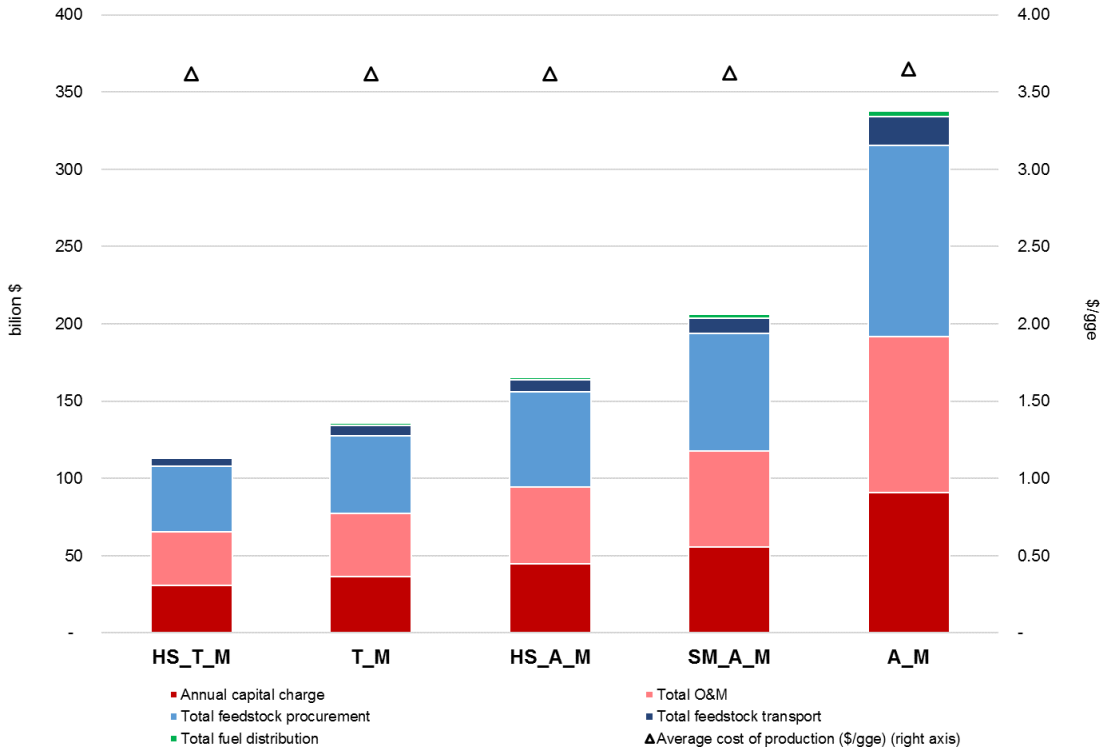


Figure 3.4. Breakdown of industry costs by scenario. The total costs of the biofuel industry per year shown as the sum of the annualized one-time investment cost (capital charge), total operating and maintenance (O&M) costs, costs of procuring the biomass feedstock and transporting it to the biorefineries, and finally the cost of distributing the biofuel output CONUS-wide (in billion 2012 dollars; left axis). Average cost of production is the ratio of total costs ($) to total biofuel output (mgey) (in 2012 $/gge terms; right axis).

Nearly half of the conversion costs in our analysis come from annualized upfront capital costs. In comparison, feedstock costs (crude oil) constitute the majority (nearly 60% over 2009-2018) of the costs for petroleum products in the U.S. Refining costs, which would be comparable to the conversion costs for biofuels, generally make up less than one-fifth of total costs for petroleum products (based on data from EIA 2019 a,b,c). The lower share of refining in total costs for petrolueum products (as opposed to conversion costs in total biofuel costs) is because the oil refining industry has for the most part recovered its capital costs. The cellulosic biofuel industry, on the other hand, is still in its infancy, with a relatively new and unproven technology.

3.5 Integrated modeling and hydroclimate feedback on yields

We compare the average bioenergy crop yields (in mg/ha) measured at and grouped by different components or stages of our integrated model to illustrate the output impact of our approach. Because we do not alter any other model parameter or assumption for each of the scenarios besides the underlying land base and simulated yields associated with that land base, the supply estimates closely reflect underlying variation in yields. The relationship (between average output and average yields) is especially strong at price points $4 and above when the model utilizes nearly all supply points for each scenario (Fig. 3.5).

The largest single impact on yields and supply is associated with the bioenergy crop selection. In the economic model, provided a choice between miscanthus and switchgrass, miscanthus always emerges as the preferred crop. Even though the costs of

growing switchgrass are lower, the yield difference is large enough to tip the scale in favor of miscanthus. For each grid cell (supply point) in our analysis, the simulated yield for growing miscanthus in that grid cell is never lower than that for switchgrass. On average, and all else equal, restricting feedstock choice to switchgrass would result in an immediate halving of biomass quantity (Fig. 3.5a). The loss rate for total *biofuel* output could be even higher as the economic model tries to optimize supply over fewer profitable supply points for switchgrass than miscanthus at a given price level (e.g., supply curves T_S vs. T_M in Fig. 3.2A).

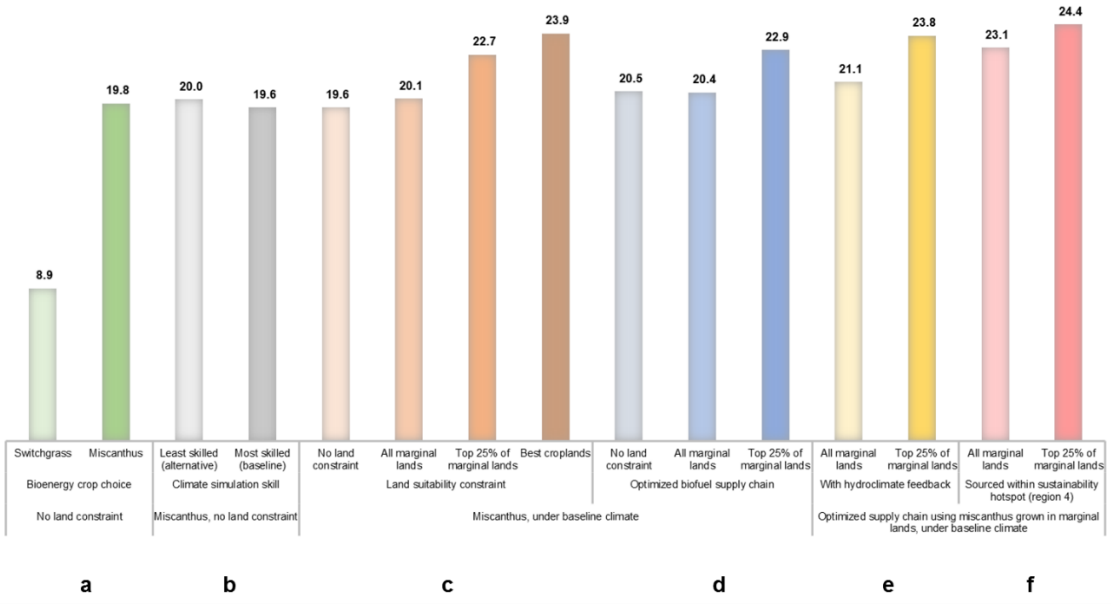


Figure 3.5. Influence of integrated modeling on output. Demonstrated via variation in bioenergy crop yields. Going from left to right, each bar considers average bioenergy crop yields (in mg/ha) measured at and grouped by different components or stages of our integrated model. For example, the left-most set of columns show the variation in biomass yields by bioenergy crop choice, without any restrictions. The right-most two columns indicate effective (optimized) miscanthus yields reflecting the hydroclimate feedback affect derived under the baseline climate conditions, plus land use constrained to those marginal lands that are hotspots for hydroclimate benefits.

Hence, the following discussion is predicated on miscanthus being the crop of choice as biofuel feedstock. The gridded data from the least skilled hydroclimate simulation produced higher average miscanthus yields in ecosystem modeling (Fig. 3.5b, 20 mg/ha vs. 19.6 mg/ha; also see supply curves in Fig. 3.2a, T_M vs. T_Mc), but the difference is not significant. Thus, we rely on modeled yields using the output from the most-skilled hydroclimate simulation, because it is the best representation of contemporary climate.

In our analysis, constraining the land base has a favorable influence on miscanthus yields (Fig. 3.5c). Our definition of marginal lands (per Cai et al 2011) include marginal croplands and exclude lands that are used as pasture/range, while the unconstrained land base includes all land that is not urban, forest, or water/wetland. Since most pasture/range land is of lower quality and productivity, the marginal land constraint as applied in our analysis effectively removes some lower quality land, resulting in higher average miscanthus yields (20.1 mg/ha vs. 19.6 mg/ha, or by 3%). It follows that the top 25% of suitable marginal lands, which are primarily marginal croplands, have an even higher average miscanthus yield (22.7 mg/ha; 13% higher than the yield for miscanthus over all marginal lands). Yet, the marginal lands used in our analysis do not overlap with the most productive of croplands in Eastern CONUS (e.g., heart of the corn/soy belt in Indiana-Illinois-Iowa). These areas, if converted, would have generated the highest possible miscanthus yields (average of 23.9 mg/ha; the "best croplands" column in Fig. 3.5c).

As indicated earlier, the economic optimization model selects supply points that are profitable, which tend to have higher biomass yields. Comparing the average miscanthus yields in Fig. 5b and 5c illustrates the benefits of economic optimization (as opposed to

simply aggregating biomass output within a given area). These benefits are the largest (5%) when underlying land base is most diverse (unconstrained). When miscanthus is sourced from all marginal lands or the top 25% of suitable marginal lands, the yields of selected supply points, are on average, 2% higher.

Next, we consider the role of hydroclimate feedback. We measure this impact comparing the yields simulated using climate forcing generated with and without the bioenergy-crop landscape modification over marginal lands. When the landscape modification involved all marginal lands, the average increase in simulated yields is about 3% (Fig. 5e vs 5d), and it is about 4% when the top 25% of marginal lands were converted. The improvement in yields is a function of hydroclimatic impacts associated with summertime cooling increase the time during the growing season when the temperature is in the optimal range for growing miscanthus.

There is a latitudinal pattern to the hydroclimate impact on yields. The average impact is larger--5.3% when the land base includes all marginal lands, and 4.4% when the land base is restricted to the top 25% of marginal lands--for supply points located in mid-latitudes (30°N - 40°N). Moreover, when the underlying land base includes all marginal lands, about 34% of the supply points selected fall north of 40°N latitude: for this subset, we find negative (-1.7%) hydroclimate impact on yields. In far northern latitudes of our study area, the landscape modification-induced summertime cooling impedes the growth of miscanthus, which, though cold-tolerant, is esentially a warm-season C4 grass.

Lastly, our analysis also imposes regional boundaries (Fig. 3.1) to the biomass supply shed. These boundaries differentiate regions where reductions in near-surface

temperatures during summertime indicated no sustained loss of soil moisture (the soil moisture retention zones of regions 4 and 5) versus a depletion of soil moisture (region 3) based on hydroclimate simulations. When we restrict the marginal land base to source biomass from supply points only from the hydroclimatic sustainability hotspot (region 4), which exhibits the most stable soil moisture profile over a 10-year simulation period, average yields are the highest (Fig. 3.5f). When the biomass supply shed is restricted to all marginal land pixels within region 4, the corresponding yield increase is 10% (over levels in Fig. 3.5e). Note that this is incremental yield improvement given existing hydroclimate feedback. For the top 25% of marginal lands, the comparable incremental yield impact is less pronounced at 3% (also over levels in Fig. 3.5e). Nonetheless, the average yields at 24.4 mg/ha exceed the modeled miscanthus yields in the top crop producing regions in our study area of 23.9 mg/ha (the "best croplands" column in Fig. 5c; measured without a hydroclimatic feedback.) Thus, restricting the biomass supply shed to the sustainability hotspot of region 4 not only is beneficial from a soil moisture preservation standpoint, but also has the added advantage of sourcing biomass from the more productive subset of marginal lands (i.e., higher baseline yields).

Overall, through are integrated modeling, we can concentrate sustainable biofuel production in areas with average effective yields >24 mg/ha. The improvement in yields and output due to beneficial hydroclimate feedback averages 3.4% across scenarios, and as high as 6%.

3.6 Limitations and uncertainty

In our analysis, the hydroclimate simulations are conducted using WRF coupled to a land-surface model (Noah LSM); this coupling allows us to simulate the hydroclimate impacts with and without landscape modification. However, there is a potential mismatch in the representation of land use and land cover (LULC) that is used in Noah LSM and Cai et al (2011)'s analysis of marginal lands suitable for bioenergy crop deployment. Thus, alternative representations of marginal lands in the U.S. that are more consistent with the Noah LSM LULC could produce different impacts due to landscape modification. The hydroclimate impacts can be further refined using more complex land surface models, such as Leaf-2, that include representations of groundwater and river dynamics.

The range of modeled bioenergy crop yields (mg/ha) are within observed ranges (Miguez et al 2012, Li et al 2018) for the United States and consistent with estimates obtained from other ecosystem models (e.g. Hudiburg et al 2016). Nonetheless, actual yields could be more variable in far southern and northern latitudes than simulated by the ecosystem model. In our analysis, a larger proportion of biomass supply points underlying the supply curves shown in Fig. 2a are located between 30°N and 40°N (shares ranging from 64% to 91% by scenario), which mitigates the impact of yield uncertainty associated with this limitation. There are also fewer measurements yields conducted in field experiments in latitudes south of 30°N and north of 40°N to validate the models for these regions.

The economic optimization model incorporates a cropland supply - rent (price) elasticity estimate that allows the cost of acquiring the biomass feedstock to increase linearly with

the demand for feedstock (see Methods). However, the model does not incorporate all possible feedbacks from agricultural and energy markets as a consequence of large-scale land conversion into bioenergy crops.

Additionally, our supply estimates are most consistent with the hydroclimate impacts driving the simulated yields under near-full land utilization with respect to marginal land deployment scenarios (top 25% and 100%, or all). For instance, the outcomes from the hydroclimate simulations based on landscape modification of the top 25% marginal land use scenario represent modification of all possible pixels identified in that scenario. The economic model, on the other hand, would preferentially select an optimal subset of these pixels at each price point. The model approaches full utilization of the underlying land base as fuel price increases, permitting more expensive supply points (pixels) to be selected (become economical). In our analysis, the consistency with the underlying modified landscape is achieved along the portions of the supply curve where wholesale fuel price ≥$4/gge.

Some of the larger sources of uncertainty surrounding our estimates, and the development of sustainable biofuels generally, involve markets and prices. The supply of sustainable biofuels will be highly sensitive to demand for oil products (aviation fuel) and oil prices. High oil prices would encourage switching to cleaner liquid fuels, while low oil prices would depress the demand for such fuels. Carbon prices (taxes) would also play a role. Even though any level of carbon price would make crude oil-based fuels more expensive in the short run, too high a carbon price could eventually drive down crude oil prices. The impacts would also depend on to what extent biofuel prices would be affected by the

carbon price scheme (e.g., not designed to capture land use impacts of fuels, or supplies are drawn from lands that generate a carbon debt, or indirect land use effects emerge). Similarly, a large quantity of biofuels reaching the market would lower the demand and prices for crude oil products. Lower prices then cause demand to "rebound", an effect whose magnitude may be larger beyond U.S. borders (Rajagopal, 2013).

Because cellulosic biofuel technology is yet commercially unproven, uncertainty surrounding conversion yields and costs of conversion remains. There are also additional costs (that vary by biomass feedstock and conversion pathway) to make these biofuels suitable for use as jet fuel, which is subject to more stringent quality standards than gasoline. To date, several biojet pathways have been approved for blending biofuels with jet fuel (up to 50%) in military and commercial applications, and others are in progress (Wang et al 2016, Wang and Tao, 2016). Recent findings also indicate biofuel blending reduces particle emissions from aircrafts during flight (Moore et al 2017).

3.7 Conclusions

Using a coupled hydroclimate-ecosystem-economic modelling framework that considers hydroclimatic impacts from landscape modification by bioenergy crops, we develop cellulosic biofuel supply estimates under various marginal land use and price scenarios. A wholesale fuel price of $4/gge emerges from our analysis as a lower-bound at which several different land use scenarios are able to produce sufficient cellulosic biofuel to meet the target demand of 30 billion gallons. At this price point, the highest output of cellulosic ethanol is over 90 billion gallons and involves the conversion of 79.3 million

ha of marginal lands. Meeting the 30-billion gallon target at $4/gge while using the least amoung of land involves converting 27 million ha. The analysis in this paper shows it is not possible to entirely avoid displacing marginal croplands if reducing stress on surface and ground water resources by preserving valuable soil moisture is a priority. Still any cropland displaced would mainly offset those devoted to the supply of conventional biofuels, with limited impact on food supply. In addition, the value of our integrated modeling framework is demonstrated by the positive hydroclimatic feedback on modeled yields. On average, this leads to higher biofuel output by 3.4% under different scenarios. The bioenergy crop-modified landscapes in our analysis have the potential to be GHG-neutral. Landscape management approaches can manage any negative environmental consequences associated with large-scale deployment of these crops, particularly miscanthus, which has higher yields and better economics than switchgrass. The obstacles to development of a viable cellulosic biofuel industry are primarily economic: This analysis shows that at $4/gge and with a carbon price above $77 tCO_2-eq, the industry can compete with petroleum-based liquid fuels. As industry closes in on making biojet (biofuels for aviation) commercially feasible, it is important to continue to iterate on what it means to transform our lands to supply low-carbon liquid fuel alternatives in a sustainable fashion.

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3.10 Supplemental information: methods

Marginal land base suitable for bioenergy crop cultivation. Is based on the global assessment conducted by Cai et al (2011). We use the most realistic scenario (4) that estimates a total of 123 million hectares available for conversion to perennial bioenergy crops in the Contintental US (CONUS). This land base includes marginal mixed crop and vegetation land, grassland, and scrubland with marginal productivity, but excludes current pastureland. For the purposes of our analysis, we reclassify the raw pixel values in Cai et al (2011) that represent land suitability for second generation bioenergy crops into quartiles. Values ranging from [0, 25%), [25%, 50%), [50%, 75%) and [75%, 100%) are labeled as low, moderate, high, and most suitable, respectively.

Hydroclimate modeling. The Weather Research and Forecasting Model (WRF) (Skamarock et al 2008) coupled to the Noah land surface model (Noah-LSM) (Chen and Dudhia 2001; Ek et al 2003) was used to simulate the hydrclimatic response to landscape modification. The landscape modification involved a perennial bioenergy crop conversion of the marginal lands described above for the entire CONUS. Two marginal land conversion scenarios were considered: areas that are in the top quartile of suitability (top 25%) and all suitable locations (100%) for bioenergy crop cultivation.

Perennial bioenergy crop conversion was represented in Noah-LSM by altering the albedo, LAI (leaf area index) and vegetation fraction values in the contemporary land cover (1-km modified IGBP MODIS 20-category land use/land cover dataset). The representative values for perennial bioenergy crops were based on reported ranges (e.g. Dohleman and Long 2009).

Among an ensemble of eight sets of control simulations (E1–E8), the E8 model configuration best reproduced near surface climatic conditions, and the E1 model performed the worst. In this paper, we refer to the E8 as the baseline or "most skilled," and E1 as the alternative or "least skilled" hydroclimate simulation. These two simulations are then each performed with and without the attendant landscape modification scenarios, resulting in a total of six scenarios.

The simulations produce several variables of interest: near-surface (2m) temperature (variable T2, in Kelvin), downwelling shortwave radiation at surface (variable SWDOWN, in W/m^2), surface pressure (variable PSFC, in Pa), near-surface wind speed (variables U and V, in m/s), 2m specific humidity (or water vapor mixing ratio, variable Q2, in kg/kg or 1), and precipitation (variable PRECIPHR, in mm). Each variable is obtained as raw output of WRF in three-hour frequency from 2000 to 2009, except precipitation. The 3-hourly precipitation amount is derived from accumulated precipitation. Further methodological details and results from the hydroclimate simulations are presented in Wang et al (2017).

Ecosystem modeling. Gridded climate data from the hydroclimate modeling described above are then used as inputs in the Integrated Biosphere Simulator – agricultural version (Agro-IBIS) to estimate dry matter yields (mg/ha) for two perennial bioenergy crops, miscanthus (Miscanthus x Giganteus) and switchgrass. Agro-IBIS is a dynamic global vegetation model (DGVM) that simulates the growth of natural vegetation as well as major U.S. crops (such as maize, soybean, and spring and winter wheat).

Agro-IBIS was used to simulate a perennial biofuel feedstock system representing either miscanthus or switchgrass, previously demonstrated in VanLoocke et al (2010). Ten-year simulations that capture the lifecycles of these perennial grasses were conducted over the entire CONUS at 10-km grid size and at the individual crop level. This granularity captures the variation in soil characteristics in addition to climate forcing. Control simulations used the two climate forcings generated by the "most skilled" and "least skilled" simulations without any landscape modification from the previous modeling step. Then, yields were simulated using climate forcings generated following the two scenarios of landscape modification: 25% and 100% of marginal lands. In all, 12 different scenarios of dry matter yields were simulated (see Tables 3.1 and 3.3). Summary statistics of the 10-year average simulated yields by 10km grid cell (each with at least 50 acres of available land) are provided in Table 3.1 below organized by crop, hydroclimate simulation skill and marginal land deployment (for landscape modification). (Note that these statistics represent unconditional yields: effective yields by scenario are shown in Figure 3.2c, which are averages by supply point selected in each scenario at $4/gge reference biofuel price level). Figure 3.6 and Figure 3.7 graph average simulated yields for miscanthus and switchgrass, respectively, over grid cells by year of simulated stand life (1-10), organized similarly to Table 3.1.

Table 3.1

Summary statistics for 10-year average simulated yields (mg/ha) by crop, hydroclimate simulation skill and marginal land deployment

Crop	Hydroclimate simulation	Landscape modification	N (no. of grid cells)	Mean	SD	P5	P95	Min	Max
Miscanthus	Most-skilled Baseline (E8)	Top 25% of marginal lands	68749	19.9	4.8	12.1	26.8	0.0	30.0
		100% of marginal lands	68749	20.0	4.2	13.2	26.5	0.0	30.4
		None (control)	68749	19.5	4.7	12.0	26.7	0.0	29.9
	Least-skilled alternate (E1)	Top 25% of marginal lands	68749	21.0	6.4	10.2	31.1	0.0	36.0
		100% of marginal lands	68749	22.3	5.7	13.1	31.7	0.0	37.3
		None (control)	68749	20.0	6.3	9.7	30.7	0.0	36.8
Switchgrass	Most-skilled Baseline (E8)	Top 25% of marginal lands	68749	9.6	3.5	3.6	14.4	0.0	17.7
		100% of marginal lands	68749	10.1	3.0	4.4	14.4	0.0	17.7
		None (control)	68749	9.0	3.3	3.6	13.6	0.0	16.5
	Least-skilled alternate (E1)	Top 25% of marginal lands	68749	9.7	4.4	2.5	16.2	0.0	19.7
		100% of marginal lands	68749	10.9	4.1	4.0	16.9	0.0	20.7
		None (control)	68749	8.8	4.3	2.4	16.0	0.0	19.5

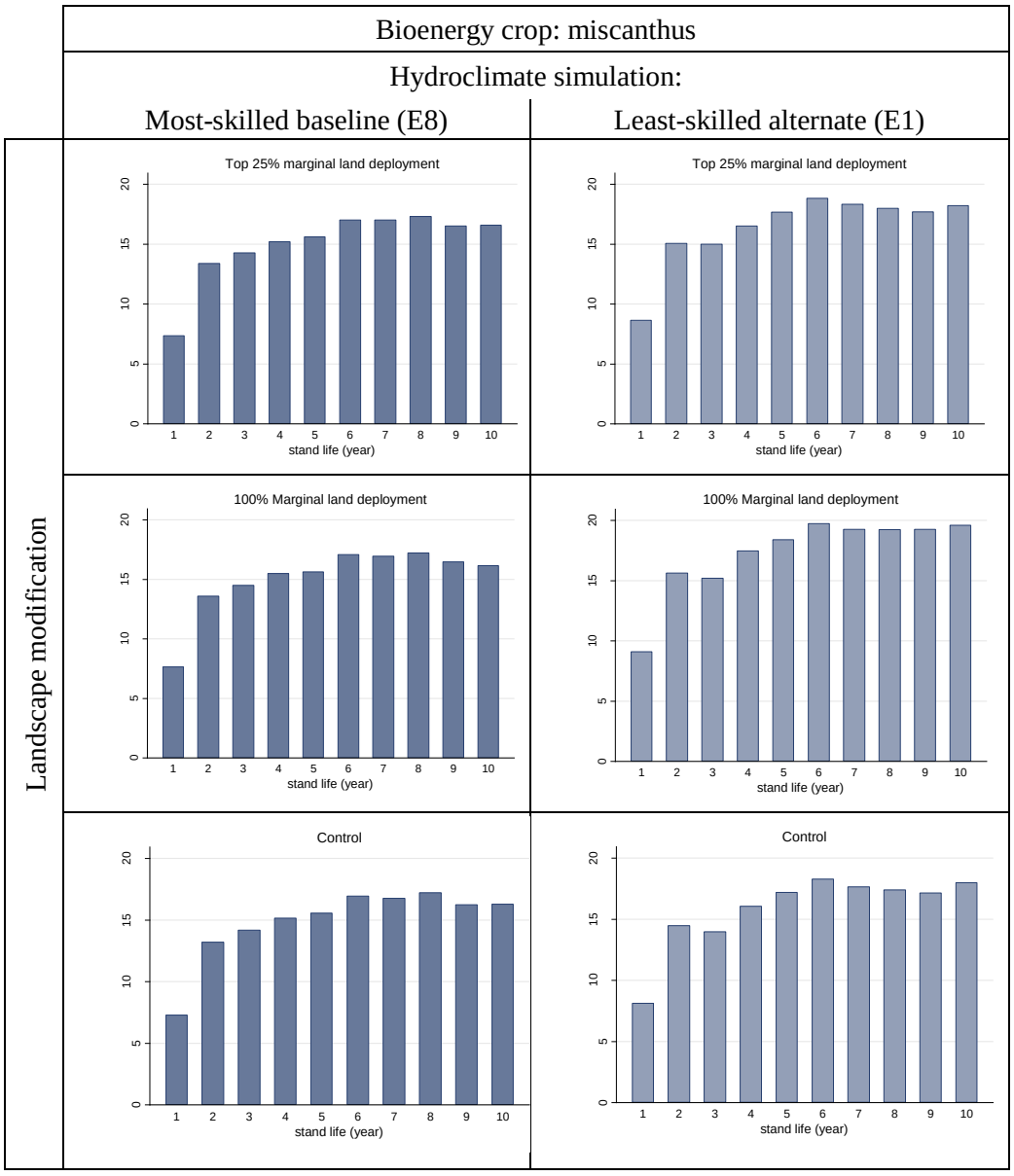


Figure 3.6. Spatially-averaged simulated miscanthus yields. Summarized by bioenergy crop, hydroclimate simulation skill and marginal land deployment (mg/ha) over stand life (year).

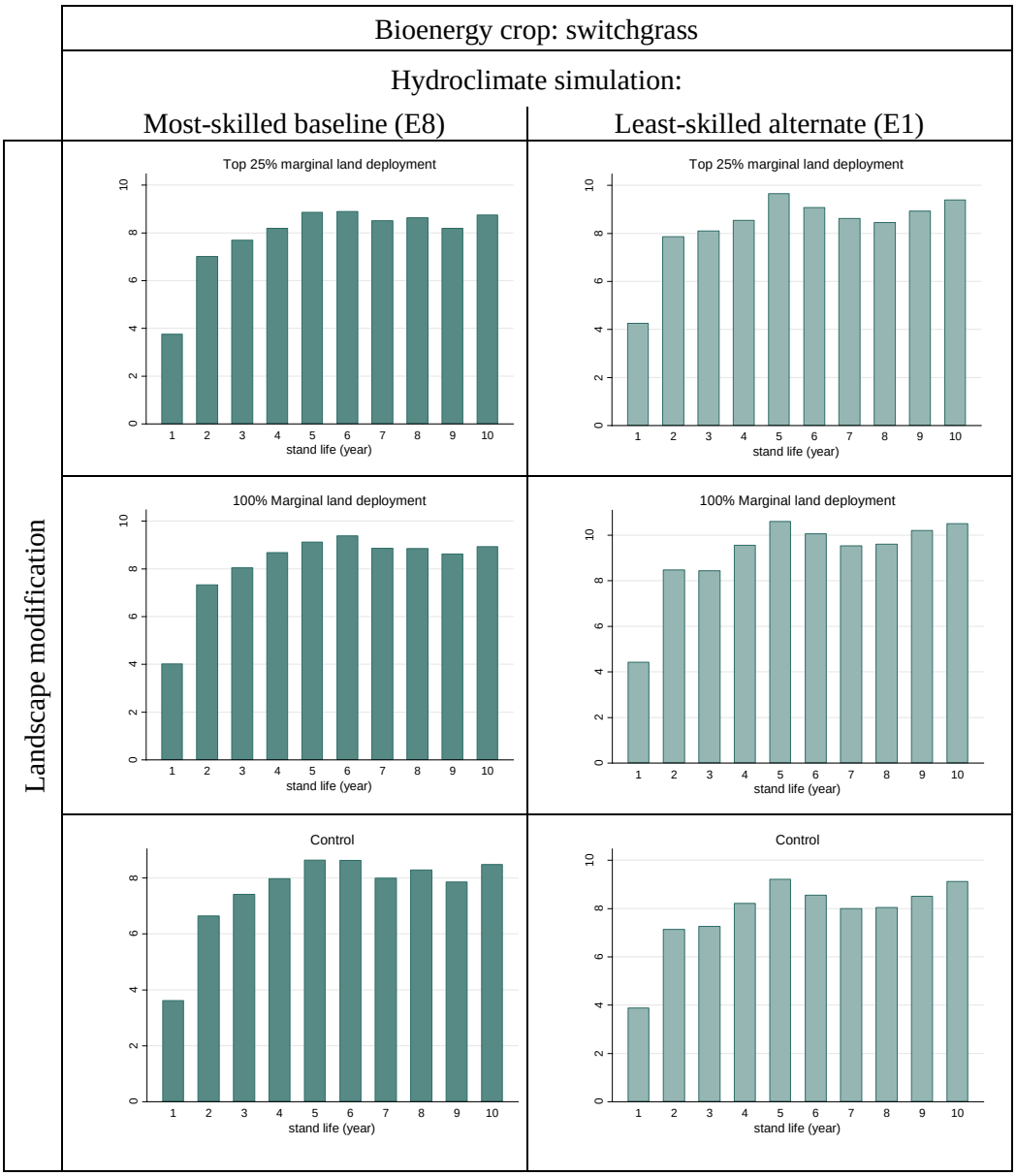


Figure 3.7. Spatially-averaged simulated switchgrass yields. Summarized by bioenergy crop, hydroclimate simulation skill and marginal land deployment (mg/ha) over stand life (year).

Economic optimization and derivation of biofuel supply curves. Is performed by the GBSM, a bottom-up economic model that constructs biofuel supply chains. The model has a flexible, spatially-explicit setup that uses mixed-integer optimization to estimate supply of several types of biofuels (including but not restricted to cellulosic ethanol) under scenarios of resource, technology and policy constraints. The model's objective is to maximize the profit of the biofuel industry over the full supply chain.

At the heart of GBSM is the the supply chain optimization model that performs the siting of and biomass resource allocation to biorefineries, and then distributing the biofuel produced by these biorefineries to the demand centers. Several sub-models provide inputs to this model; the resource assessment (consisting of biomass yields, land assessment and WTA calculations), biorefinery cost and performance specifications, transportation network and cost model, and a model of spatial demand assignment to fuel distribution terminals.

Here, the GBSM is setup to compare scenarios of available marginal lands for cultivating biomass feedstock from miscanthus and switchgrass, and the corresponding biofuel output is cellulosic ethanol. The model compares the industry profitability of using either crop against incumbent land uses over space, and chooses the feedstock that is more profitable.

Therefore, for the purposes of this paper, the two key inputs to the model are (1) marginal land scenarios and (2) the dry matter yields (mg/ha) for miscanthus and switchgrass that are obtained with and without the hydroclimatic impacts due to landscape modification using bioenergy crops (see Table 3.3 for a description of various scenarios under which

yields are simulated). Additionally, the hydroclimate simulations reveal regional differences in near-surface temperature and soil moisture impacts due to landscape modification. In the economic model, these regions are imposed as further sustainability constraints on (marginal) land availability.

The economic model is run as a snapshot in time, with inputs representative of the U.S. land use and the farm and fuel sectors circa year 2012. Because the model is run for one year, it uses the 10-year average of the simulated miscanthus and switchgrass yields by grid cell. The average yields to capture the variable growing conditions over the stand life of each crop. We conservatively include, rather than exclude, the lower yields observed during the establishment year (year 1) yields in the 10-year average (see Fig. 3.6), which lowers the average biomass harvest quantity.

In GBSM, the biofuel supply is optimized to be distributed over the entire CONUS (based on existing wholesale fuel distribution networks), consistent with the simulations in the preceding steps. However, the underlying land base for the biomass cultivation is restricted to Eastern CONUS, to take advantage of rainfed agriculture conditions. Here, we define Eastern CONUS as the portion of CONUS that lies to the East of 104th Meridian West, following administrative (county boundaries).

Land assessment. As a refinement to the 1 km resolution marginal lands data (Cai et al 2011) and to identify incumbent land uses with which bioenergy crops will compete, we use the U.S. Department of Agriculture's Cropland Data Layer (CDL, 30 m resolution) product for 2012. All water bodies and urban, forest and wetland pixels as well as perennial food cropping systems, like orchards, and other fruit and vegetable farms are

excluded from available lands for bioenergy crop cultivation. Wherever such excluded land uses overlap with existing marginal lands defined by Cai et al (2011), we remove those pixels from available marginal lands as well. Next, the remaining cropland uses in CDL are reclassified into 14 major crops (corn, soy, wheat, small grains, etc.), plus a non-cropland land use category that includes land used for pasture, rangelands, and shrublands.

A grid of 10-km size cells over the entire Eastern CONUS is constructed (71,586 cells in total), where each cell constitutes an areal unit of analysis. The cell size is intended to match the spatial resolution of the Agro-IBIS simulations to ensure the variability in dry matter yields for miscanthus and switchgrass is fully captured. The 10-km cell size also means the economic modeling is conducted at the sub-county scale. Generally, the smallest scale of analysis in comparable, recent modeling efforts for U.S. has been the county (e.g., Hudiburg et al 2016). Using a finer spatial scale allows for a more realistic representation of the transportation network and transportation costs in the model.

For each 10-km grid cell, we count (1) the marginal land (Cai et al 2011, 1-km) pixels and (2) the reclassified existing land use pixels (CDL, 30m). Then, we estimate the total marginal land area and total area by incumbent land uses per grid cell based on the respective pixel counts. By utilizing these land use datasets in their native resolutions and not aggregating up to the 10-km scale, we avoid either over- or under-representing the area estimates used in our analysis.

Deriving farmgate "willingness to accept" estimates. The willingness-to-accept (WTA) is the lower bound of price, in $/ton of biomass terms, at which a hypothetical farmer

would switch from the incumbent land use to cultivating miscanthus (or swithcgrass). The WTA, in simple terms, compares the economic return of the incumbent land use with that of cultivating miscanthus (or swithcgrass) to ensure it is profitable to cultivate bioenergy crops.

We calculate the WTA estimates by each grid cell and across each set of simulated dry matter yields for miscanthus and switchgrass (Table M1). This involves estimating the costs associated with the perennial bioenergy crop systems versus the costs of and returns to competing land uses.

We first construct representative 10-year "crop budgets" for miscanthus and switchgrass which include establishment costs (e.g. seed or rhizome purchase, field preparation), operating expenses (input use, like agrichemicals), and costs associated with harvest (e.g., storage, machinery use) (see Hoque et al 2014 and 2015). There are several differences between switchgrass and miscanthus in terms of cultivation and crop management. Unlike switchgrass, miscanthus in commercial settings is generally cultivated from live rhizomes and not seeds, which increases the upfront cost. It also requires more fertilizer (nitrogen) use than switchgrass, particularly in the first three years of establishment. As a result, total costs (summed over 10-years, in 2012 dollar terms) for miscanthus are higher than switchgrass.

A comparable database of crop yields and detailed crop-budgets for the incumbent (i.e., competing) crop- and non-cropland is constructed next. This database combines information collected by various agencies of the U.S. Department of Agriculture. We combine detailed cost information with crop yields and market prices to determine

(annual) returns to incumbent land uses (positive or truncated at zero) [1]. Then, a WTA ($/ton) estimate is derived for each grid cell that has a non-zero available area for cultivating bionergy crops, and under each scenario (Table 1) of simulated dry matter yields. The WTA_{ijkm} is the $/ton price for biomass (whether miscanthus or switchgrass) that equals to the return from incumbent land use [2].

$$Incumbent\ return_{jk} = \left(Price_{jk} * Yield_{jk}\right) - Total\ costs_{jk} \quad [1]$$

$$WTA_{ijkm} = \frac{\left(Total\ costs_{ik} + Incumbent\ return_{jk}\right)}{Dry\ matter\ yield_{ik}} \quad [2]$$

Where:

i = bioenergy crop (miscanthus or switchgrass)

j = incumbent crop or land use (15 categories)

k = grid cell with non-zero available land for bioenergy crop cultivation

m = simulated yield scenario (12 scenarios)

$Total\ costs_{ik}$ and $Incumbent\ return_{jk}$ are discounted (at a rate of 5% per annum) to present (here, year 2012) and summed over 10-years.

Because there are not many detailed cost estimates for growing miscanthus in different parts of the U.S., we represent spatial variation in total costs for these crops by substituting the specific land charge (cost of land) used in the crop budgets with county-level cash rents from the USDA. We further tailor this land charge with respect to the incumbent crop type / land use. For instance, if miscanthus (or switchgrass) is competing with both corn and pasture lands in each grid cell, we use cash rents for croplands for that county for the WTA calculation involving corn, and use cash rents for pasture lands for

the same calculation involving pasture lands. Additionally, if an incumbent crop is traditionally irrigated (e.g. rice), representative cash rents for irrigated croplands are used in the corresponding WTA calculations.

We develop a cropland supply - rent (price) elasticity estimate that allows the cost of acquiring the biomass feedstock to increase linearly with the demand for feedstock. As croplands are converted into perennial grasses, a reduction in supply of crops would cause crop prices to rise. This would increase returns to growing crops, and in turn, land owners would also demand higher rents for cropland, which would also increase feedstock costs for bioenergy crops.

There is no comparable scale of land use change that took place in the U.S. (e.g. >20 million ha) over a relatively short period of time that would serve as a benchmark to model the price response of supply (land) after. However, market response to the introduction of the ethanol quantity quota in 2005 is illustrative of potential implications. The U.S. farmers responded strongly to the policy change by planting more corn – not just on existing crop and pasturelands but also by clearing new croplands (mainly out of grasslands) in the span of 2-3 years. At its peak, around 2007-2008, the total expansion in croplands exceeded 7 million acres U.S.-wide. While much of this was the result of idled and pastured croplands being brought into production, about 1.7 million acres were estimated to be conversions from previously uncultivated grasslands. The main crop planted in these incremental croplands was corn (Lark et al 2015). Even though land use for corn cultivation expanded, the diversion of corn output for ethanol production was large enough to reduce the amount available for food. This also contributed to higher

crop prices (Condon et al 2015, Roberts & Schlenker 2013). A similar dynamic could play out if the displaced output, particularly of corn and soy, due to bioenergy crop conversions of croplands exceeds the amount currently used for ethanol and biodiesel production.

For our analysis, we conducted a fixed effects linear regression of average cash rents (in 2012 dollars) on cropland size (acres) for a panel of U.S. states, using data from 1994 to 2016, assuming a two-year lag for rents to respond. The estimated coefficient –rent elasticity estimate for cropland-- for using all states in the sample was -0.38 $/thousand acres ($p<0.05$) (i.e., the coefficient is negative because rents increase with decreasing amount of cropland). For a subset of seven corn belt states, the estimate was higher at -4.28 $/thousand acres ($p<0.05$). In our model, we used $5 per thousand acres as an indicative land response adder.

County-level crop yields, state-level received prices by crop, and county-level non-irrigated pasture land rents are obtained from USDA NASS. Wherever available, crop-specific, county-level yield information is used. In other cases, crops with data availability at the county level are used as a reference for other, similar crops (e.g., sunflower becomes a stand-in for non-soy oil crops, and beans for legumes). Representative crop-level costs for 2012 by farm resource regions are obtained from USDA Economic Research Service.

Other inputs to GBSM. These include the engineering and economic characteristics of a cellulosic ethanol biorefinery; potential biorefinery sites, transportation costs, and fuel demand.

The engineering-economic model of biorefineries characterizes of the cost and performance of each biorefinery that would be built. These parameters are developed based on literature (e.g. Humbird et al 2011) and our review of information reported on facilities that were built. In our analysis, the single conversion technology assumed is a biochemical pathway for a generic cellulosic biofuel. Based on the data, the fixed and variable costs of production are described as linear functions of the input capacity of the biorefinery in units of tons of feedstock per year. The intercept and slope are parameters for cost functions used in the optimization model. These cost estimates representative only for the analyzed range of facility capacities.

The model avoids the construction of biorefineries below the minimum size (2,000 bdt/day), because these would have very high average cots. The upper limit on plant size is specified as a constraint in the model.

The capital costs are annualized using discount rate of 10% and over the economic life of the biorefinery (20 years). Operating and maintenance costs include the non-feedstock input (enzyme) costs. The annualized costs of production are consistent with the size range considered valid for cellulosic ethanol biorefinery technology with a minimum feedstock (input) capacity of 2,000 bdt/day. The engineering and cost parameters are summarized in Table 3.2.

The key structural components of biomass are cellulose, hemicellulose, and lignin. Cellulose and hemicellulose are the portions converted to biofuel and lignin is a co-product that can be used for biopower generation. Table 3.3 shows feedstock composition for miscanthus and switchgrass assumed in our analysis.

Table 3.2

Engineering and cost parameters for model biorefinery

(a) Engineering-economic parameters for biorefineries	Base scale	Maximum scale
Plant size (tons of dry biomass per day, bdt/day) (C)	2,000	3,600
Ethanol output (million gallons per year)		112.5
Capital Cost at base scale ($ million)	550	973
Scaling Factor	0.88	
Fixed O&M ($/mgey)*	3.6917 + (C * 0.4528)	
Variable Costs ($/gal)*	0.4356 + (C * 0.0595)	
Cellulose to ethanol conversion efficiency	72.0%	
Hemicellulose to ethanol conversion efficiency	76.5%	
Cellulose to ethanol conversion factor (gallons product per ton feedstock)	172.85	
Hemicellulose to ethanol conversion factor (gallons product per ton feedstock	176.9	

(*) *Linearized as function of feedstock capacity.*

Table 3.3

Feedstock composition assumptions for miscanthus and switchgrass

(b) Feedstock composition	**Miscanthus**	**Switchgrass**
Cellulose	0.4080	0.3321
Hemicellulose	0.2598	0.2635
Lignin	0.1000	0.1813
Higher heating value (HHV)	19.7	18.7

When selecting potential biorefinery sites, the goal is to include all potentially optimal ones, but limit the total number to reduce computation time. Our approach identifies potential locations for a biorefinery based on logical criteria, and then eliminates redundant locations. We consider potential biorefinery sites only within Eastern CONUS,

because it is easier (cheaper) for the biofuel to be transported over long distances and not the biomass feedstock.

The decision criteria for identifying potential biorefinery locations are (1) proximity to urban centers with population >10,000 to supply the workforce, (2) access to railroads and pipelines for natural gas, petroleum products or hydrocarbon gas liquids (HGL), or (3) existing comparable industrial facilities nearby (power plants, petroleum product terminals, biodiesel and ethanol plants, petroleum refineries, and paper and pulp mills). Using GIS, we determine how many of our 10-km grid cells satisfy these criteria. For (1), the threshold is the share of population within the 3x3 neighborhood of each grid cell that is >10,000. For (2), we consider a grid cell if a major railroad or pipeline intersects the 3x3 neighborhood surrounding it. Meeting criterion (3) means the 3x3 neighborhood of a grid cell contains at least one of the specified industrial facilities. We then develop a scoring system, where a grid cell that satisfy all three criteria would score ≥3 (a score above 3 is possible because a grid cell may contain multiple instances of industrial facilities, or both an industrial facility and a pipeline intersecting it). We filter out grid cells that score ≤1.

We then eliminate potential locations that are in proximity (within 50 km) of each other. This saves the additional computing time needed for the model algorithm to choose between two nearly identical outcomes. During this elimination, whenever two (or more) sites are within 50 km of one another, we favor the site that had highest score. We end up with 1,170 potential sites in Eastern CONUS.

The transportation costs are modeled using the Network Analyst feature in ArcGIS. The transportation network consists of a GIS network of road routes, as we anticipate biomass will be transported by truck while the fuel is transported by rail. The unit costs for all potential links – cost of transporting one ton of biomass between each biomass supply point and biorefinery or one gallon of biofuel between each biorefinery and fuel distribution terminal – are calculated over the transportation network. We restrict the transport network between biomass supply points and biorefineries to those (origin, destination) pairs that include the 20 nearest biorefinery to a given supply point, or links with a total transportation cost less than $40/dry ton. Costs include loading and unloading in addition to the distance dependent costs. [3]

Total transportation $cost_{od}$ ($/dry ton) = time cost + distance cost + loading cost [3]

Total transportation $cost_{od}$ ($/dry ton) = [(2 * 3.90 $/hour) * hours] + [(0.038 +0.037) $/dry ton/mile * miles] + $3.24/dry ton

Where o stands for origin and d for destination.

The spatial distribution of biofuel demand is modeled by assigning census tracts (population) to fuel distribution terminals using a service area analysis. The locations of fuel distribtion terminals are known (https://www.irs.gov/businesses/small-businesses-self-employed/terminal-control-number-tcn-terminal-locations-directory). Fuel demand for road transport is an imperfect proxy for aviation fuel demand. However, the finest granularity data on aviation fuel consumption in space is at the state level. We err on the side of using data with a higher spatial resolution. To the extent that aviation fuel demand

is associated with presence of large airports, which tend to be correlated with large metropolitan areas (population), our approach is reasonable. Table 3.4 describes the complete scenario space for the integrated modeling.

Table 3.4

Scenario space for the integrated modeling

Modeling sequence		Landscape modification / marginal land deployment	Hydroclimate simulations	Ecosystem modeling	Economic optimization
		1km, CONUS	20km, CONUS	10km, CONUS	10km, CONUS
Scenario family	Controls (No landscape modification)	Existing land cover/ land use	(1) Most-skilled baseline (E8) and (2) least-skilled alternate (E1) in replicating contemporary climate	Miscanthus yields, using climate forcing from most (mcE8) and least-skilled (mcE1) simulations (sgE8, sgE1) Same as (mcE8, mcE1) but simulating switchgrass yields instead	Using mcE8 yields over: • top 25% of marginal lands • 100% of marginal lands and • no marginal land constraint (all possible lands)
	High Suitability Deployment	Top quartile of suitable marginal lands (top 25%)	(3) Under landscape modification of top 25% of suitable marginal lands, using most-skilled simulation parameters (4) Same as (3), but using least-skilled simulation parameters	Miscanthus yields, using climate forcing from most (mcE8025) and least-skilled (mcE1025) simulations, each under landscape modification of top 25% of suitable marginal lands (sgE8025, sgE1025) Same as (mcE8025, mcE1025) but simulating switchgrass yields instead	Using mcE8025, mcE1025, sgE8025 yields over: • top 25% of marginal lands and • top 25% of marginal lands overlapping with region 4* (see Fig. 1 for regions) (*) Regions 3 and 5 omitted because they do not significantly overlap with the top 25% of marginal lands
	Full Deployment	All suitable marginal lands (100%)	(5) Under landscape modification of 100% of suitable marginal lands, using most-skilled simulation parameters (6) Same as (5), but using least-skilled simulation parameters	Miscanthus yields, using climate forcing from most (mcE8100) and least-skilled (mcE1100) simulations, each under landscape modification of 100% of suitable marginal lands (sgE8100, sgE1100) Same as (mcE8100, mcE1100) but simulating switchgrass yields instead	Using mcE8100** yields over: • 100% of marginal lands and • 100% of marginal lands overlapping with regions 3 or 4 or 5 (see Fig. 1 for regions) (**) Limited to yields under baseline hydroclimate simulation (mcE8100)

CHAPTER 4

ROLE OF LAND QUALITY IN CORN ACREAGE RESPONSE TO PRICE AND POLICY CHANGES: EVIDENCE FROM THE WESTERN CORN BELT

ABSTRACT

Land quality influences how farmers allocate croplands in response to market forces. Farmers in the Western Corn Belt (WCB) have historically utilized the highest quality lands for corn cultivation, putting lower quality lands to other uses. This paper questions whether high corn prices influenced expansion of corn cultivation on lower quality lands, and the role played by US biofuel policy in such land use change. Using three decades of data, I estimate that the proportional change in corn acreage to rising corn prices is nearly three times larger in counties with lowest land quality versus those with the highest land quality. This variable response, however, is driven by the changes in cropland use for corn cultivation after 2006, the period following the change in US biofuel policy, and punctuated by two crop price spikes. Marginal agricultural lands and other lower quality lands, such as grasslands used for range or pasture, are therefore prone to conversion into corn cropping disproportionately during high price periods. High price responsiveness of lower quality lands also suggests these lands may cycle in and out of corn cropping opportunistically. This has implications for marginal land availability for bioenergy crops, and poses environmental concerns to the extent lower quality lands are also more environmentally sensitive.

4.1 Introduction

Globally, corn prices rose between 2005 and 2008 from $2 per bushel to nearly $8 per bushel, with similar price increases observed for soy, wheat, and rice. Since most of the world food supply is connected to these staple grains, the higher crop prices meant higher food prices for many, with the burden falling disproportionately on the poor who spend a larger share of their income on food (Mitchell 2008). Between 2008 and 2010, high prices for food items led to protests across the world (Baker 2008, the UN 2011). The World Bank (2011) predicted high prices might have pushed as many as 44 million people into extreme poverty.

Biofuel mandates that had recently been introduced in the US and the EU contributed to the increase in crop and food prices (see Roberts and Schlenker 2013; the National Research Council 2011; Chen and Khanna 2013; and Condon et al 2015), and in the US, there was a sharp increase in the use of corn as the feedstock for ethanol production. The policy has transformed US corn production, impacted world food supply, and reshaped agricultural land use within and outside of the US (Khanna and Crago 2013 and Nuñez et al 2013), despite a subsequent restriction on the share of corn ethanol in total biofuels quota in 2007.

The US biofuel policy had been motivated by the desire to reduce dependence on fossil fuels and mitigate climate change, but the conversion of previously uncropped lands into the cultivation of corn may have been an unintended consequence of its implementation (Wright and Wimberly 2013; Lark et al 2015). Such land use/land cover change has adverse consequences for natural habitats (Meehan et al 2010), while also adding to the

planet's carbon debt by possibly reversing the environmental benefits of a switch to ethanol from gasoline (Searchinger et al 2008 and Fargione et al 2012).

One way to mitigate the negative land use effects of the ethanol industry is to use perennial bioenergy crops (miscanthus and switchgrass) grown on marginal, abandoned, or idled agricultural lands as feedstock rather than corn (Field et al 2008, Somerville et al 2010, Gelfand et al 2011, Georgescu et al 2011, and Wang et al 2017). Earlier studies estimated the supply of such lands at the global and US-scales (Cai et al 2011; Zumkehr and Campbell 2013). Yet, some of these newly added corn acres could also be encroaching on available marginal lands. Under current policy, about 90 million acres of corn is harvested every year in the US, compared to fewer than ten thousand acres dedicated to perennial bioenergy crops (USDA NASS Census of Agriculture).

Both previously uncropped lands and marginal or abandoned agricultural lands are likely to be of lower quality and more environmentally vulnerable. Such lands tend to have more sloped and erodible soils, experience more nutrient runoff, and may harbor more species of imperiled plants and animals than croplands that remain in cultivation (Lubowski et al 2006a). The features that make these lands more susceptible to environmental degradation may be exacerbated by farming practices. For example, sloping lands are prone to erosion and difficult to farm, and their very erodibility may be worsened by farming (Pimentel 2006).

All else equal, farmers would not choose to cultivate on lower quality lands because of their poor productivity and higher upfront conversion costs, both of which translate into lower profitability. However, when crop prices are high it becomes easier to clear these

hurdles. Thus, an interesting question is how corn cultivation has expanded on lower quality lands in response to high corn prices, and whether such response has varied before and after the change in US biofuel policy, which now routinely consumes over a third of US corn output. If the lands that move in and out of corn cultivation have lower productivity, are more environmentally vulnerable, and at the same time more sensitive to high prices, then high prices would generate larger environmental impacts while delivering limited economic (output) benefits (Lubowski et al 2006a).

Earlier economic studies that considered land quality and acreage response did so in the context of conversion of non-croplands into croplands. For example, in a national study, Lubowski et al (2006b) find the category least likely to convert into croplands are rangelands, which are of lowest quality. Claassen et al (2011) report higher economic responsiveness for rangelands to transition to croplands when studying 77 counties in the Dakotas for a more recent time period. Miao et al (2016) also estimate corn acreage response to price, but their analysis ends in 2007 and does not investigate the role of land quality. This paper considers the influence of land quality on acreage decisions involving a major crop (corn) in the WCB rather than broad categories of land use; it covers the post-biofuels period that has experienced not one but two crop price spikes; and combines high resolution spatial and high frequency time series data in its analysis. In addition, it revisits the 2007 change in US biofuel policy and examines the policy's land use impacts on the use of marginal lands and grasslands over a longer timeframe in the economically and environmentally significant WCB region.

4.2 Data and methods

4.2.1 Study area

Consisting of Iowa, Minnesota, Nebraska, North Dakota and South Dakota, the WCB is an important US corn producing region. It represents 44% of corn acres planted and nearly 50% of corn production in the US (USDA NASS). The WCB is environmentally significant, as Nebraska, North Dakota and South Dakota contain almost all (99.5%) of the remaining tall and mixed grass prairie in the Corn Belt (Mac et al 1998) and ecological hot spots such as the Prairie Pothole Region (PPR) are within its boundaries. The WCB also has a higher concentration of marginal agricultural lands suitable for bioenergy crop cultivation (Cai et al 2011), a more sustainable alternative to corn for producing biofuels.

4.2.2 Land quality metric and GIS analysis

There are several acceptable measures of land quality. Some previous studies use focused measures like topography (Muller and Zeller, 2002), soil water holding capacity (Lichtenberg 1989), or yield proxies like the NCCPI (Claassen et al 2013). Here, I use the non-irrigated land capability classes (NICC) from the Natural Resources Conservation Service's gridded 90m soil data (gSSURGO) product (Soil Survey Staff 2016). The NICC is a broad-based measure of land suitability for agriculture that captures soil and climate characteristics along with land management considerations. It has been used widely in the economic analysis of land use change literature (Hardie and Parks 1997; Lubowski et al 2006a, 2006b, 2008; Lewis and Plantinga 2007; Claassen et al 2011 and Radeloff et al 2012). It is a categorical measure that ranges from 1 to 8, where 1 indicates

the highest land quality (most suitable for cropping), and 8 indicates the lowest (least suitable). Classes 4 and below are arable lands, although classes 3-4 are considered to have severe to very severe limitations for cropping. Best use for classes 5 and above is pasture/range.

Here the terms NICC score, average NICC score or mean NICC score are used interchangeably. They all correspond to land capability classes either at the 90m spatial resolution or an area-weighted average computed at coarser scale (e.g. county). To assess the distribution of land quality in the WCB, I perform a frequency analysis of NICC scores at the 90m resolution for the entire region. The NICC scores are then aggregated to the county (US Census Bureau) scale on an area-weighted basis for econometric analysis. GIS analysis of the land quality by land use utilizes fine spatial scale satellite land use data from the CDL (USDA NASS Cropland Data Layer) and coarser scale county-level data on corn acreage (USDA NASS) with a longer history. For consistency, I use the CDL product from 2010 to 2016, resampled from 30m to 90m to match the resolution of gSSURGO. (Prior year CDL products had different specifications, and comparisons across products were not recommended (Kline et al 2013). In 2017, CDL for 2008 and 2009 were reissued at 30m. As this extension does not cover the pre-biofuel period, the analysis is limited to post-2010).

Because the CDL contains over 100 distinct land use categories, land uses are first reclassified into groups: corn (including double croppings); soy or wheat; other field crops (e.g. oats); all other crops (e.g. vegetables); cultivated hay (including alfalfa); idle/fallowed croplands; grass/pasture; and non-cropland uses (e.g. urban). Then, the

area-weighted average NICC scores is calculated for each combination of crop group and year for the entire study area and at the county level.

4.2.3 Price response of corn acreage under variable land quality

Price elasticity measures percent change in the supply of a product in response to percent change in its own price. Because the supply of a crop--here corn--is the product of its acreage and yield, one can estimate the price elasticity of acreage by substituting acreage for total supply.

If higher corn prices result in an increase in area under corn cultivation (positive price elasticity of acreage), does such response vary with land quality? The hypothesis is that the acreage response to higher corn prices would be greater in areas with lower and more heterogeneous land quality compared to those with uniformly high land quality. In an area where land is relatively homogeneous and high quality, farmers would already have cropped most of the land, and there might not be much additional acreage for expanding corn cultivation. While more land might be available for corn cropping in areas of heterogeneous land quality, these would be of lower average quality. When corn prices rise, however, farmers can justify using these lower quality lands that might have been unprofitable corn cropping before.

To test this, I develop a multivariate model of price response that relates corn acreage and corn prices. My sample includes 396 counties in the WCB (out of 398, as two reported no corn acreage for the sample period) and covers 1986 to 2015.

The dependent variable is planted (not harvested) corn acreage because it is more representative of the farmers' planting decision. During the sample period, planted corn

acres in the WCB rose by 6.9 million acres (23%). At the county level, there is considerable variation in planted acres. The top quartile of counties plant corn, on average, five times more than the bottom quartile, though the gap has narrowed since 2006 (Fig. 4.1A).

The main explanatory variable is the state-year average of received corn prices (USDA NASS). Comparable price series for major crops grown as alternatives to corn are control variables. Based on the average planted acreage over 1986-2015, these alternative crops are soy, oats and hay (Iowa) and soy, wheat and hay (Minnesota, Nebraska, North Dakota, and South Dakota). Prices for corn and these major alternative crops follow similar trends during 1986-2015 (Fig. 4.1B).

I use one-year lagged prices to capture farmers' expectations of current prices, which are formed prior to planting season. Use of lagged prices also helps control for the simultaneous causal relationship between current year supply (acreage) and price. If there are unobserved supply shifts in the current year–e.g. a reduction in corn harvest due to an unanticipated drought–these would affect the current price. Using current prices in the model would thus cause price and the error term to be correlated, creating a biased price elasticity estimate. However, a shift in supply in the current period could not have influenced past prices.

The fertilizer price index (USDA NASS) is used as a proxy for changing input costs. Corn is a fertilizer intensive crop, making this index an appropriate measure of input costs (Fig. 4.1C).

All price variables are in 2015 dollars obtained by deflating nominal prices by the consumer price index (Bureau of Labor Statistics 2016).

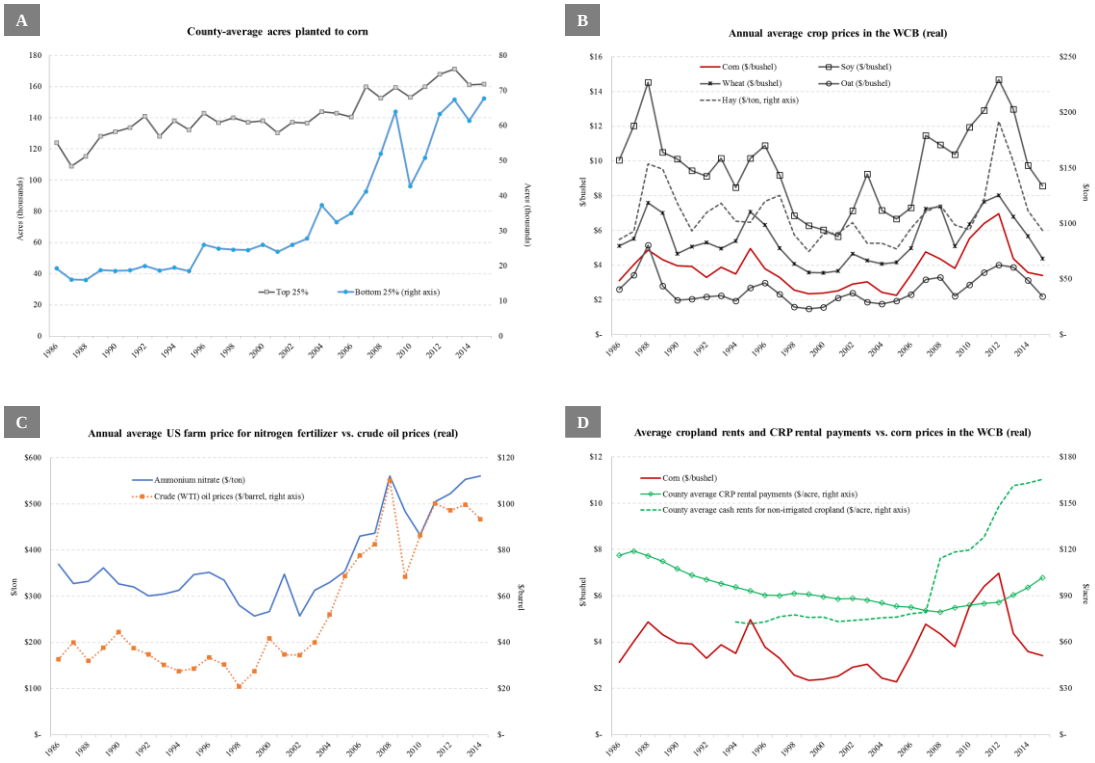


Figure 4.1. Key data used in econometric analysis. (A) Trend in corn cultivation for the top and bottom quartiles of WCB counties in terms of planted corn acres. (B) Annual average of state-level prices for corn, soy, wheat, oat and hay, in real terms, 1986-2015. (C) Annual US average farm price for ammonium nitrate, a widely used nitrogen fertilizer, and crude oil prices, in real terms, 1986-2014. (D) County average cash rents for non-irrigated cropland, county average CRP rental payments versus corn prices (same as A and B), in real terms, 1986-2015. All price series are from the USDA NASS, except the West Texas Intermediate (WTI) crude oil prices, which is from BP (2016).

In the US, federal policy plays an important role in shaping farmers' land use decisions. I consider three relevant policy events. Acreage reduction programs compensated farmers to idle otherwise productive croplands during periods of low prices. The latest

incarnation, known as “set asides”, was eliminated in the 1996 farm bill (FAIR Act). This is represented by a dummy variable that takes on the value of 0 before 1996, and 1 after. The Conservation Reserve Program (CRP) is also a type of acreage reduction program. Under CRP, farmers are paid to temporarily remove cropland from production, though the program’s main goal is environmental, and limiting surplus crop production and providing income to farmers are secondary (Goodwin and Smith 2003). I include the county average CRP rental payments (USDA Farm Service Agency) in the model. These payments are customized to each contract’s environmental benefits index (EBI) score, but pegged to past three-year’s non-irrigated cropland rental rates in the county (Fig. 4.1D). Another dummy variable captures any non-price related effects of the Energy Policy Act of 2005 (EPAct), which created an annual quantity mandate for biofuels, mainly ethanol, to be blended with gasoline. In the US, the primary feedstock for ethanol continues to be corn.

I control for pre-planting (April average) precipitation, since excess moisture adversely affects the decision to plant corn. The spatially interpolated, gridded daily precipitation data (2.5-mile cell size) is from Schlenker and Roberts (2009), updated through 2015. The model is a multivariate panel regression of county-year corn acreage data on corn prices and these control variables, estimated using ordinary least squares (1).

$$(1)\quad A_{jt} = \beta_1 corn_{j,t-1} + \beta_2 soy_{j,t-1} + \beta_3 wheat/oat_{j,t-1} + \beta_4 hay_{j,t-1} + \beta_5 fert_t + \beta_6 CRP_{j,t-1} + \Psi_1 ppt_{j,t} + \beta_7 D1996 + \beta_8 D2006 + \varphi_j + f(t) + \nu_{jt}$$

The dependent variable is the natural log of planted acres of corn, A_{jt} , in county j and year t. The explanators are $corn_{j,t-1}$, the log of lagged received state-level corn prices; and $soy_{j,t-1}$, $wheat/oat_{j,t-1}$, $hay_{j,t-1}$ the similar price series for alternative crops; and w_t, the fertilizer price index. The CRP variable is the log of county-level CRP rental payments. The log-log specification allows us to estimate percentage changes linearly. Average April precipitation is represented by $ppt_{j,t}$. The dummy variable for the FAIR Act is labeled D1996, and D2006 marks the pre/post-biofuels period. Finally, $f(t)$ is the linear time trend and v_{jt} is the error term. The coefficient on corn price (β_1) is the measure of price elasticity, which predicts percentage change (increase) in acreage given percentage change (increase) in crop prices.

Land quality variables are not included in the model directly. Instead, the model is estimated for sub-samples of counties based on quartiles of county-average NICC scores. The quartile breakpoints reflect the benchmark NICC scores by land use derived from the 2010-2016 CDL data. The breakpoint between the 1st and 2nd quartiles (2.65) is comparable to the average NICC score for corn-planted areas. Moving from the 2nd to the 3rd quartile, the breakpoint (3.05) is indicative of the transition from active to idled/fallowed croplands and hay. The NICC scores for counties in the 4th quartile (3.97 and above) signal shift into grass/pasture. Also, the county average corn yield declines with each quartile. The yield differences between quartiles are statistically significant (pairwise comparison of means, $p<0.01$), indicating each quartile is distinct in terms of land productivity.

The county-level fixed effects (φ_j) control for average differences across counties in (observable or unobservable) predictors that are not explicitly included in the model. This insures against a type of omitted variables bias in case there are fixed parameters common to the sample counties. Analyzing a large sample of counties (99 for each land quality quartile) increases statistical power of the model and generates an "average effect" for the sample.

4.3 Results and discussion

4.3.1 Analysis of land quality

Land quality in WCB is, on average, suited to cropping. The area-weighted average NICC score for the entire region is 3.75 with 72% of all lands belonging to class 4 or lower (Fig. 2A). The higher quality lands are concentrated between the rivers Upper Mississippi and the Missouri, and lie in Iowa, southwestern Minnesota, and the eastern portions of South Dakota, North Dakota, and Nebraska.

Satellite data from the CDL for 2010-2016 permits a high-resolution assessment of how land use and land quality interact. The mean NICC score for corn planted areas during any of these 7 years is 2.45: lower and statistically different from the mean score for the entire WCB. The lower standard deviation of NICC scores for corn cultivation indicates the activity is concentrated in high land quality areas (Fig. 4.2B). Ninety-six percent of all corn cultivation in the WCB takes place on arable lands (NICC score ≤ 4). The majority of the remaining 4% occurs on lower quality lands in Nebraska. There, farmers

can irrigate crops using groundwater from the Ogallala Aquifer, offsetting the effect of lower land quality (Lichtenberg 1989).

Over a third of WCB was planted to corn at least one year between 2010-2016. Most often, farmers planted corn in the same fields 3-5 years out of the 7 analyzed here. Opportunistic cultivation of corn (1-2 years out of 7) is less prevalent, and so is near-continuous cultivation of corn (6-7 years out of 7). Corn is still likely planted in rotation with other crops. Moreover, a substantial portion of arable lands were never planted to corn (34%) (Fig. 4.2B).

Benchmark NICC scores by different agricultural land uses in the WCB indicate corn cultivated areas to have the highest land quality and grass/pasture the lowest (mean NICC scores of 2.45 vs. 4.75). The average NICC score for the other two main cash crops, soy and wheat, is slightly higher at 2.49, followed by other field crops, croplands left idle or fallowed, and cultivated hay (Fig. 4.3A). Each pair of means are statistically different from each other at 1% based on t-tests. Non-parametric ranksum test also indicates corn is different from all other land use categories in terms of land quality (at $p<0.01$), except for the soy/wheat planted areas.

Clearly, farmers consider land quality when choosing land use, and allocate the highest quality lands to the primary cash crops. If farmers have been allocating increasingly poorer quality lands into corn cultivation over the years, using a more recent (post-2010) estimate of average land quality would tend to raise the average NICC scores. However, the relative positions of different land uses with respect to land quality would remain comparable.

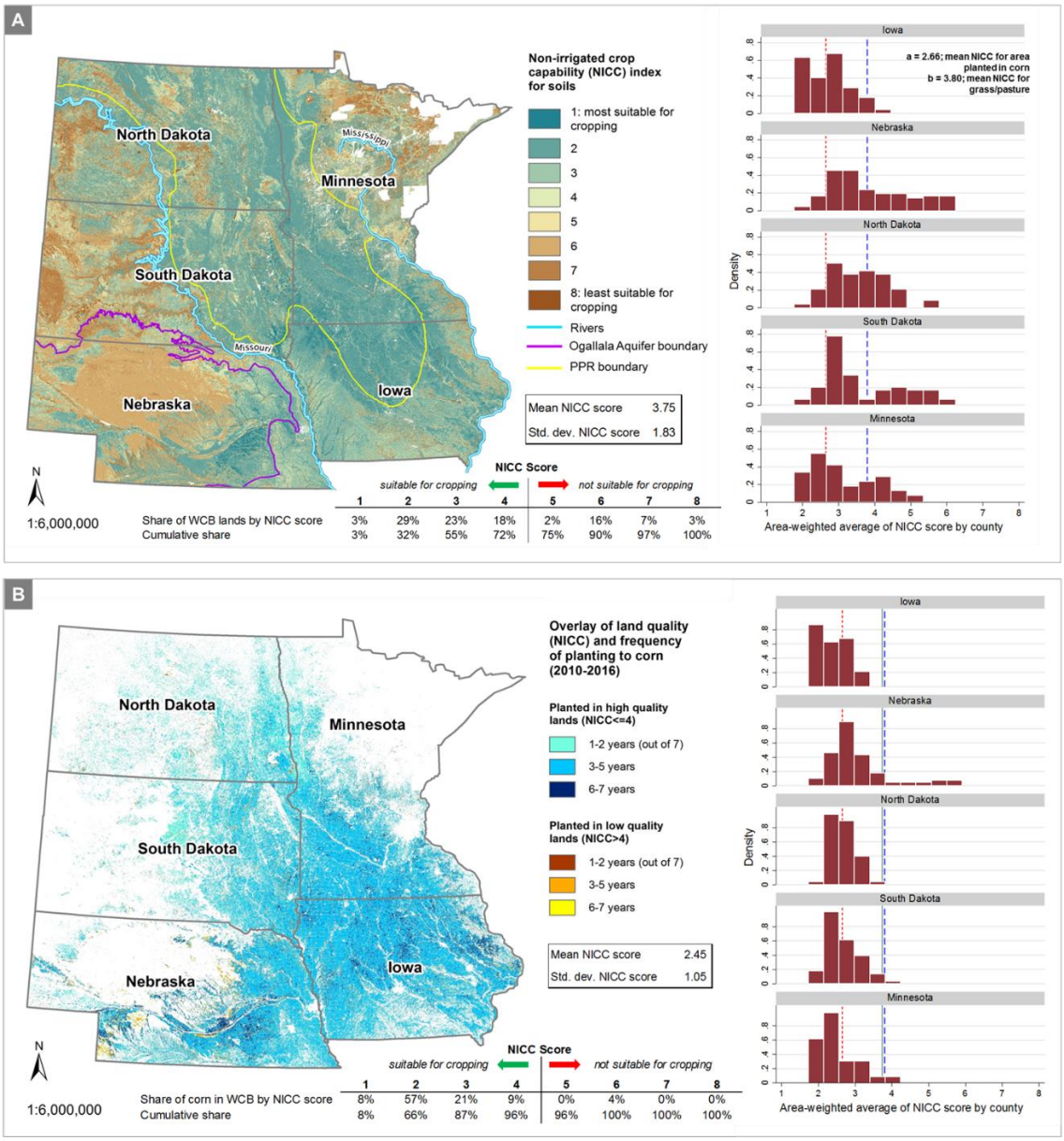


Figure 4.2. Land quality and corn cultivation in the WCB. (A) Land quality in the WCB based on the non-irrigated crop capability (NICC) score (90m resolution). The rivers Missouri and Mississippi, and boundaries for the Ogallala Aquifer and the Prairie Pothole Region (PPR) are denoted. (B) Areas planted to corn in the WCB from CDL (2010-2016) grouped by frequency of planting and overlaid with NICC scores high (≤4) or low (≥4). The right-hand side histograms in both (A) and (B) depict the distribution of county-average NICC scores by state with reference lines that correspond to (i) areas planted to corn and (ii) lands identified as grass/pasture (CDL 2010-2016), respectively.

Since satellite-based cropland data does not go as far back as survey-based, county-level land use data, I pair the high-resolution data on soil quality with high temporal resolution county crop acreage data from the USDA for the econometric analysis. Due to aggregation and within-county variation in land quality, the mean NICC scores by land use are slightly different at the county level (2.66 for corn and 3.80 for grass/pasture as bookends). Of the 396 WCB counties in the sample, 103 have average land quality scores at or below 2.66. Most are in Iowa (46) and Minnesota (34). Another 110 have an average NICC score>3.80, the benchmark score for grass/pasture, majority of which are in Nebraska (40) and South Dakota (23). Nearly half of them (183) have average NICC scores that fall in between (see histograms in Fig. 4.2A).

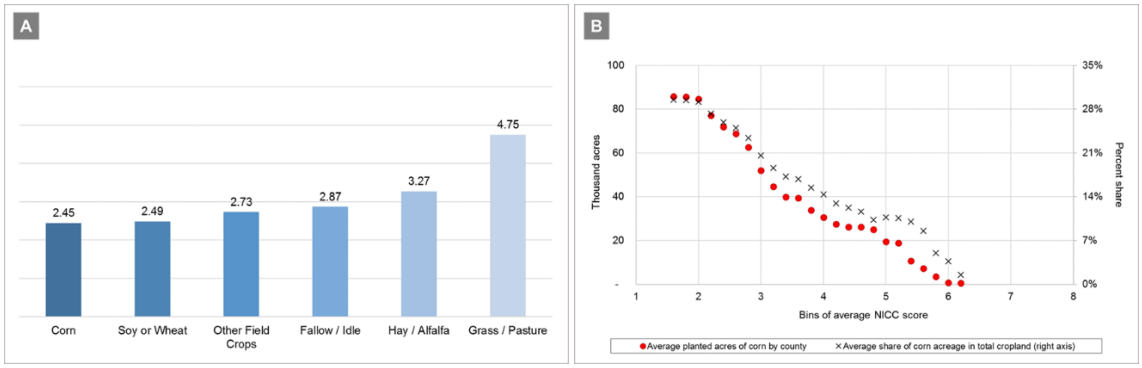


Figure 4.3 Land quality by land use in the WCB. (A) Average NICC scores calculated for different major agricultural land uses based on observed land use 2010-2016 from the CDL. (B) Corn cultivation by land quality using county level acreage data. Round markers show average planted corn acreage (using annual survey data over 1986-2015, USDA NASS, left axis). Cross markers depict average share of corn acreage in total county cropland (using 5-yearly data from the USDA NASS Census of Agriculture, 1982-2012, right axis). The x-axis shows bins of average NICC score.

County level data confirms the link between higher land quality and corn cultivation. Between 1986-2015, the better the average land quality in a county, the more acres have been planted to corn. This pattern is not driven by cropland size, since the relationship holds for the share of corn cultivated areas in total cropland by county (Fig. 4.3B).

4.3.2 Analysis of price response

The response of corn acreage to higher corn prices—the coefficient estimate on corn price—is positive and significant in the full sample as well as for the four quartiles of county-average NICC scores (Table 4.1). The positive sign and the magnitude of the coefficient is consistent with the literature for the supply of crops, and specifically, acreage (Chavas and Holt 1990; Chembezi and Womack 1992; Lin and Dismukes 2007 and Miao et al 2016). A positive price elasticity indicates farmers' ability to expand corn acreage in response to rising corn prices. Additionally, the higher the price elasticity, the more flexibility farmers have in expanding acreage.

Moreover, the price elasticity estimates are decreasing in land quality, indicated by the higher coefficient on corn price for successive quartiles. This aligns with the prediction that counties with more heterogeneous and thus lower average land quality would be more flexible in adding corn acreage when corn prices rise (Table 4.1, quartile columns). The log-log specification of the empirical model allows the interpretation of the coefficients as percentage changes. So, when corn prices double (in real terms), the planted acreage of corn can be expected to increase by 52% in counties that are in the bottom quartile for land quality. However, the comparable response would be only 18% for the counties in the top quartile. Evaluated at the means, these translate to an

expansion of corn acreage by approximately 18,900 acres for the counties in the bottom quartile, compared to 24,700 acres for those in the top quartile. Thus, the magnitude of impacts are closer in terms of levels than in percentage terms. This is because, on average, the planted corn acreage in the bottom quartile counties is much lower than the top quartile (36,500 vs. 140,800 acres; see the last row in Table 4.1 for the mean planted acres of corn by land quality quartile).

Corn acreage generally responds negatively when the prices of alternative crops (except for soy) and CRP rental payments are higher (Table 4.1, full sample). This suggests wheat, hay and oats, and enrolling croplands into the CRP program are substitute land uses to corn cultivation in the WCB.

The coefficients on soy prices are positive for the full sample and the land quality quartiles (and significant except for the first quartile). Over the sample period, the dominant relationship of corn and soy with respect to land use appears to be one of complementarity. This is consistent with agronomic (Edwards et al 1988 and Crookston et al 1991) and field (Wallander 2013) evidence that corn-soy rotations improve yields and control pests, and continuous (at least three years in a row) cropping of either has been uncommon.

The generally expected relationship between prices of inputs, like fertilizer, and supply is negative. Higher input prices may cause farmers to switch away from corn to save on costs, especially in areas with low corn yields, to less fertilizer-dependent crops. Indeed, higher fertilizer prices have a statistically significant and negative impact on corn acreage across all quartiles.

Excessive precipitation during the planting season has a discouraging effect on corn planting in the WCB as suggested by the negative coefficient on average April precipitation for the full sample as well as the counties in quartiles I-III.

The 1996 dummy measures the impact of the passage of the FAIR Act in 1996, which eliminated the acreage set asides. In the mid 1980s, as many as 80 million acres of cropland were idled under this program nationwide (Glauber and Effland 2016). The model suggests, overall, the FAIR Act led to lower corn acreage in the WCB based on the negative and significant coefficients (for all but the 4th quartile counties). Initially, farmers welcomed the elimination of acreage controls and the planting flexibility afforded by the act, and corn acreage expanded nationwide. This expansion, however, led to lower crop prices and a subsequent adjustment in acreage and output.

The 2006 dummy for EPAct is positive and significant for the full sample as well as the land quality quartiles. Thus, the change in the US biofuel policy led to an increase in corn acreage beyond its influence on crop prices. Also, the coefficients are larger in magnitude for quartiles III and IV, suggesting this effect has been more pronounced for counties with lower land quality.

The coefficients of the linear trend component for all counties but those in quartile IV are positive and significant. Thus, even if prices and CRP payments had been constant, and there had been no policy changes, the corn acreage in the WCB would have risen over this period for counties in the first three quartiles. For the counties in quartile IV there is no discernible time trend to corn cultivation; a possible sign of opportunistic cultivation of corn in these counties. Lastly, the constant term confirms the baseline amount of corn

planted declines with land quality since the estimates are smaller for the third and fourth quartile.

Table 4.1

Price responsiveness of planted acres of corn, full sample and by land quality quartiles

Variable:	Full sample	Quartile I	Quartile II	Quartile III	Quartile IV
Lagged corn ("own") price	0.291 ***	0.176 ***	0.216 ***	0.289 ***	0.519 ***
2015 $/bu	*16.71*	*5.66*	*6.26*	*7.47*	*10.42*
Lagged soy price	0.144 ***	0.050	0.255 ***	0.247 ***	0.165 ***
2015 $/bu	*5.61*	*1.06*	*4.12*	*4.63*	*3.33*
Lagged hay price	-0.034 *	-0.117 ***	-0.072 *	0.023	0.088 *
2015 $/ton	*-1.69*	*-4.2*	*-1.96*	*0.62*	*1.93*
Lagged wheat/oat prices[i]	-0.126 ***	-0.010	-0.129 ***	-0.139 **	-0.365 ***
2015 $/bu	*-6.34*	*-0.46*	*-3.01*	*-2.57*	*-6.11*
Lagged county average CRP	-0.564 ***	-0.736 ***	-1.088 ***	-0.859 ***	-0.490 ***
rental payments, 2015 $/acre	*-8.77*	*-3.92*	*-5.29*	*-7.29*	*-4.55*
Fertilizer price index	-0.294 ***	-0.261 ***	-0.317 ***	-0.365 ***	-0.221 ***
	-10.86	*-9.03*	*-5.94*	*-6.75*	*-3.46*
April precipitation (mm)	-0.0007 ***	-0.0003 ***	-0.0012 ***	-0.0010 ***	-0.0003
	-9.64	*-4.03*	*-7.26*	*-6.27*	*-1.33*
Time trend	0.0147 ***	0.0162 ***	0.0210 ***	0.0162 ***	-0.0007
	13.78	*11.22*	*9.55*	*7.96*	*-0.24*
Dummy for FAIR Act (1996)	-0.0574 ***	-0.0675 ***	-0.1562 ***	-0.1488 ***	0.0735 *
	-4.13	*-3.45*	*-5.62*	*-4.98*	*1.83*
Dummy for EPAct (2006)	0.1150 ***	0.0780 ***	0.0872 ***	0.1394 ***	0.1210 ***
	10.25	*5.6*	*4.53*	*6.1*	*4.61*
Constant	14.09 ***	16.31 ***	16.98 ***	15.01 ***	11.77 ***
	47.84	*18.21*	*18.03*	*28.34*	*22.97*
No. of observations	10734	2841	2818	2750	2325
Mean acres of corn planted (1986-2015)	89,810	140,774	102,906	72,826	36,453

The full sample includes 396 counties in the WCB. Model estimated with county-year data 1986-2015. T-statistics are reported below coefficients and significance levels of coefficients are reported as (*) 10%, (**) 5%, and (***) 1% level. (i) For counties in Iowa, the third major alternative crop is oat and not wheat, therefore oat prices are used in lieu of wheat prices.

Next, I investigate whether the acreage response varies before and after the change in biofuel policy using a break point at 2006. The biofuel mandate introduced by EPAct

started at 4 billion gallons in 2006, which rose to nearly 17 billion gallons by 2015. Assuming 2.7 bushels of corn is needed to produce one gallon of ethanol (Rajagopal et al 2007), the 2015 mandate accounted for 46% of that year's US corn output. More than a third of US corn crop is now consistently used for ethanol (average is 36% for 2007–2015), compared to 8% in the decade prior (USDA ERS Bioenergy Statistics).

The rapid ascent in corn prices around 2006 can be attributed partly to this policy change (Roberts and Schlenker 2013). Prior to 2006, received corn prices by farmers in the WCB averaged $2.1/bushel, which jumped to $4.4/bushel 2006-2015. Also, over 65% of the regional expansion in corn acreage in the WCB occurred post-2006.

Splitting the sample at 2006 reveals that the full (30-year) sample results (Table 4.1) are driven by the price response post-2006 (Table 4.2, A vs. B). The price response is much higher in magnitude post-2016 (0.48 vs. 0.19) for the complete set of counties. Additionally, the inverse relationship between land quality and price response is not apparent in the pre-2006 period, with the top quartile emerging as most price responsive. Estimates of the post-2006 period, on the other hand, strongly exhibit the relationship we observed in the full sample estimates, where price responsiveness rises with diminishing land quality.

There are two plausible, non-mutually exclusive explanations: first, the use of lower quality lands for corn cropping responds non-linearly to price. In other words, prices need to be above a certain threshold for farmers to utilize the lower quality lands for corn cultivation. The second is associated with declining availability of higher quality lands in the WCB over time for corn cultivation.

With respect to other control variables, the sign on soy price flips from positive before 2006 to negative after 2006. This suggests corn and soy have become substitutes in land use in the biofuel era.

The influence of CRP payments on corn acreage also appears to have diminished pre/post 2006. CRP enrolments peaked in 2007 at 36.8 million acres, about 10 million of which were in the WCB, and have steadily declined since then. In 2007, many CRP contracts were not renewed nationwide. Nearly 1.1 million CRP acres returned to cropping in the WCB. Several hundred thousand acres exited the program early in 2008, despite facing payback of past rents plus penalties (Heimlich 2010). Since 2005, CRP rental payments have not been able to keep up with high prices and lure farmers away from cropping. While cropland rents have adjusted to reflect high crop prices, CRP rental payments have trailed behind (Fig. 4.1C).

Between 2004-2008, oil prices nearly tripled, resulting in higher fertilizer prices (Fig. 1C). Despite this, the impact of high fertilizer (and, generally, input) prices on corn acreage in the post-2006 period has been more muted than the pre-2006 period (for the full sample, -0.31 vs. -0.23). It is possible that after 2006, rather than switching to alternative crops, some corn farmers have chosen to substitute land for inputs. This is a more meaningful strategy in the higher land quality counties, and those in quartiles I and II have either lower or not significant coefficient estimates for fertilizer price index post-2006. Strikingly, for the bottom quartile of counties (lowest quality), the sensitivity to input prices is substantially higher post-2006 than pre-2006. This may be due to the low

productivity of these lands, and high fertilizer use may be necessary to extract satisfactory yields (Cassman 1999).

Overall, the results indicate a different pattern of farmer choices with respect to corn cultivation based on land quality. Based on a joint Wald test of coefficients (after estimating pooled model of all quartiles) I reject the hypothesis that the counties in different quartiles respond to corn price changes identically (F value significant at $p<0.01$).

Clustering the standard errors by agricultural statistical districts (USDA groupings of 8-9 counties where similar crops are grown) to address potential spatial correlation in the dependent variable results in lower t-statistics and wider confidence intervals for some of the estimated coefficients. However, the price elasticity estimates remain robust (at $p<0.01$) and my conclusions do not change. Similarly, the results and their interpretation do not change materially if the NICC metric is derived from higher resolution (10m) gridded SSURGO data instead of the 90m (see Supplemental Information, section 4.6).

Table 4.2

Price responsiveness of planted acres of corn, pre-/post-biofuels period, full sample and by land quality quartiles

	(A) 1986 - 2005					(B) 2006 - 2015				
Variable:	**Full sample**	**Quartile I**	**Quartile II**	**Quartile III**	**Quartile IV**	**Full sample**	**Quartile I**	**Quartile II**	**Quartile III**	**Quartile IV**
Lagged corn ("own") price	0.191 ***	0.272 ***	0.157 ***	0.184 ***	0.223 ***	0.480 ***	0.268 ***	0.471 ***	0.577 ***	0.775 ***
2015 $/bu	*8.48*	*9.06*	*3.62*	*3.61*	*3.11*	*13.98*	*5.32*	*7.7*	*9.61*	*6.91*
Lagged soy price	0.161 ***	0.037	0.208 ***	0.263 ***	0.145 ***	-0.359 ***	-0.369 ***	-0.423 ***	-0.442 ***	-0.212 **
2015 $/bu	*7.19*	*0.91*	*4.74*	*5.3*	*2.82*	*-10.26*	*-7.55*	*-5.09*	*-8.73*	*-2.21*
Lagged hay price	0.018	-0.088 ***	-0.039	0.079 *	0.136 ***	-0.001	0.050	-0.008	-0.005	-0.150 *
2015 $/ton	*0.84*	*-3.36*	*-0.97*	*1.81*	*2.84*	*-0.05*	*1.42*	*-0.21*	*-0.12*	*-1.92*
Lagged wheat/oat prices[i]	-0.171 ***	-0.018	-0.170 ***	-0.224 ***	-0.341 ***	-0.010	-0.006	0.029	0.033	0.043
2015 $/bu	*-6.66*	*-0.71*	*-3.28*	*-3.22*	*-4.62*	*-0.42*	*-0.2*	*0.72*	*0.75*	*0.45*
Lagged county average CRP	-0.314 ***	-0.314 ***	-0.725 ***	-0.735 ***	-0.331 ***	-0.174 **	-0.157	-0.280	-0.288 ***	0.001
rental payments, 2015 $/acre	*-5.14*	*-3.09*	*-3.71*	*-6.67*	*-2.89*	*-2.55*	*-1.14*	*-1.37*	*-2.78*	*0.01*
Fertilizer price index	-0.307 ***	-0.292 ***	-0.296 ***	-0.347 ***	-0.268 ***	-0.234 ***	0.076	-0.230 *	-0.363 ***	-0.850 ***
	-9.31	*-9.01*	*-5.03*	*-4.85*	*-3.31*	*-3.33*	*0.71*	*-1.67*	*-3.17*	*-3.55*
April precipitation (mm)	-0.0005 ***	-0.0001	-0.0008 ***	-0.0009 ***	-0.0003	-0.0003 ***	-0.0003 ***	-0.0003 **	-0.0004 **	0.0002
	-6.08	*-0.9*	*-4.31*	*-4.97*	*-1.33*	*-3.1*	*-3.47*	*-2.44*	*-1.99*	*0.61*
Time trend	0.0113 ***	0.0191 ***	0.0119 ***	0.0059 *	-0.0114 **	0.0279 ***	0.0091 **	0.0332 ***	0.0412 ***	0.0432 ***
	6.7	*14.65*	*4.49*	*1.74*	*-2.41*	*8.74*	*2.48*	*3.29*	*7.27*	*5.96*
Dummy for FAIR Act (1996)	0.0055	-0.0386 ***	-0.0418 ***	-0.0491 **	0.1546 ***	.	.	.	.	.
	0.41	*-3.02*	*-2.65*	*-2.15*	*3.32*					
Dummy for EPAct (2006)		.	.	.	.	.	.	.	.	.
Constant	12.87 ***	14.07 ***	15.36 ***	14.44 ***	11.48 ***	12.41 ***	12.27 ***	13.13 ***	12.88 ***	12.75 ***
	42.2	*24.01*	*16.58*	*23.46*	*19.73*	*29.48*	*13.51*	*12.72*	*20.16*	*11.99*
No. of observations	7230	1863	1866	1845	1656	3504	978	952	905	669

The full sample includes 396 counties in the WCB. Sub-model (A) estimated with county-year data over 1986-2005 and (B) over 2006-2015. T-statistics are reported below coefficients and significance levels of coefficients are reported as (*) 10%, (**) 5%, and (***) 1% level. The 1996 dummy is omitted from (B) because it falls outside of the period covered in the sub-model, and the 2006 dummy is omitted from both (A) and (B) since the sample is divided at 2006. (i) For counties in Iowa, the third major alternative crop is oat and not wheat, and therefore oat prices are used in lieu of wheat prices.

4.3.3 Implications for marginal lands and grasslands in the WCB

About 41% of the WCB are marginal agricultural lands suitable for bioenergy crop cultivation based on assessment of Cai et al (2011), conducted globally at 1km scale. Higher resolution data from the NICC and CDL can be used to refine Cai et al (2011) estimates for marginal lands for bioenergy crops in the WCB. Marginal land quality in the WCB is on average worse than the non-marginal portions based on NICC (mean scores of 3.8 vs. 3.6, with the two groups statistically different at 1%). Using data from the CDL, it is observed that nearly a quarter of these marginal lands have already been planted to corn at least one year between 2010-2016, concentrated in portions with lower NICC scores (mean 1.87, Fig. 4.4). Also, on a county level, half of the counties in the WCB had 15% or more of their cropland described as marginal.

Based on this evidence, the highest quality marginal lands (NICC score ≤ 2) can be excluded from an assessment of available marginal lands, as these are likely to be used for conventional cropping. This shrinks available marginal land area in the WCB by almost 36%. Further excluding the lowest quality lands (NICC score ≥ 7) reduces the amount of available land for bioenergy crops by another 11% (Fig. 4.4). The remainder constitutes roughly one-fifth of the entire land area of WCB (about 48 million acres), still a sizeable land base for bioenergy crop cultivation.

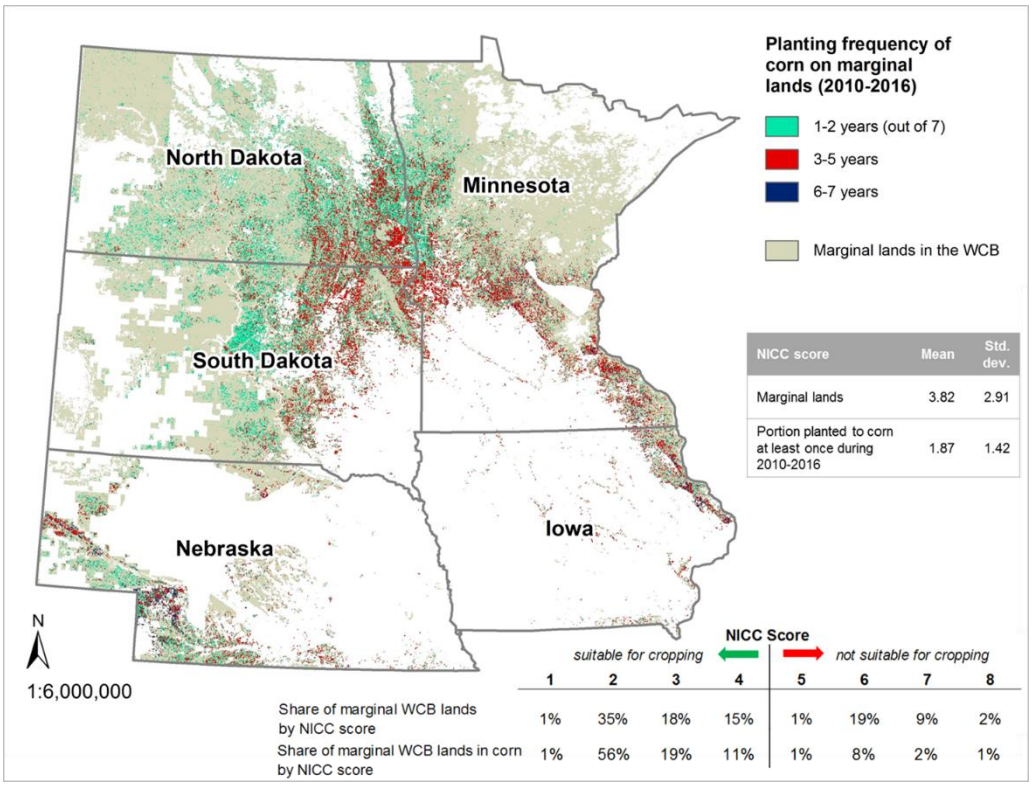


Figure 4.4. Land quality and corn cultivation over marginal lands. Extent of marginal agricultural lands in the WCB (per Cai et al 2011) in the WCB at 1km resolution); portion planted in corn at any given year during 2010-2016; and the observed overall land quality and for the portion planted to corn.

Nonetheless, bioenergy crops might need to compete with corn even on this subset of lands in the event of another price spike. For example, between 2009-2012, corn prices increased again three years in a row (by 46%, 16%, and 9%), going from $3.8 to nearly $7/bushel (in real terms), reflecting the effects what became one of the worst droughts in the US history (Rippey 2015).

The analysis presented in this manuscript predicts this price increase would lead to a larger percentage expansion in planted corn acres for the WCB counties with lower land quality versus those with higher land quality. On an aggregate basis, total corn planted

area expanded in the WCB by 5.52 million acres from 2010 to 2012, or by 15%. About 40% of this expansion, or 2.24 million acres, occurred in counties that fall within our refined marginal lands criteria, representing a growth rate of 29% The comparable growth rate for the rest of WCB counties over the same time frame was 6.6%. (Note acreage is responding to prices with a year's delay). Thus, not only available marginal land area for bioenergy crops is smaller than estimated when refined using higher-resolution land use and land quality data, but this land is also more likely to be planted to corn during high crop price periods.

My results also speak to earlier work concerning the transition of grasslands to croplands in the WCB (Wright and Wimberly 2013; Claassen et al 2011). Based on recent satellite data, the grass/pasture land use category has the poorest land quality in the WCB (Fig. 4.3A). Also, the counties with the highest average NICC scores have a larger proportion of land use in grass/pasture. During 2010-2016, the proportion of grass/pasture in the fourth land quality quartile of counties averaged 58%, compared to 10% for those in the first land quality quartile. Based on the econometric predictions presented here, high prices will cause a larger relative expansion of corn cultivation in the counties with the largest grass/pasture land use.

Marginal agricultural lands and grass/pasture areas to some extent overlap in the WCB, but not all marginal lands are grass/pasture (and vice versa). However, during 2010-2012, the areas where most low quality marginal lands transitioned to corn also appear to be those that registered largest net losses of grass/pasture. Over one million acres of grass/pasture in 2010 appeared in corn cultivation in 2012. These hotspots for marginal

land and grass/pasture losses also largely overlap with the fragile PPR wetlands (Fig. 4.5). The loss of grasslands to corn documented in the immediate aftermath of the change in US biofuel policy (Wright and Wimberly 2013) has apparently resumed with the next price spike. Since most of the CO_2 emissions from previously uncultivated grassland conversion to cropping are generated within the first 2-3 years (Grandy and Robertson, 2006), it is preferable to target bioenergy crop cultivation on these low quality marginal lands that have already been cropped. However, as this analysis demonstrates, these are precisely the areas farmers would choose to revert to cropping when economic conditions are favorable.

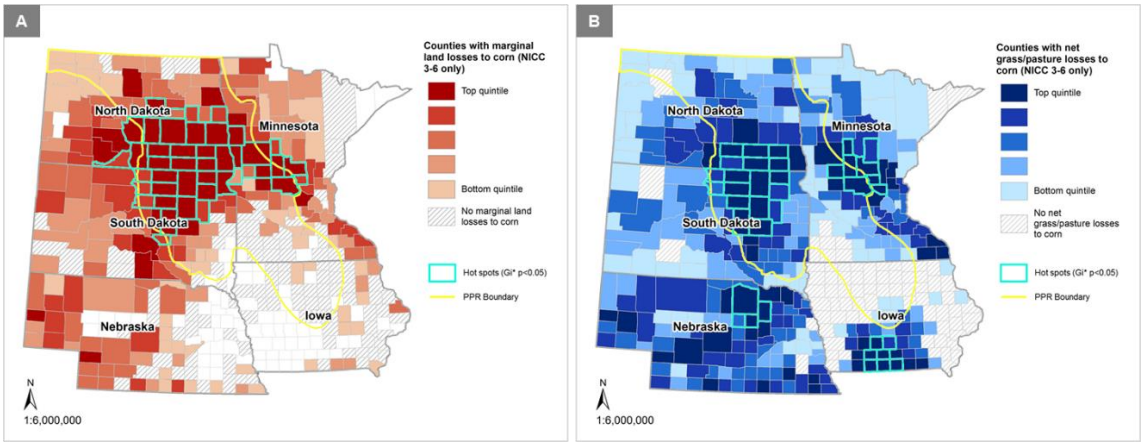


Figure 4.5. Marginal land and grass/pasture losses to corn in the WCB, 2010-2012. (A) Quintiles of counties where marginal lands with NICC scores ranging from 3-6 were planted to corn during 2010-2012. (B) Quintiles of counties where a net change from grass/pasture lands (NICC scores 3-6) to corn is observed. In both (A) and (B) top quintile consists of counties with the highest change, with green boundaries indicate hot spot counties for marginal or grass/pasture land use for corn based on Gi* statistic with 95% confidence. The yellow boundary marks the portion of the Prairie Pothole Region (PPR) within the WCB.

4.3.4 Data limitations and uncertainty

The work presented here has certain unavoidable limitations due to data availability. To utilize the three decades of yearly corn acreage data, the spatial scale of the econometric analysis is fixed at the county level. Hence, the NICC scores used to represent land quality are aggregated up from 90m resolution to county average scores making the land quality metric less precise (mean county land area exceeds 0.5 million acres in the WCB). Using county-average land quality scores obscures precisely on what type of land within a county corn acreage would expand in response to high prices. This requires longitudinal land use data at the farm (or field) level. Also, the CDL data does not allow for identification of previously uncultivated grasslands versus managed range/pasture (Kline et al 2013). Therefore, there is uncertainty about the land quality of and the applicability of results to native grasslands.

The efficiency of the econometric estimation could be improved with a simultaneous equations approach where the acreage response for corn and the alternative crops/land uses would be estimated jointly. However, data coverage is substantially incomplete for the alternative crops, except for soy. Limiting the analysis to the subset of county-years with complete data could bias the representation of the variable land quality observed in the WCB and cause a loss of statistical power.

4.4 Conclusions

Land quality influences acreage response to prices through land availability. In the WCB, corn cultivation expanded by nearly seven million acres over 1986-2015. Most corn is

still grown in counties with the highest quality lands that are physically well-suited to corn cropping. However, because there is a limited supply of such lands, when corn prices rise, there is a relatively small margin of land not already used for corn that can be shifted into corn production. On the other hand, in counties with a lower and more heterogeneous mix of land quality, there could be a significant margin of land not actively used to grow corn. In these counties, I find the proportional expansion in corn acreage in response to high crop prices to be more pronounced.

As corn cultivation creeps into lower-quality marginal agricultural lands, available land for bioenergy crops is further constrained. If lower quality lands are also more environmentally vulnerable, like the remnants of native prairie in the WCB, then their large sensitivity to corn prices is a concern, for this indicates such lands can easily cycle in and out of corn cultivation. Furthermore, any loss of native prairie is noteworthy because it can trigger indirect emissions of greenhouse gasses (Searchinger et al 2008), and lead to habitat loss for native species (Mushet et al 2014) and loss of biodiversity (Werling et al 2014). When lower quality, environmentally sensitive lands are utilized for corn cultivation, there are further environmental concerns involving the heavy use of nitrogen fertilizers and attendant issues of runoff and leaching, as well as groundwater depletion in parts of the WCB overlying the Ogallala aquifer (Dennehy et al 2002; Donner and Kucharik 2008; Langpap and Wu 2011; and Keeler and Polasky 2014). Policies that may alter cropland use by creating financial incentives tied to the farming of specific crops, such as the US biofuel policy, needs to consider the policy-price feedback and higher price responsiveness of low quality lands. Although the higher price elasticity

estimates signal farmer flexibility, suggesting land transitions to corn need not be irreversible, discouraging opportunistic farming on the most environmentally vulnerable areas might reduce stress on ecosystems.

4.5 Acknowledgements

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4.7 Supplemental information

Sensitivity of results to using 10-m gridded SSURGO data for land quality:

The analysis in the main text uses gridded SSURGO (gSSURGO) data at 90m resolution. The gSSURGO product exists at two finer spatial scales: 10m and 30m. However, the land quality and econometric analysis and results presented in the main text are not sensitive to using gridded data at scales finer than the 90m. I demonstrate this by deriving summary statistics using the 10m data as well as repeating the analysis in Table 4.1 to show there are no material differences in results. By extension, this discussion applies to the use of 30m data.

The 10m, 30m, and 90m gSSURGO are each generated independently from the vector data of soils through a vector-to-raster conversion. As the target raster resolution becomes coarser, the underlying vector data omits or eliminates polygons that are inconsequential at the target spatial resolution.

First, in the WCB, the number of eliminated polygons is not large by number or area when going from 10m to 90m. In the 2016 data, the 10m gridded product is based on 47,547 underlying map units (unique groups of polygons) for the WCB. The 90m gridded product is based on 47,249 map units. On an aggregate (WCB) level, the difference is 298 fewer map units in the 90m gridded product than the 10m gridded product, and the total subsumed area is 440 acres. This is not statistically different from zero compared to 228.6 million acres of total WCB land area. Generally, the underlying polygons eliminated when going from the 10m to the 90m data represent man-made features. Therefore, the omission of such polygons in the 90m data is not consequential in the

analysis of land quality for agricultural land using a summary metric like the NICC, particularly if such analysis is done at the county-scale.

Second, any differences in the NICC values in the 10m versus the 90m gridded data does not lead to material changes in the results. For example, the mean NICC scores reported for the entire WCB (Fig. 4.2A) and the corn cultivated portions (Fig. 4.2B) vary only after the third decimal place when calculated at the 10m resolution. Similarly, any difference in average NICC scores calculated for different land use categories based on the 2010-2016 CDL data (Fig. 4.3A) also occurs at the third decimal place when using the 10m instead of the 90m data.

Third, any differences across the county average NICC scores, which determine the "land quality quartiles" in the econometric analysis are statistically insignificant between the 10m and the 90m data (based on a two-sample Wilcoxon ranksum test at $p<0.01$). Below, Table 4.3 shows the average NICC score for all counties and the quartile breakpoints up to five decimal places, and Figure 4.6 shows the distribution of county average NICC scores at the 10m and 90m resolution. Both illustrate how close the NICC scores are at the county level using the two different datasets. Basically, the county averages using the 10m and 90m data are equal if rounded to two decimal places.

Table 4.3

Comparison of county average NICC values using 90m vs. 10m gSSURGO data for the WCB

County-level NICC values (N=396)	90m resolution	10m resolution
Mean	3.33679	3.33553
25th percentile (Breakpoint between 1st and 2nd quartiles)	2.64641	2.64792
50th percentile (Breakpoint between 2nd and 3rd quartiles)	3.05030	3.05317
75th percentile (Breakpoint between 3rd and 4th quartiles)	3.97310	3.96840

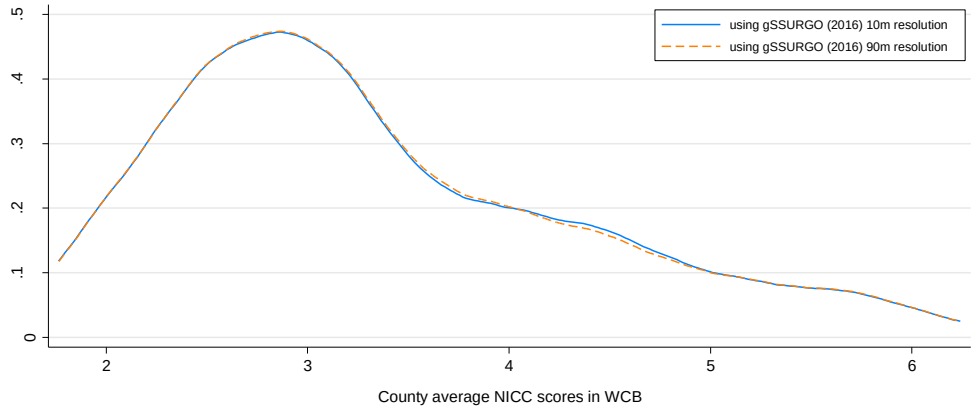


Figure 4.6. Land quality indicator with different resolutions of soil data. Kernel density plots of county average NICC scores in the WCB using the 90m versus the 10m gridded SSURGO data.

It follows that the econometric results that find the price elasticity of corn acreage for counties grouped by land quality (in quartiles) also do not vary materially. Indeed, I find that if the quartiles are constructed using county average NICC scores that are rounded to two decimal places, the results are identical. If I use the NICC scores without any rounding ("high precision"), the coefficient estimates for the 2nd, 3rd and 4th quartiles

change slightly but not materially, and the conclusions remain the same. These are reported in Table 4.4 where I repeat the analysis in Table 4.1 with the 10m data.

Table 4.4

Price responsiveness of planted acres of corn, full sample and by land quality quartiles using 10m data

Reproduction of Table 1 using high-precision NICC values from 10m gSSURGO

Variable:	Full sample	Quartile I	Quartile II	Quartile III	Quartile IV
Lagged corn ("own") price	0.291 ***	0.176 ***	0.213 ***	0.287 ***	0.537 ***
2015 $/bu	*16.71*	*5.66*	*6.12*	*7.65*	*10.90*
Lagged soy price	0.144 ***	0.050	0.275 ***	0.220 ***	0.174 ***
2015 $/bu	*5.61*	*1.06*	*4.46*	*4.23*	*3.44*
Lagged hay price	-0.034 *	-0.117 ***	-0.072 *	0.019	0.087 *
2015 $/ton	*-1.69*	*-4.20*	*-1.96*	*0.53*	*1.88*
Lagged wheat/oat prices[i]	-0.126 ***	-0.010	-0.136 ***	-0.122 **	-0.388 ***
2015 $/bu	*-6.34*	*-0.46*	*-3.08*	*-2.39*	*-6.39*
Lagged county average CRP	-0.564 ***	-0.736 ***	-1.140 ***	-0.794 ***	-0.516 ***
rental payments, 2015 $/acre	*-8.77*	*-3.92*	*-5.54*	*-6.98*	*-4.61*
Fertilizer price index	-0.294 ***	-0.261 ***	-0.327 ***	-0.350 ***	-0.223 ***
	-10.86	*-9.03*	*-6.17*	*-6.64*	*-3.43*
April precipitation (mm)	-0.0007 ***	-0.0003 ***	-0.0012 ***	-0.0010 ***	-0.0003
	-9.64	*-4.03*	*-7.28*	*-6.20*	*-1.26*
Time trend	0.0147 ***	0.0162 ***	0.0211 ***	-0.0159 ***	-0.0008
	13.78	*11.22*	*9.85*	*8.05*	*-0.30*
Dummy for FAIR Act (1996)	-0.0574 ***	-0.0675 ***	-0.1590 ***	-0.1410 ***	0.0703 *
	-4.13	*-3.45*	*-5.97*	*-4.72*	*1.69*
Dummy for EPAct (2006)	0.1150 ***	0.0780 ***	0.0943 ***	0.1300 ***	0.1250 ***
	10.25	*5.6*	*4.69*	*6.00*	*4.69*
Constant	14.09 ***	16.31 ***	17.20 ***	14.78 ***	11.83 ***
	47.84	*18.21*	*18.42*	*29.09*	*22.02*
No. of observations	10734	2841	2817	2782	2294

Underlying land quality quartile breakpoints for grouping counties calculated using high-precision (not rounded) county-average NICC values from the 10m gSSURGO dataset. The full sample includes 396 counties in the WCB. Model estimated with county-year data 1986 to 2015. T-statistics are reported below coefficients and significance levels of coefficients are reported as (*) 10%, (**) 5%, and (***) 1% level. (i) For counties in Iowa, the third major alternative crop is oat and not wheat, therefore oat prices are used in lieu of wheat prices.

The minor differences are due to six (out of the 396 in my sample) counties moving into either the next upper or the next lower quartile when the quartile breakpoints are constructed with high-precision NICC values from the 10m data. (Observe the sample sizes by quartile in Table 4.4; note that the sample size is county*year). The results and their interpretation do not change materially (neither in economic magnitude nor in terms of statistical significance). In sum, the land quality analysis using the NICC metric from either the 10m or 90m gridded SSURGO data does not result in material changes to my results or conclusions, but the 90m data has space- and memory-saving advantages that aid in computation and ease of replicability.

CHAPTER 5

CONCLUSIONS

This dissertation evaluates sustainability contributions and implications of deploying underutilized land resources for alternative uses and at multiple scales: urban, continental, and regional. These include spaces in urban areas like vacant lots and building surfaces (rooftops and façades). Outside of urban areas, they include marginal (agricultural) lands. The main premise of the dissertation is that while these land resources are currently underutilized for a variety of physical and economic causes, it is still important to assess the scale of the existing resource and evaluate alternative uses that are potentially more sustainable (than current use).

The first paper considers the use of underutilized spaces at the urban scale—vacant lots, rooftops and building façades for urban agriculture (UA). Fine-scale data on these urban surfaces are used to assess available area, which are then combined with local inputs to determine the contribution of UA towards meeting the desired outcomes associated with local sustainability goals for Phoenix, Arizona. These sustainability goals concern local food supply, buildings and energy use, and green open spaces. The analysis in Paper 1 shows that UA is very successful in meeting the desired outcomes for local food supply, and can meet 90% of a typical adult's annual consumption of fresh fruits and vegetables. UA can also provide fresh produce in all known food deserts in Phoenix. An important co-benefit of UA quantified in this paper involves the expansion of green open spaces available to public, which reduces the number of underserved areas (block groups) by existing public parks from 713 to 260, and increases total green open space from 12.2%

to over 14% of the land area in Phoenix. This is a notable increase considering only 8% of existing park area is made up of community parks in the City of Phoenix, and the majority (91%) are large regional parks that are beyond the reach of most city residents on a day-to-day basis.

A third co-benefit involves savings associated with buildings and energy use. Even though these appear small in aggregate (baseline estimates >125 GWh per year and >50,000 tons of CO_2), they are on-par with building-scale savings from similar interventions elsewhere (about 3% per year per building). These savings are also potentially underestimated because of the conservative assumptions used in analysis (e.g., the subset of buildings selected are only 35% of available flat rooftops).

As a sustainability solution, UA can also improve and encourage multi-purpose utilization of existing capital stock (infrastructure, buildings) in cities. Also, further co-benefits from UA exist for Phoenix that were not quantified in this analysis. For example, converting currently unvegetated vacant lots into UA applications would help with runoff regulation and reduce local flash flooding during monsoon season. This would also help reduce fugitive dust emissions--a source of particulate matter pollution in Phoenix--as these were less directly connected with the City's sustainability goals. In other words, use of underutilized spaces for activities like UA can provide a multitude of direct and indirect benefits, and when such benefits line up well with policy goals they may be easier to execute. Overall, findings in this paper underscore the value of a data-driven, city-scale assessment of UA tied explicitly to locally-defined goals for sustainability.

Performing similar assessments for other cities would clarify under what conditions UA can be a sustainability solution for cities.

Paper 2 then zooms out to the continental U.S. scale (CONUS) to investigate scenarios of sustainable use of marginal lands for the cultivation of perennial bioenergy crops (miscanthus and switchgrass) from an economic standpoint. The harvested biomass from these crops is converted into cellulosic biofuels, whose supply is benchmarked against the future (2040) liquid fuels demand in the U.S. aviation sector that cannot be readily electrified. This paper constitutes the third and final phase of a larger project, which couples hydroclimatic (Phase I), ecosystem (Phase II) and economic (Phase III) models. Using a spatially-explicit economic optimization model, Geospatial Bioenergy System Model (GBSM), the analysis in this paper determines the optimal industry configuration that ensures a profitable cellulosic biofuels industry in the United States, using biomass sourced from environmentally sustainable marginal lands. Essentially, the model calculates total cellulosic biofuel quantity produce (in million gallons of gasoline equivalent per year, or mggey) given a wholesale fuel price level (in $/gge) that maximizes profits for the cellulosic biofuel industry.

Under different land use scenarios, the paper demonstrates a wide range of profitable cellulosic biofuel supply outcomes are achievable. The low-end of this range is <1,000 mggey produced using switchgrass as feedstock given a price level of $3.4/gge. The high-end estimate is close to 100,000 mggey produced using miscanthus as feedstock given a price level of $5/gge.

Nonetheless, it appears that there are diminishing output benefits observed at prices higher than $4/gge. As prices exceed $4/gge, biomass is supplied from increasingly lower productivity marginal lands, at lower yields. As a result, the unit increase in land base returns a less than proportional increase in biofuel quantity.

Using $4/gge as a benchmark price, the biofuel output range is narrowed to 30.8 to 90.7 billion gge/year. This is sufficient to meet the future non-electrifiable aviation fuel demand in the U.S. (projected to be about 30 to 33 billion ggey in 2040). From a market perspective, $4/gge is about $0.8/gge higher than the historical maximum wholesale fuel price in the U.S. A carbon price of $77/ton CO_2eq would close this gap; a carbon price >$140/ton CO_2eq would make cellulosic biofuels competitive with gasoline-based liquid fuels today, assuming cellulosic biofuels are net carbon sinks.

The underlying marginal land use at $4/gge ranges from 22.3 to 79.4 million hectares under different scenarios, with existing cropland displacement declining as the land base grows. (Ranging from 52% to 34% of total underlying marginal land use). The higher the proportion of existing croplands displaced, the higher are average biomass yields, which necessitates using less land.

Therefore, some displacement of (marginal and less productive) croplands for bioenergy crop cultivation is not entirely avoidable in the U.S. setting, provided that meeting the targeted liquid fuels demand and ensuring sustainability of water resources are prioritized. This is because displacing croplands (as opposed to range and shrubland) reduces stress on surface and ground water resources by preserving valuable soil

moisture. Additionally, any cropland displaced would mainly offset those devoted to the supply of conventional biofuels, potentially with limited impact on food supply. Miscanthus emerges as the feedstock of choice under all marginal land use scenarios and at different price levels considered. This is because switchgrass has lower yields than miscanthus, and it becomes necessary to use a much larger land base to obtain the same amount of output. This yield difference is so substantial that it cannot be overcome by the lower costs of switchgrass cultivation compared to miscanthus.

The results also quantify a methodological innovation due to the coupled modelling framework. In this framework, hydroclimatic impacts arising from landscape modification by bioenergy crops are explicitly incorporated into simulated yields. We estimate an average of 3.5% positive hydroclimatic feedback on modeled biomass yields, which reduces the amount of land base needed. Last but not least, the paper's contribution to the literature comes not only from using a coupled environmental-economic modeling approach, but doing so at a fine (10 km) spatial scale that captures the variation in biomass supply and costs more realistically. In sum, the U.S. has sufficient land resources to profitably produce sustainable biofuels. Based on the constraints imposed–using available marginal lands, minimizing impact on water resources–and the demand target of 30-33 billion gallons of liquid fuel per year, a cellulosic biofuel industry will likely take shape around the margins of corn/soy belt in the Eastern U.S. There is increasing interest in reducing emissions from aviation and for cleaner aviation fuels, yet carbon pricing (or comparable regulatory intervention) remains critical to for cellulosic biofuels to compete with conventional liquid fuels.

The third paper differs from the first two papers because it takes a historical perspective and looks at actual land use. In this paper, the role of land quality in the expanding use of marginal lands for corn cropping is investigated. The paper uses the Western Corn Belt (WCB) as a laboratory, a commercially important and environmentally sensitive region of the U.S. It is generally hypothesized that marginal agricultural lands have low productivity for (food) cropping purposes, and therefore remain available for alternative uses, such as cultivation of perennial grasses like miscanthus and switchgrass that are feedstocks for cellulosic biofuel. Because of their low productivity (associated with low land quality), such lands would not compete with prime agricultural lands for growing cash crops like corn.

Paper 3 upends this notion, and provides an economic basis supporting recent evidence that indicate cropping has been expanding into marginal lands in the WCB. The main finding of this paper is that under sufficiently high prices, the rate of corn acreage expansion is has been disproportionately higher in WCB counties with a lower and more heterogeneous mix of land quality. These counties that also have large proportions of marginal agricultural lands. Thus, high corn prices reduce available marginal lands for bioenergy crops. An expectation of high prices would also dissuade farmers from considering bioenergy crops that are perennial grasses, like switchgrass and miscanthus, as an alternative crop on these lands because of the multi-year commitment required for their cultivation. In addition, if lower quality lands are also more environmentally vulnerable, then their large sensitivity to corn prices is a concern, for this indicates such lands can easily cycle in and out of corn cultivation.

In sum, the research in this dissertation demonstrates how underutilized land resources can be used sustainably at continental scale towards achieving decarbonization in the transportation sector (Paper 2) or to meet local sustainability goals via urban agriculture deployment at the city-scale (Paper 1). The longer perspective offered by Paper 3 underscores the need to understand potential price-policy feedbacks, like the one generated by the U.S. biofuel policy, and and the role market forces play in supporting or impeding the sustainable uses of underutilized land resources. Ultimately, the larger themes emerge from this work are as follows:

(i) The use of data driven frameworks and integrated modeling approaches are important to account for co-benefits and feedbacks that provide a more complete assessment of sustainable uses of underutilized spaces and that improve their relative economics.

(ii) The contribution of underutilized spaces towards sustainability goals at small and large scales can be complementary. Urban agriculture can reduce pressures on croplands and complement non-urban food production and this complementarity can help diversify agricultural land use for meeting other goals, like supplying biofuels.

(iii) Unintended land use and environmental consequences associated with land use generally results from an incomplete understanding of market forces and economic linkages. It is important to incorporate social and economic considerations into the study of sustainable land use—and in general, the study of sustainability—to better anticipate such consequences.

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