AGRICULTURAL LANDUSE CHANGE IMPACTS ON BIOENERGY PROUDCTION, AVIFAUNA, AND WATER USE IN NEBRASKA'S RAINWATER BASIN

by

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Agriculture is an economically important form of landuse in the North American Great Plains. Since 19th Century European settlement, conversion of grasslands to rowcrops has increased food and bioenergy production, but has decreased wildlife habitat. Future agricultural landuse changes may be driven by alternative energy demands and regional climatic changes. Landuse change and its drivers could affect bioenergy production, wildlife populations and natural resources, and considering the potential impacts of impending changes in advance could assist with preparations for an uncertain future.

This study addressed how the conversion of marginally productive agricultural lands in the Rainwater Basin region of south–central Nebraska, U.S.A. to bioenergy switchgrass (*Panicum virgatum*) might impact ethanol production, grassland bird populations and agricultural groundwater withdrawals. This study also used multi–model inference to develop predictive models explaining annual variation in springtime wetland occurrence and flooded area in the Rainwater Basin.

Results suggest that producing adequate biomass for year round cellulosic ethanol production from switchgrass and residual maize (*Zea mays*) stover within existing starch–based ethanol plant service areas is feasible at current feedstock yields, removal rates and

bioconversion efficiencies. Throughout the Rainwater Basin, the replacement of marginally productive rowcrop fields with switchgrass could increase ethanol production, conserve groundwater and benefit grassland birds under novel future climatic conditions. However, converting Conservation Reserve Program (CRP) grassland to switchgrass could be detrimental to grassland bird populations. Predictive wetland inundation models suggest that springtime wetland inundation in the Rainwater Basin is a complex process driven by individual wetland characteristics, surrounding landuse and local weather events. The impacts of future climatic and landuse changes in the Rainwater Basin and surrounding Great Plains is ultimately likely to depend on which forms of alternative landuse are adopted and on how intensely change occurs.

DEDICATION

To my wife Annika, for the true and continual blessing you are.

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Chapter 1: INTRODUCTION

The Great Plains is an expansive North American ecoregion, stretching from the Rocky Mountain foothills in the west to deciduous forest in the east, and from Canada in the north, southward into Texas and Mexico (Samson & Knopf 1994; Ricketts et al. 1999). Historic Great Plains landcover consisted primarily of varying types of short, tall and mixed grass prairie, with perennial grasses constituting the dominant species (Weaver 1968; Barker & Whitman 1988; Hart & Hart 1997). Several major wetland complexes, composed of shallow rain–fed wetlands, also dotted the landscape along its north – south extent (Barker & Whitman 1988; Bolen et al. 1989).

Since settlement, agricultural practices have converted grasslands and wetlands into cropland for the production of maize (*Zea mays*), soybeans (*Glycine max*), wheat (*Triticum aestivum*), milo (*Sorghum bicolor*), alfalfa (*Medicago sativa*) and cotton (*Gossypium spp.*) (Musick et al. 1990; Guru & Horne 2000; Mitchell et al. 2010). As global food and bioenergy demands continue to rise, agricultural conversion and intensification continue (Tilman et al. 2002; Fargione et al. 2008). According to current estimates, approximately 99% of the Northern Great Plains is either farmed or grazed by livestock (Forrest et al. 2004). Mixedgrass prairie is an endangered ecosystem type and tallgrass prairie is critically endangered (Ricketts et al. 1999; Samson et al. 2004). Historic wetland area has decreased significantly in some areas and remaining wetlands are degraded by agricultural practices (Gersib 1989; Gibbs 2000; Higgins et al. 2002; LaGrange 2005).

Reductions in grassland and wetland area have decreased habitat availability for grassland birds, waterfowl and shorebirds (Samson & Knopf 1994; Higgins et al. 2002;

Brennan & Kuvlesky 2005). Grassland birds have experienced greater, faster and more widespread declines than any other ecological guild on the North American continent (Herkert et al. 1993; Knopf 1994; Sauer et al. 2011). Migratory waterfowl and shorebirds have also exhibited negative responses to reductions in grassland and wetland area (Higgins et al. 2002). Today, remnant and restored grasslands provided feeding and breeding habitat for grassland birds (Knopf 1994; Sauer et al. 2011), and migratory waterfowl and shorebird populations utilize remnant and restored wetlands for stopover habitat (Moore et al. 2005; Bishop & Vrtiska 2008).

Global climate models predict that the Great Plains region will continue to experience climatic changes throughout the 21st Century (IPCC 2007; Karl et al. 2009). Climate change in the Great Plains is expected to be characterized by maximum and minimum temperature increases, winter precipitation increases, summer precipitation decreases and greater frequencies of major storm occurrence (IPCC 2007; Karl et al. 2009). Changes could directly and indirectly impact societies, economies, and the environment (Root et al. 2003; Karl et al. 2009), and could place additional stresses on species already negatively influenced by anthropogenic landuse change (Moeller et al. 2008; Fontaine et al. 2009).

Demand for clean, renewable energy has resulted in recent, large scale efforts to utilize maize (*Zea mays*) grain for ethanol production (Schnoor et al. 2008). Despite extensive development of the starch-based ethanol industry, ethanol production from maize grain remains controversial, due to uncertainties over its net energy production, water use efficiency, ability to reduce atmospheric greenhouse gas emissions and concentrations, and competition with food production for landuse (Berndes 2008;

Searchinger et al. 2008; Tilman et al. 2009; Dale et al. 2010). Meanwhile, the benefits of second generation biofuels are increasingly promoted (Tilman et al. 2009). One alternative biofuel feedstock proposed for large scale cellulosic ethanol production in Great Plains agricultural landscapes is bioenergy switchgrass (*Panicum virgatum*) (Mitchell et al. 2012).

Switchgrass is a warm–season, perennial, C4 grass species, native to the Great Plains (Vogel 2004; Kaul et al. 2006). Switchgrass has been the subject of long-term agronomic research (Schmer et al. 2008; Mitchell et al. 2012), and in recent years has been promoted as a bioenergy crop (Sanderson et al. 2004; Vogel 2004; Mitchell et al. 2008). Switchgrass thrives in rain fed systems east of the 100^{th} Meridian (Vogel 2004; Kaul et al. 2006) where non-irrigated (dryland) farming can be conducted in most years (Mitchell et al. 2010). Cellulosic ethanol production from switchgrass has not yet been implemented on a commercial scale in the United States, due to a variety of factors, including a lack of infrastructure for converting plant biomass to ethanol (Mitchell et al. 2012). However, United States government mandates aimed at increasing second generation biofuel production (US EPA 2011) could spur exploratory development.

Switchgrass is heralded for its environmental and economic benefits (Perrin et al. 2008; Dale et al. 2010; Mitchell et al. 2010); however, the ecological impacts of converting marginally productive rowcrop fields to switchgrass are unclear. Most studies addressing the impacts of switchgrass monocultures on wildlife are conducted in Conservation Reserve Program (CRP) switchgrass plantings, which are managed less intensively and are more structurally and florally diverse than bioenergy switchgrass stands (McCoy et al. 2001; Gardiner et al. 2010). Switchgrass stands may provide birds with habitat that annual rowcrops do not, because the conversion of rowcrops to switchgrass involves introducing a native perennial grass back into a highly fragmented agricultural landscape (Robertson et al. 2010). However, switchgrass stands could be detrimental to grassland birds if they replace CRP enrolled grasslands or native prairie polycultures, both of which can be more structurally and florally diverse than switchgrass stands (Murray & Best 2003; Gardiner et al. 2010).

My research objectives were to assess the impacts of past agricultural landuse change and future landuse and climatic changes on bioenergy production, migratory and resident avifauna, and agricultural groundwater use in the Rainwater Basin region of Nebraska, U.S.A. I utilized scenario planning (Peterson et al. 2003; Williams et al. 2009) to explore a range of potential futures for the Rainwater Basin pertaining to bioenergy production, climate change, agricultural landuse and wildlife conservation. Results provide insights into how future changes might affect energy production, groundwater use and avifauna, and could but used to inform future conservation management actions.

The Rainwater Basin is a major watershed of the Greater Platte River Basins (USGS 2009), covering 15,800 km^2 in all or portions of 21 south-central Nebraska counties (LaGrange 2005). In this intensively farmed area, irrigation and dryland farming are the dominant landuses. Groundwater for irrigation is pumped from the underlying Ogallala Aquifer (McGuire 2011), and the majority of the agricultural landscape is utilized for maize and soybean (*Glycine max*) production, although small grain farming and cattle ranching are also conducted on smaller scales (Gilbert 1989; Bishop & Vrtiska 2008). Hundreds of remnant and restored rain–fed wetlands dot the agriculturally dominated landscape, providing critical stopover habitat to migratory waterfowl and shorebird populations (Lagrange 2005). Remnant and restored grassland areas are limited, but still afford critical breeding and feeding habitat to various grassland bird species (Delisle & Savidge 1997; Utrup & Davis 2007; Ramirez–Yanez 2011).

THESIS OVERVIEW

Since 19th Century European settlement, the conversion of Great Plains grasslands and wetlands to agriculture has increased food and bioenergy production, but has decreased grassland bird, waterfowl and shorebird habitat. The High Plains Aquifer has been utilized intensively for rowcrop irrigation, and groundwater supplies have decreased in some areas. Future biofuel–based landuse change and climatic changes could further impact energy production, avian habitat and groundwater withdrawals in the region.

This thesis is organized into 6 chapters that address the impacts of past landuse change and potential future landuse and climatic changes on bioenergy production, resident and migratory avifauna, and water use in the Rainwater Basin region of south–central Nebraska, U.S.A. In this 1st chapter, I provide background information pertinent to the analyses presented hereafter. In the 2nd chapter, I assess the feasibility of supplying adequate biomass for year–round cellulosic ethanol production from residual maize stover and switchgrass within a 40 kilometer service area of the Abengoa Bioenergy ethanol plant near York, Nebraska. In the 3rd chapter, I address how the conversion of marginally productive rowcrop fields to switchgrass under scenarios of climate change could affect Rainwater Basin grassland bird abundances within 40 kilometer starch–based ethanol plant service areas. In the 4th chapter, I consider how conversions between Conservation Reserve Program (CRP) grassland, switchgrass and rowcrops might affect Rainwater Basin grassland bird abundances within 40 kilometer starch–based ethanol plant service areas. In the 5th chapter, I assessed how the adoption of switchgrass as a bioenergy crop could contribute to water conservation goals in water stressed regions under future climatic changes. In the 6th chapter, I utilize multi–model inference to develop predictive models explaining annual variation in the springtime occurrence and flooded area of Rainwater Basin wetlands. In the 7th and final chapter, I present a summary of study results.

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Chapter 2: THE FEASIBILITY OF SUPPLYING ADEQUATE BIOMASS FOR YEAR ROUND CELLULOSIC ETHANOL PRODUCTION AT AN EASTERN NEBRASKA ETHANOL PLANT

ABSTRACT

Government mandates aimed at increasing second generation biofuel production could spur exploratory development in the United States cellulosic ethanol industry. Cellulosic ethanol production has not yet been implemented commercially in the U.S.A., at least partly due to the lack of infrastructure required for converting plant biomass into ethanol. The Abengoa Bioenergy ethanol plant near York, Nebraska has been identified as a potential cellulosic ethanol producer in the near future. To assess the feasibility of supplying adequate biomass for year–round cellulosic ethanol production from residual maize (*Zea mays*) stover and bioenergy switchgrass (*Panicum virgatum*) within a 40 kilometer service area of the York Abengoa Bioenergy ethanol plant, I identified 14,113 hectares of marginally productive cropland within the service area suitable for conversion from annual rowcrops to switchgrass and 131,532 hectares of maize enrolled cropland from which maize stover could be collected. Combined annual maize stover and switchgrass biomass supplies within the service area could range between 428,595 and 752,004 Megagrams (Mg). Approximately 143 – 251 million liters of cellulosic ethanol could be produced from this quantity of biomass, rivaling the current 208 million liter annual starch-based ethanol production capacity of the plant. I conclude that sufficient quantities of biomass could be produced from maize stover and switchgrass within the Abengoa Bioenergy ethanol plant 40 km service area to support year round cellulosic

ethanol production at current feedstock yields, removal rates and bioconversion efficiencies.

INRODUCTION

The United States ethanol industry has developed significantly since production was initiated in the 1980s (Solomon et al. 2007). Despite extensive development, the production of starch-based ethanol from maize (*Zea mays*) grain remains controversial, due to uncertainties over its net energy production, greenhouse gas emissions and competition with food production for landuse (Hill et al. 2006). The potential economic, environmental and ecological benefits of second generation biofuels are increasingly promoted (Tilman et al. 2009), and the production of cellulosic ethanol from plant biomass is the subject of continuing research (Schmer et al. 2008; Mitchell et al. 2012).

Cellulosic ethanol production has not yet been implemented on a commercial scale in the United States, due to a variety of factors including a lack of infrastructure for converting plant biomass to ethanol (Mitchell et al. 2012). However, United States government mandates aimed at increasing second generation biofuel production (US EPA 2011) could spur exploratory development. The Abengoa Bioenergy plant near York, Nebraska has been identified as a candidate for cellulosic ethanol development in the future (LJS 2011). Currently, only starch-based ethanol is produced at the plant (LJS 2011).

Although a variety of plant materials can be converted into cellulosic ethanol, few of them are readily available as feedstocks in the Abengoa plant vicinity. Maize stover is a readily available feedstock in the plant service area (Graham et al. 2007), and is defined as all non-grain, aboveground portions of the maize plant (Wilhelm et al. 2007). Although as much as 75% of maize stover can be removed from fields with conventional farm machinery, only 30 – 50% of maize stover can be sustainably removed annually

(Sheehan et al. 2004; Graham et al. 2007). Retaining residual stover is necessary for preventing erosion and maintaining soil chemistry (Jarecki & Lal 2003; Sheehan et al. 2004). If the Abengoa plant does initiate cellulosic ethanol production, it is likely that maize stover would be the primary feedstock initially supplied to the plant (LJS 2011), although alternative feedstocks could supplement stover in subsequent years.

Switchgrass (*Panicum virgatum*) is an alternative biofuel feedstock that could supplement maize stover for cellulosic ethanol production at the Abengoa Bioenergy ethanol plant in eastern Nebraska, with economic and environmental benefits (Dale et al. 2010; Mitchell et al. 2010). Simple sugars from switchgrass cell walls can be fermented to produce cellulosic ethanol (Dein et al. 2006; Schmer et al. 2008). Economically, switchgrass is a relatively drought tolerant crop (Vogel 2004; Sarath et al. 2008), produces large quantities of biomass on marginally productive croplands (Vogel 2004; Schmer et al. 2008), requires less water and chemical input than annual row crops (Mitchell et al. 2010), requires less intensive management than annual row crops, and could help diversify farmer income (Sanderson et al. 2004; Gopalakrishnan et al. 2009; Mitchell et al. 2010). Environmentally, switchgrass is a near carbon neutral fuel source (McLaughlin et al. 2002; Fargione et al. 2008) that releases less carbon into the atmosphere than the cultivation of traditional row crops (Adler et al. 2007) and sequesters carbon in prairie soils (McLaughlin et al. 2002; Vogel 2004; Mitchell et al. 2010). Perennial grasses like switchgrass are common components of Conservation Reserve Program (CRP) conservation plantings and have been promoted for reducing soil erosion and protecting water resources (McLaughlin & Kszos 2005). Switchgrass is also a net energy positive fuel source (Schmer et al. 2008). While switchgrass is not likely to replace annual rowcrops on productive soils or irrigated fields, it could replace rowcrops on non-irrigated, marginally productive agricultural lands.

Marginally productive lands can include small, complexly shaped, non-irrigated portions of agricultural fields (Mitchell et al. 2012). However, soil type, mean annual precipitation, and irrigation limitations could also be important for defining marginally productive agricultural lands. Some non-irrigated fields may remain profitable for rowcrop production if they receive adequate precipitation and lie on fertile soils, and therefore should not be considered marginal. Alternatively, larger, non-irrigated fields on poor soils and in dry areas may be marginally productive for rowcrop agriculture. However, not all rowcrop fields identified as marginally productive may be initially converted to alternative forms of landuse, because individual farmers will decide how to manage their land.

It is unclear if adequate biomass can be produced in proximity to cellulosic ethanol plants to support year-round cellulosic ethanol production. In this study, I assessed the feasibility of producing cellulosic ethanol from potential maize stover and switchgrass biomass supplies in a 40 kilometer service area around the Abengoa Bioenergy ethanol plant near York, NE, USA. I employed a conservative approach to identify marginally productive rowcrop fields that might be converted to switchgrass in the future, estimated potential switchgrass and maize stover biomass supplies available for annual removal, projected cellulosic ethanol yield at current feedstock bioconversion efficiencies, and compared cellulosic plant production potential to current starch-based ethanol plant production capacities.

METHODS

STUDY AREA

The 40 kilometer service area of the York Abengoa Bioenergy ethanol plant encompasses portions of Butler, Polk, Seward and York counties in south-central Nebraska. The service area is situated in an intensively farmed agricultural landscape, where both irrigation and dryland farming are common. Groundwater for irrigation is obtained from the underlying Ogallala Aquifer (McGuire 2011). The majority of the agricultural landscape is used for maize and soybean (*Glycine max*) production, although small grain farming and cattle ranching are also conducted on smaller scales (Gilbert 1989; Bishop & Vrtiska 2008).

DATA SOURCES

Agricultural irrigation type GIS data was provided by the Rainwater Basin Joint Venture (http://www.rwbjv.org), and Nebraska roads GIS data was downloaded online from the Nebraska Department of Natural Resources (http://dnr.ne.gov/databank/statewide.html). Geographic coordinates of the Abengoa ethanol plant location were obtained from Google Earth satellite imagery and digitized in ArcGIS. Average 2010 maize grain yields for Butler, Polk, Seward and York counties were obtained online from the National Agricultural Statistics Service (NASS) (http://www.nass.usda.gov), and mean switchgrass yields for the Northern Great Plains were from Schmer et al. (2008). Prior to analysis, all GIS data layers were projected in the North American Datum 1983 Universal Transverse Mercator Zone 14 North coordinate system.

AGRICULTURAL LANDUSE

Within the Abengoa plant service area, agricultural landuse was categorized into 4 types: center-pivot irrigated fields, pivot corners, gravity irrigated fields and dryland fields. A Center-pivot is a large sprinkler system generally anchored at a field center point and connected to a groundwater well. Groundwater is pumped through a pipe extending from the field center to the least distant field perimeter, with multiple two – wheeled moving towers supporting the pipe along its extent. As the center-pivot moves in a circular motion around the field, sprinklers connected to the pipe release water to the soil surface. Pivot corners result from irrigating square shaped agricultural fields with circular center-pivot irrigation patterns. Because center-pivots fail to move across pivot corners, the corners are not supplied with water. Several means of irrigating pivot corners exist, including center-pivot corner systems or lateral irrigation pipes; however, farmers in the Rainwater Basin commonly raise crops on pivot corners without irrigation. A gravity irrigation system consists of a temporary lateral irrigation line extending along the field edge with the greatest altitude and perpendicular to the direction of crop rows. Water from lateral lines is released into furrows between crop rows and is pulled by the force of gravity toward the opposite end of the field. A dryland field is not irrigated by any means. In years with adequate growing season precipitation, dryland field and pivot corner grain yields are comparable with irrigated croplands; however, in drier years, they yield less.

CROPLAND CLASSIFICATION

A list of marginal criteria making agricultural fields in the Rainwater Basin region of Nebraska, U.S.A. suitable for conversion from annual rowcrops to switchgrass was compiled and consisted of irrigation type, agricultural suitability of underlying soils, field size and shape complexity, mean annual precipitation, and relative risk of experiencing additional irrigation limitations in the future. Each field was assigned to 1 of 24 marginality classes, based on the number of marginal criteria it satisfied. The more marginal criteria a field satisfied, the more suitable it was considered for conversion from rowcrops to switchgrass.

Dryland fields and pivot corners were considered more marginal than gravity and center-pivot irrigated fields, due to the lack of irrigation systems on dryland fields and pivot corners and the fact that switchgrass is more drought tolerant and water use efficient than maize (Kiniry et al. 2008). Fields were also classified according to the agricultural suitability of soils underlying field center points. The USDA Natural Resource Conservation Service (NRCS) groups soils into land capability classes, based on their suitability for agricultural production. Soils in classes 1 and 2 are considered most suitable for agriculture, while soils in classes 7 and 8 are considered completely unsuitable. Soils in classes 3, 4, 5 and 6 can be described as marginally productive agricultural lands, and may be better suited to less intensive forms of agricultural landuse, which could include seeding with perennial grasses like switchgrass. Fields located on soils in NRCS land capability classes 3, 4, 5 or 6 were considered more marginal than fields located on soils in classes 1 or 2.

Small, complexly shaped dryland fields were considered more marginal than larger, more uniformly shaped dryland fields. Farming rowcrops on small, complexly shaped fields with increasingly large, modern farm equipment can be inconvenient and time consuming, and these fields could be better suited to raising less management intensive switchgrass stands. All pivot corners were considered marginal, as were dryland fields with areas less than the mean pivot corner area (3.7 hectares). Dryland fields with areas greater than 3.7 hectares, but less than the 25th percentile value for dryland field area (4.7 hectares), and with a shape index greater than the 75th percentile value for dryland field shape index (1.56), were considered more marginal than larger and more uniformly shaped dryland fields.

Due to the rain shadow effect of the Rocky Mountains, precipitation increases from west to east across the Rainwater Basin, with drier areas located in the western half (Rickets et al. 1999). Fields in areas with a mean annual precipitation of 63.5 centimeters or less were considered dry and more marginal than fields in areas with a mean annual precipitation more than 63.5 centimeters, since switchgrass is more productive than rowcrops under drier conditions (Kiniry et al. 2008). Finally, center-pivot irrigated and gravity irrigated fields were classified according to their potential to experience additional irrigation limitations in the future. Fields were assigned to a high risk or low risk category, based on the Natural Resource District (NRD) in which they were located. NRDs with histories of implementing moratoriums or stays on wells and hectares were combined and classified as being at high risk for additional irrigation limitations, whereas those without previously implemented moratoriums or stays were combined and classified as being at low risk for future limitations. If NRDs restrict agricultural irrigation in these regions in the future, switchgrass could replace rowcrops on some previously irrigated fields.

Croplands classified as most marginal and suitable for conversion to switchgrass under present climatic conditions were pivot corners and small, complexly shaped dryland fields, located on soils in NRCS land capability classes 3, 4, 5 or 6 and in areas with annual average precipitation of 63.5 centimeters or less. Croplands classified as least marginal and unsuitable for conversion to switchgrass were gravity and pivot irrigated fields, and large, uniformly shaped dryland fields located on soils in NRCS land capability classes 1 or 2, in areas with annual average precipitation greater than 63.5 centimeters and at low risk of experiencing irrigation limitations. Remaining croplands were placed into intermediate classes, according to the number of marginal criteria they satisfied (Table 1).

GIS ANALYSIS

The Network Analyst extension in ArcGIS was used to generate a 40 kilometer service area for the Abengoa Bioenergy ethanol plant, using all Nebraska roads as travel corridors (Figure 1). Forty kilometers is recognized as the approximate maximum distance at which producers can economically transport grain or other feedstocks to biorefineries for processing (Khanna et al. 2008; Mitchell et al. 2012). The Abengoa plant service area overlaps with 40 kilometer service areas of three neighboring starch-based ethanol plants. However, the Abengoa plant service area was allowed to encroach into neighboring plant service areas, since none of these plants have been identified as cellulosic ethanol producers in the near future.

The geostatistical analyst extension in ArcGIS was used to assign percentages of fields in different marginality classes to be converted to bioenergy switchgrass. I used a conservative approach to determine rowcrop fields that could be converted to switchgrass in the future. Even though many fields possess marginal characteristics, not all fields are expected to be immediately converted to switchgrass, since individual farmers will decide which landuse types to enroll their properties in. I assigned greater conversion percentages to classes satisfying more marginal criteria. However, only classes satisfying all marginal criteria had 100% of fields converted to switchgrass. Classes satisfying fewer marginal criteria were assigned lower switchgrass conversion percentages of 75%, 50%, 25% or 0%. Fields not assigned to switchgrass conversion were assumed to remain in rowcrop production. The conservative approach to switchgrass conversion ensured that only a proportion of rowcrop fields satisfying at least one of the marginal criteria were converted to switchgrass.

Shapefiles representing croplands converted to switchgrass in each of the 24 marginality classes were combined into a single shapefile. Shapefiles representing remaining rowcrops were combined similarly. Total rowcrop and switchgrass shapefiles for the entire Rainwater Basin region were restricted to the previously generated 40 kilometer Abengoa Bioenergy ethanol plant service area (Figure 2), and resulting rowcrop and switchgrass shapefiles were converted from vector to raster format. Resulting raster layers were reclassified into single classes and total bioenergy switchgrass and rowcrop areas were obtained by inputting reclassified rasters into the program Fragstats.

BIOMASS SUPPLIES

Total maize enrolled hectares in the service area were calculated by multiplying the number of hectares remaining in annual rowcrop production after the conversion of marginally productive agricultural lands to switchgrass by 0.5. A 1:1 ratio of maize to soybean hectares for remaining annual rowcrops in the service area was assumed. This may represent a conservative estimate of total maize enrolled hectares, because some fields in the region are not rotated between maize and soybean production semi-annually, but are instead used to grow maize for at least two consecutive years. Average maize grain yield for the plant service area was determined by averaging mean maize grain yields from the four Nebraska counties the service area occupies. A 1:1 weight distribution between maize grain and aboveground non-grain maize stover (Graham et al. 2007) was assumed, and mean maize grain and stover weight per hectare were considered equal. Mean maize grain and stover weight per hectare was obtained by multiplying mean maize grain yield per acre by 47 pounds, the dry matter (DM), or 0% moisture weight, of one bushel of maize grain (Graham et al. 2007), and then converting the result to kg ha^{-1}. Annual maize stover removal rates of 30 – 50% were considered sustainable for maintaining soil chemistry. To determine the maize stover weight range that could be collected per hectare at 30% and 50% removal rates, total maize stover weight per hectare was multiplied by 0.3 and 0.5. Total maize stover weight available for annual removal from the Abengoa service area was calculated by multiplying the upper and lower weight ranges of sustainably removable maize stover per hectare by the total number of maize enrolled hectares in the service area.

Switchgrass biomass yields for the service area were assumed to average between 5 and 11 Megagrams (Mg) ha^{-1} (Schmer et al. 2008). The potential range in switchgrass biomass quantity for the service area was calculated by multiplying the total number of hectares considered suitable for switchgrass production by 5 Mg ha^{-1} and 11 Mg ha^{-1}. Total switchgrass and maize stover biomass weights for the service area were summed to determine the total quantity of biomass that could be sustainably supplied to the Abengoa plant annually. Both maize stover and switchgrass biomass were assumed to have bioconversion efficiencies of 334 l Mg^{-1} (Varvel et al. 2008; Mitchell et al. 2012), which are less than reported theoretical maximum bioconversion efficiencies for those crops (Schmer et al. 2008). Multiplying the total maize stover and switchgrass biomass yield range for the plant service area by 334 l Mg^{-1} yielded the potential range in cellulosic ethanol volume that could be sustainably produced annually at the Abengoa plant.

RESULTS

Mean 2010 maize grain dry DM yield for the four counties occupied by the Abengoa Bioenergy ethanol plant 40 kilometer service area was 9,074 kg ha^{-1}. A 1:1 weight distribution between maize grain and stover results in a maize stover weight of 9,074 kg ha^{-1}. A stover removal rate of 30 – 50% allows for 2.72 – 4.54 Mg of stover to be sustainably collected annually. Of the 277,177 hectares of cropland in the plant service area, 14,113 hectares were found suitable for conversion to switchgrass, and the remaining 263,064 hectares of non-bioenergy switchgrass cropland were assumed to consist of 131,532 hectares of maize and 131,532 hectares of soybeans. Multiplying the total number of bioenergy switchgrass enrolled hectares by the average switchgrass DM yield range of 5 – 11 Mg ha^{-1} results in a total switchgrass yield of 70,565 – 155,243 Mg (Table 2). Multiplying the number of maize enrolled hectares by the megagrams of maize stover available for removal per hectare results in an annual removal of 358,030 – 596,761 Mg of maize stover (Table 3). The sum of switchgrass and maize stover supplies in the plant service area is a total annual biomass production potential of 428,595 – 752,004 Mg. At a bioconversion efficiency of 0.334 l kg^{-1}, the annual cellulosic ethanol production capacity of the Abengoa plant is 143,150,730 – 251,169,336 liters.

DISCUSSION

Results of this analysis suggest that in addition to the 208 million liters of starch-based ethanol already produced at the Abengoa Bioenergy plant annually (NEB 2011), another 143 – 251 million liters of cellulosic ethanol could be produced annually from maize stover and switchgrass. Mitchell et al. (2012) recommend supplying 115 – 120% of required biomass to cellulosic ethanol plants annually, in order to account for biomass yield variability and storage losses. The 428,595 – 752,004 Mg of estimated annual biomass produced within the Abengoa service area provides 77 – 135% of the 556,990 Mg of biomass necessary to support a cellulosic ethanol plant with an annual ethanol output of 189,270,590 liters.

These production estimates assume the conversion of 14,113 hectares of marginally productive agricultural lands to switchgrass within the plant service area (~5% of total cropland area). Without converting any hectares from rowcrops to switchgrass, the 138,589 maize enrolled hectares within the plant service area could supply 377,239 – 628,778 Mg of biomass annually at a stover removal rate of 30% – 50%. This quantity of corn stover provides 68 – 113% of the 556,990 Mg of biomass necessary to support a cellulosic ethanol plant with an annual ethanol output of 189,270,590 liters. Because farmers may not be willing to plant bioenergy switchgrass without an operational biorefinery already in place (Mitchell et al. 2012), maize stover may be utilized exclusively as a feedstock while switchgrass stands establish in the years following plant construction. Even if marginally productive croplands are taken out of production and seeded to switchgrass, the 131,532 hectares of remaining maize enrolled cropland could supply 358,030 – 596,761 Mg of maize stover biomass, or 64 – 107% of the biomass necessary for supporting an ethanol plant with an annual output of 189,270,590 liters.

Although not considered in this analysis, grasslands enrolled in the Conservation Reserve Program (CRP) could be converted to bioenergy switchgrass stands or back to annual rowcrops, and thereby contribute to biomass supplies. There are approximately 493 hectares of CRP enrolled grassland within the 40 kilometer Abengoa ethanol plant service area. Converting these lands to switchgrass could yield an additional 2,465 – 5,423 Mg of switchgrass or 1,342 – 2,237 Mg of maize stover biomass annually.

In the future, switchgrass yields are projected to increase with the introduction of improved hybrids (Vogel & Mitchell 2008), resulting in greater biomass quantities being supplied to the plant. Similarly, extending the Abengoa plant service area farther than 40 kilometers could increase biomass supplies. Farmers may be willing to transport feedstocks farther than 40 kilometers if economic incentives are provided and if there is only 1 plant producing cellulosic ethanol in the vicinity. Increased biomass supplies would make cellulosic ethanol production more feasible and less vulnerable to variations in annual biomass supply. Even without increased biomass supplies, supplying adequate biomass for year round cellulosic ethanol production at the York Abengoa Bioenergy ethanol plant appears to be feasible at current maize stover and switchgrass biomass yields and bioconversion efficiencies.

Results of this small scale analysis provide insights into the feasibility of cellulosic ethanol production in the surrounding Great Plains. The Abengoa plant is located in a highly cultivated landscape, where maize and soybean production dominates landuse. In the Abengoa plant service area, it is likely that maize stover will be more available than switchgrass. However, less cultivated landscapes, which likely have a greater proportion of marginally productive lands, may be capable of producing more switchgrass. Regardless of if maize stover or switchgrass is utilized as the primary feedstock, adequate biomass supplies for year-round cellulosic ethanol production can be produced in close proximity to ethanol plants.

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TABLES AND FIGURES

Table 1: List of 24 rowcrop field marginality classes and percentages of marginality classes converted to switchgrass in the Abengoa Bioenergy ethanol plant service area. Fields were classified according to irrigation type, soil quality, mean annual precipitation and potential for experiencing irrigation limitations in the future. Conversion percentages of 0, 25, 50, 75 or 100 were assigned to marginality classes, according to the number of marginal criteria fields composing the class satisfied.

Landuse classification	**Conversion**
Pivot corners + poor soils + dry area	100%
Pivot corners + poor soils + wet area	75%
Pivot corners + good soils + dry area	75%
Pivot corners + good soils + wet area	50%
Small dryland fields + poor soils + dry area	100%
Small dryland fields + poor soils + wet area	75%
Small dryland fields + good soils + dry area	75%
Small dryland fields + good soils + wet area	50%
Large dryland fields + poor soils + dry area	25%
Large dryland fields + poor soils + wet area	0%
Large dryland fields + good soils + dry area	0%
Large dryland fields + good soils + wet area	0%
Gravity + poor soils + dry area + high risk of irrigation limitations	0%
Gravity + poor soils + wet area + high risk of irrigation limitations	0%
Gravity + good soils + dry area + high risk of irrigation limitations	0%
Gravity + good soils + wet area + high risk of irrigation limitations	0%
Gravity + poor soils + wet area + low risk of irrigation limitations	0%
Gravity + good soils + wet area + low risk of irrigation limitations	0%
Pivots + poor soils + dry area + high risk of irrigation limitations	0%
Pivots + poor soils + wet area + high risk of irrigation limitations	0%
Pivots + good soils + dry area + high risk of irrigation limitations	0%
Pivots + good soils + wet area + high risk of irrigation limitations	0%
Pivots + poor soils + dry area + low risk of irrigation limitations	0%
Pivots + good soils + wet area + low risk of irrigation limitations	0%

Table 2: Potential annual switchgrass biomass and ethanol production potential within the 40 kilometer road network service area of the Abengoa Bioenergy ethanol plant, assuming 5 Mg ha^{-1} and 11 Mg ha^{-1} switchgrass DM yields and an ethanol bioconversion efficiency of 334 l Mg^{-1}.

Switchgrass yield	Total biomass	Ethanol produced
5 Mg/ha	70,565 Mg	23,568,710 liters
11 Mg/ha	155,243 Mg	51,851,162 liters

Table 3: Potential annual maize stover biomass and ethanol production potential within the 40 kilometer road network service area of the Abengoa Bioenergy ethanol plant, assuming a maize stover DM yield of 9,074 kg ha^{-1}, 30% and 50% annual maize stover removal rates, and an ethanol bioconversion efficiency of 334 l Mg^{-1}.

Maize stover DM yield	**Stover removal**	**Total biomass**	**Ethanol produced**
9.074 Mg ha^{-1}	30%	358,030 Mg	119,582,020 liters
9.074 Mg ha^{-1}	50%	596,761 Mg	199,318,174 liters

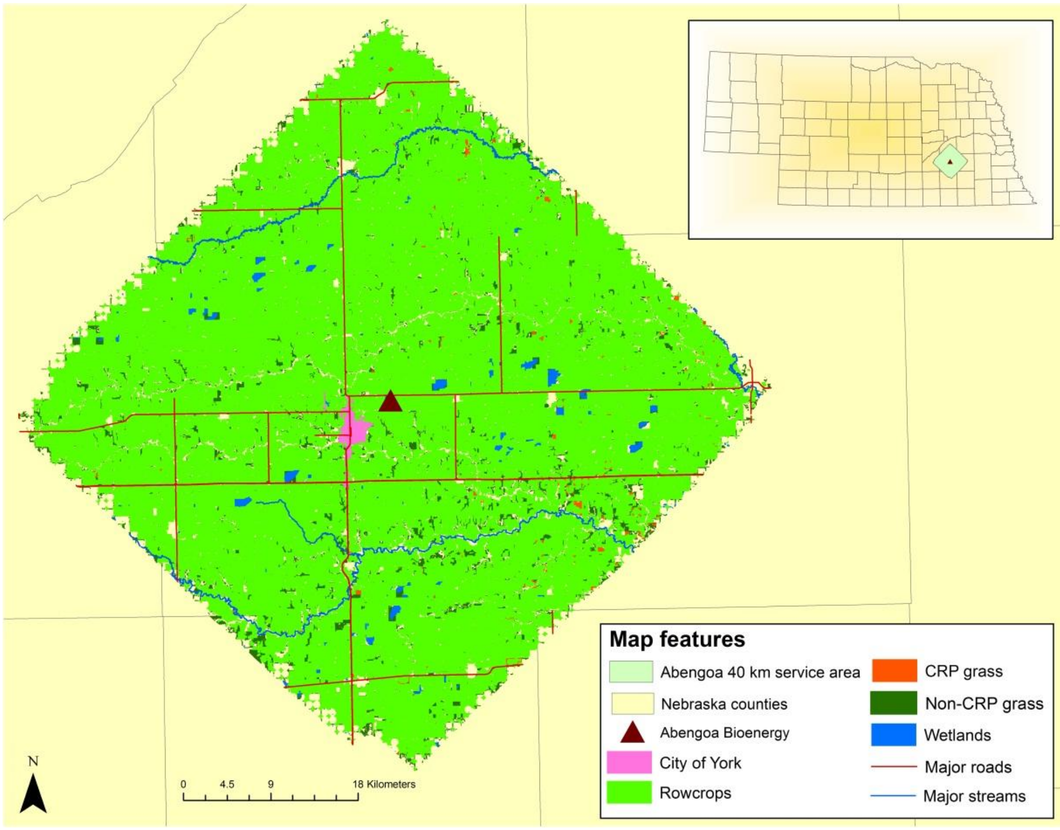

Figure 1: Location of and current major landcover classes within a 40 kilometer road network service area of the Abengoa Bioenergy Ethanol Plant near York, Nebraska. Rowcrops are the aggregation of all irrigated and non-irrigated rowcrop fields from Rainwater Basin Joint Venture (RWBJV) 2006 agricultural irrigation type data.

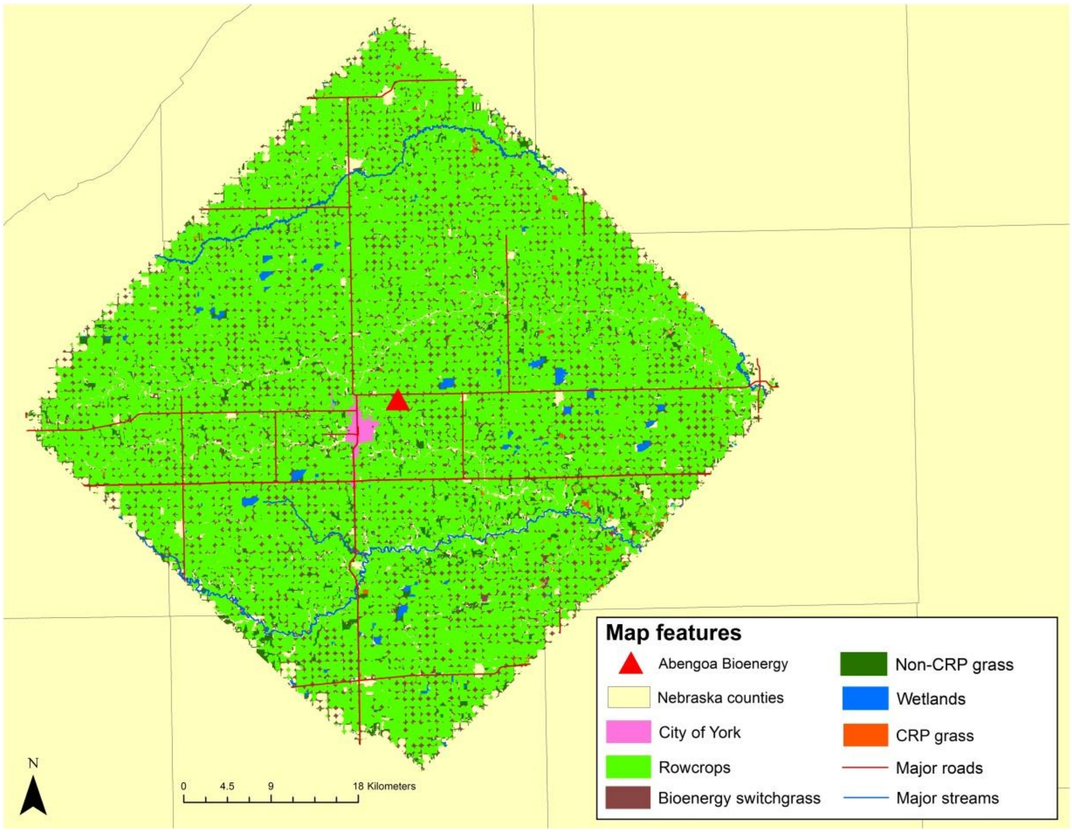

Figure 2: Major landcover classes in an Abengoa Bioenergy ethanol plant 40 kilometer service area, following the conversion of some marginally productive rowcrop fields to bioenergy switchgrass. Rowcrop fields were grouped into marginality classes, according to irrigation type, size, shape, soils and likelihood of experiencing irrigation limitations in the future. 25% – 100% of marginality classes composed of non-irrigated fields on poor agricultural soils were converted to switchgrass, according to the number of marginal characteristics fields in the classes possessed. Unconverted rowcrop fields are the aggregation of all irrigated and non-irrigated rowcrop fields from Rainwater Basin Joint Venture (RWBJV) 2006 agricultural irrigation type data.

CHAPTER 3: PREDICTED GRASSLAND BIRD RESPONSES TO BIOFUEL-BASED LANDUSE CHANGE IN NEBRASKA'S RAINWATER BASIN

ABSTRACT

The conversion of native prairie to agriculture has resulted in significant North American grassland bird population declines. Biofuel crops could further transform North American landscapes. Switchgrass (*Panicum virgatum*) is an alternative biofuel feedstock that may be environmentally and economically superior to maize (*Zea mays*) grain for ethanol production on marginally productive agricultural lands. It is unclear how the conversion of marginally productive rowcrop fields to switchgrass might impact grassland bird populations. In this chapter, I developed three agricultural landuse change scenarios for the Rainwater Basin region of Nebraska, USA, driven by potential future climatic changes, irrigation limitations, grain market prices and cellulosic ethanol demand. For each scenario, I generated spatially explicit maps of rowcrop and switchgrass distributions and calculated changes in rowcrop and switchgrass area. Changes in area were input into a customized version of the Hierarchical All Birds Strategy (HABS) model to predict changes in avian abundances. Abundances of most species increased following the conversion of rowcrops to switchgrass, with greater responses exhibited under scenarios where greater areas of rowcrops were converted to switchgrass. Species displaying the most positive responses were sedge wrens (*Cistothorus plantensis*), grasshopper sparrows (*Ammodramus savannarum*) and ring-necked pheasants (*Phasianus colchicus*). Switchgrass could improve habitat for and increase abundance of multiple grassland bird species if it replaces rowcrops, but impacts

are ultimately contingent on which forms of landuse switchgrass replaces and how switchgrass stands are managed.

INTRODUCTION

Prior to 19th Century European settlement, landcover in the North American Great Plains consisted primarily of varying types of short, tall and mixed grass prairie, with perennial grasses constituting the dominant species (Weaver 1968; Barker & Whitman 1988; Knopf 1994; Hart & Hart 1997). Since settlement, agricultural practices have converted grassland into cropland for the production of maize (*Zea mays*), soybeans (*Glycine max*), wheat (*Triticum aestivum*), milo (*Sorghum bicolor*), alfalfa (*Medicago sativa*) and cotton (*Gossypium spp.*) (Musick et al. 1990; Guru & Horne 2000; Mitchell et al. 2010b). Global food and bioenergy demands are continuing to rise, driving additional landuse conversions (Tilman et al. 2002; Fargione et al. 2008). According to current estimates, approximately 99% of the Northern Great Plains is either farmed or grazed by livestock (Forrest et al. 2004). Remaining areas of grassland are fragmented by woody vegetation (Grant et al. 2004), croplands and roads (White et al. 2000) and often do not constitute the large tracts of prairie necessary for many species (Manning 1995; Helzer & Jelinski 1999; Freese et al. 2007). North American mixedgrass prairie is an endangered ecosystem type and tallgrass prairie is critically endangered (Ricketts et al. 1999; Samson et al. 2004). In addition to prairie remnants, North American grasslands include rangelands, grassland buffers along water bodies, and Conservation Reserve Program (CRP) grasslands (Delisle & Savidge 1997; Utrup & Davis 2007). The CRP provides landowners with monetary incentives for removing highly erodible croplands from rowcrop production and seeding them with conservation plantings, which in addition to reducing soil erosion, benefit water resources and wildlife (Ribaudo 1989; Dunn et al. 1993; USDA – NRCS 2012).

Grassland birds have been negatively impacted by reductions and fragmentation of native grasslands over the past two centuries (Herkert 1994; Samson & Knopf 1994; White et al. 2000; Grant et al. 2004). Grassland birds have experienced greater, faster and more widespread declines than any other ecological guild on the North American continent (Herkert et al. 1993; Knopf 1994; Sauer et al. 2011). Remnant and restored grasslands provide grassland birds with crucial feeding and breeding habitat (Johnson & Schwartz 1993; Johnson & Igl 1995; Ramirez-Yanez et al. 2011); therefore, effective grassland conservation, restoration and management are paramount to reversing grassland bird population declines.

Global climate models predict that the Great Plains region will continue to experience climatic changes throughout the 21st Century (IPCC 2007; Karl et al. 2009). In its 4th report, the Intergovernmental Panel on Climate Change (2007) projected maximum and minimum temperature increases, winter precipitation increases and summer precipitation decreases for the Northern Great Plains during the 21st Century. Average annual temperature increases of 0.6 – 2.2 degrees Celsius are projected by 2020, and increases of 1.1 – 7.2 degrees Celsius are projected by 2090 (Karl et al. 2009). Although both maximum and minimum temperatures are projected to rise, minimum temperatures are expected to increase more (IPCC 2007). Precipitation changes are less certain than temperature changes (Ojima & Lackett 2002); however, a higher proportion of precipitation is expected to fall in fewer major storm events than does presently (IPCC 2007). These changes could directly and indirectly impact societies, economies, and the environment (Root et al. 2003; Karl et al. 2009).

Demand for clean, renewable energy has resulted in recent, large scale efforts to utilize maize grain for ethanol production (Schnoor et al. 2008). Despite extensive development of the starch-based ethanol industry, ethanol production from maize grain remains controversial, due to uncertainties over its net energy production, water use efficiency, ability to reduce atmospheric greenhouse gas emissions and concentrations, and competition with food production for landuse (Berndes 2008; Searchinger et al. 2008; Tilman et al. 2009; Dale et al. 2010). Meanwhile, the benefits of second generation biofuels are increasingly promoted (Tilman et al. 2009). One alternative biofuel feedstock proposed for large scale cellulosic ethanol production in Great Plains agricultural landscapes is bioenergy switchgrass (*Panicum virgatum*) (Mitchell et al. 2012).

Switchgrass is a warm-season, perennial, C4 grass species, native to the Great Plains (Vogel 2004; Kaul et al. 2006). Switchgrass has been the subject of long-term agronomic research (Schmer et al. 2008; Mitchell et al. 2012), and in recent years has been promoted as a bioenergy crop (Sanderson et al. 2004; Vogel 2004; Mitchell et al. 2008). Simple sugars from switchgrass cell walls can be fermented to produce cellulosic ethanol (Dein et al. 2006). Switchgrass thrives in rain fed systems east of the 100th Meridian (Vogel 2004; Kaul et al. 2006) where non-irrigated (dryland) farming can be conducted in most years (Mitchell et al. 2010b).

Switchgrass is an alternative biofuel feedstock that could be produced on a large scale in agricultural landscapes, with environmental and economic benefits (Perrin et al. 2008; Dale et al. 2010; Mitchell et al. 2010b). Economically, switchgrass is a relatively drought tolerant crop (Vogel 2004; Sarath et al. 2008), produces large quantities of biomass on marginally productive lands (Vogel 2004; Schmer et al. 2008), requires less water and chemical inputs than annual rowcrops, requires less intensive management than annual rowcrops, can be managed and harvested using traditional farm machinery (Mitchell et al. 2010b), and could help diversify farmer income (Sanderson et al. 2004; Gopalakrishnan et al. 2009; Mitchell et al. 2010b). Switchgrass is also net energy positive (Schmer et al. 2008). Environmentally, switchgrass is a near carbon neutral fuel source (McLaughlin et al. 2002; Fargione et al. 2008) that releases less carbon into the atmosphere than rowcrop cultivation (Adler et al. 2007) and sequesters carbon in prairie soils (McLaughlin et al. 2002; Vogel 2004; Mitchell et al. 2010b). Perennial grasses like switchgrass are common components of CRP conservation plantings and have been promoted for reducing soil erosion and protecting water resources (McLaughlin & Kszos 2005).

Bioenergy switchgrass is less management intensive than annual rowcrops. In the year of switchgrass seeding, herbicide application may be necessary to control non-preferred vegetation in stands (Mitchell et al. 2010a), but the need for application generally decreases in years following establishment (Sarath et al. 2008). Switchgrass stands also require the addition of nitrogen fertilizer each spring (Vogel et al. 2002; McLaughlin & Kszos 2005; Sarath et al. 2008), but less than annual row crops (Mitchell et al. 2011). Switchgrass biomass is typically harvested once each year through traditional haying methods in the late summer or early autumn, after grassland bird nesting seasons have concluded (Sarath et al. 2008; Mitchell et al. 2010b). Harvesting switchgrass at anthesis in late summer yields maximum biomass, but delaying harvest until after the first killing in autumn frost allows for nitrogen storage in plant roots, which decreases the amount of nitrogen that must be applied the following spring (Vogel et al. 2002).

It is unclear how the conversion of marginally productive agricultural lands to switchgrass might affect grassland bird populations, because most studies addressing the impacts of switchgrass monocultures on wildlife are conducted in CRP switchgrass plantings, which are managed less intensively and are more structurally and florally diverse than bioenergy switchgrass stands (McCoy et al. 2001; Gardiner et al. 2010). Structurally and florally heterogeneous grasslands provide quality habitat for multiple grassland bird species (Delisle & Savidge 1997), while more homogeneous grasslands may benefit only select species (Murray & Best 2003). Annual rowcrop fields are intensively managed, low diversity plant communities that typically support very low grassland bird densities (Johnson & Igl 1995; Best et al. 1997). Switchgrass stands would likely provide birds with habitat that annual rowcrops do not, because the conversion of rowcrops to switchgrass involves introducing a native perennial grass back into a highly fragmented agricultural landscape (Robertson et al. 2010). Nest destruction commonly associated with the early season haying practices may be avoided by the later season haying of switchgrass stands (Mitchell et al. 2010b), although lower rates of nest destruction could occur during spring fertilizer or herbicide application (Murray & Best 2003). Furthermore, annual haying, fertilizer and herbicide application, and the introduction of improved switchgrass hybrids could alter the floral and structural composition of switchgrass stands (Mitchell et al. 2012), thereby influencing the utilization of switchgrass stands by grassland birds.

Marginally productive agricultural lands can include small, complexly shaped, non-irrigated portions of agricultural fields (Mitchell et al. 2012). However, soil type, mean annual precipitation, other climatic factors, and irrigation limitations could also be important for defining marginally productive agricultural lands. Some non-irrigated fields may remain profitable for rowcrop production if they receive adequate precipitation and are located on fertile soils, and therefore should not be considered marginal. Alternatively, larger, non-irrigated fields on poor soils and in dry areas may be marginally productive for rowcrop agriculture. However, not all rowcrop fields identified as marginally productive may be initially converted to alternative forms of landuse, because individual farmers will decide how to manage their land.

Scenarios are structured accounts of possible future events (Peterson et al. 2003). Scenario planning is appropriate in situations where there is a high level of uncertainty over uncontrollable future events and their impacts (Peterson et al. 2003; Williams et al. 2009). The future and impacts of climate change are largely uncertain; however, changes are expected to proceed in the short term, regardless of changes in landuse or greenhouse gas emissions (IPCC 2007; McDonald et al. 2009). Similarly, agricultural policy adjustments resulting from climate change or other factors are speculative, but still have potential to occur (Olesen & Bindi 2002). Both climatic changes and policy adjustments could become major future drivers of landuse change in agricultural landscapes, thereby influencing grassland bird populations (Fletcher et al. 2009).

It is unclear how different forms and intensities of agricultural landuse change might occur in the future and affect Great Plains grassland bird populations. In this study, I developed three biofuel-based agricultural landuse change scenarios for the

Rainwater Basin region of south-central Nebraska, U.S.A., each driven by potential future climatic changes, irrigation limitations, commodity prices and cellulosic ethanol demand. I employed a conservative approach to determine which marginally productive rowcrop fields that might be converted to switchgrass in the future. For each scenario, I generated spatially explicit landcover maps and calculated changes in rowcrop and switchgrass area within 40 kilometer road network service areas of existing ethanol plants. Changes in area were input into a customized version of the Hierarchical All Birds Strategy (HABS) model (PLJV 2007) to predict changes in abundance for a suite of grassland bird species under different landuse change scenarios. Changes in abundance were compared between scenarios to assess how the conversion of rowcrops to switchgrass might impact different grassland bird species.

METHODS

STUDY AREA

The Rainwater Basin is a major watershed of the Greater Platte River Basins (USGS 2009), covering 15,800 km^2 in all or portions of 21 south-central Nebraska counties (LaGrange 2005) (Figure 1). This study was conducted within the 40 kilometer service areas of 11 ethanol plants currently servicing the Rainwater Basin region. In this intensively farmed area, irrigation and dryland farming are the dominant landuses. Groundwater for irrigation is pumped from the underlying Ogallala Aquifer (McGuire 2011), and the majority of the agricultural landscape is utilized for maize and soybean (*Glycine max*) production, although small grain farming and cattle ranching are also conducted on smaller scales (Gilbert 1989; Bishop & Vrtiska 2008). Hundreds of remnant and restored rain–fed wetlands also dot the agriculturally dominated landscape, providing wildlife habitat, environmental services and recreational opportunities (Lagrange 2005).

DATA SOURCES

Agricultural irrigation type and complete landcover Geographic Information System (GIS) data for the Rainwater Basin region were provided by the Rainwater Basin Joint Venture (http://www.rwbjv.org). The Soil Survey Geographic database (SSURGO) was downloaded from the U.S. Department of Agriculture (USDA) – Natural Resource Conservation Service (NRCS) Soil Data Mart (http://soildatamart.nrcs.usda.gov); road, stream and political boundary GIS data were downloaded from the Nebraska Department of Natural Resources (http://dnr.ne.gov/databank/spat.html); and mean annual precipitation GIS data were downloaded online from the USDA – NRCS (http://datagateway.nrcs.usda.gov/GDGOrder.aspx). Geographic coordinates of ethanol plants servicing the Rainwater Basin were collected from Google Earth satellite imagery, and plant locations were digitized in ArcGIS. All GIS data layers were projected in the North American Datum 1983 Universal Transverse Mercator Zone 14 North coordinate system.

ETHANOL PLANT SERVICE AREAS

Agricultural lands in close proximity to ethanol plants are more likely to experience biofuel-based landuse change in the future than lands located farther from ethanol plants, due to the availability of starch-based ethanol conversion infrastructure that could be modified for cellulosic ethanol production (Mitchell et al. 2012). I used the Network Analyst extension in ArcGIS to generate 40 kilometer network service areas for 11 ethanol plants currently servicing the Rainwater Basin, using all roads within the State of Nebraska as travel corridors. 40 kilometers is the approximate maximum distance producers are willing to transport grain or feedstocks to biorefineries for processing (Khanna et al. 2008; Mitchell et al. 2012). I used the 40 kilometer service area boundaries to restrict the Rainwater Basin landcover and agricultural irrigation type GIS data layers to service area boundaries for the development of scenarios.

LANDCOVER CLASSES

I identified 4 major agriculturally or ecologically important landcover classes that together account for over 89% of total Rainwater Basin landcover within the ethanol plant service areas. These landcover classes are rowcrops, CRP enrolled grasslands, non-CRP grasslands and wet meadows. Remaining landcover consists primarily of woodlands and developed areas. Woodlands and developed areas accounted for the majority of remaining land area; however neither are directly utilized for rowcrop production or grassland bird habitat, and therefore were omitted from further analysis.

Irrigation system types were used to distinguish between different rowcrop fields. Irrigation system types were center-pivots, pivot corners, gravity irrigation and dryland fields. A center-pivot is a large sprinkler system typically anchored at a field center point and connected to a groundwater well. Groundwater is pumped through a pipe extending from the center point to the least distant field perimeter, with multiple two–wheeled moving towers supporting the pipe along its extent. As the center-pivot moves in a circular motion around the field, sprinklers connected to the pipe release water to the soil surface. Pivot corners result from irrigating square shaped properties with circular center-pivot irrigation systems. Since center-pivots fail to move across pivot corners, the corners are not supplied with water. Several means of irrigating pivot corners exist, including center-pivot corner systems or lateral irrigation pipes; however, farmers in the Rainwater Basin commonly raise crops on pivot corners without irrigation. Gravity irrigation consists of a temporary lateral irrigation pipe extending along the field edge with the highest altitude and perpendicular to the direction of crop rows. Water is released from the pipe into furrows between crop rows and is moved by gravity toward the opposite end of the field. A dryland field is not irrigated by any means. In years with adequate growing season precipitation, dryland field and pivot corner grain yields are comparable with irrigated croplands; however, in drier years, they tend to yield less.

Rainwater Basin grasslands consist of CRP enrolled grasslands, non-CRP grasslands and wet meadows. CRP grasslands are highly erodible cropland hectares that have been removed from crop production and seeded with nonnative (CP1) or native (CP2) conservation plantings (King & Savidge 1995; USDA – NRCS 2012). CP1 plantings typically consist of smooth brome (*Bromus inermis*) and legumes, whereas native plantings are composed primarily of native tallgrass species like big bluestem (*Andropogon gerardii*), little bluestem (*Schizachyrium scoparium*), switchgrass and Indiangrass (*Sorghastrum nutans*) (Clark et al. 1993; Delisle & Savidge 1997). In this study, no differentiation was made between nonnative and native conservation plantings because it was difficult to differentiate between planting with existing GIS data. Non-CRP grassland includes remnant grasslands, cattle pastures, restored grasslands not affiliated with the CRP, grass buffers surrounding restored wetland sites, and grass linings of road ditches and canals (Utrup & Davis 2007; Ramirez-Yanez et al. 2011). Wet meadows are commonly flooded and hayed riparian grasslands dominated by sedges, rushes and other native mixedgrass prairie species (Currier 1989).

MARGINALLY PRODUCTIVE CROPLANDS

Criteria making rowcrop fields marginally productive and suitable for conversion to switchgrass were based on agricultural suitability of underlying soils, field size and shape complexity, mean annual precipitation, and likelihood of experiencing additional irrigation limitations in the future. Each field was assigned to 1 of 24 marginality classes, based on the number of criteria it satisfied (Table 1). The more criteria a field satisfied the more marginal the cropland for rowcrop production and the more suitable for conversion to switchgrass.

Dryland fields and pivot corners were considered more marginal and suitable for conversion to switchgrass than center-pivot or gravity irrigated fields, due to the lack of irrigation systems on dryland fields and pivot corners and because switchgrass is more drought tolerant and water use efficient than rowcrops (Kiniry et al. 2008). USDA – NRCS land capability classes were used to determine the agricultural suitability of field soils. Soils in classes 1 and 2 are most suitable for agriculture, whereas soils in classes 7 and 8 are completely unsuitable. Soils in classes 3, 4, 5 and 6 can be described as marginally productive, and may be better suited to less intensive forms of agricultural landuse, which could include annual haying of perennial grasses like switchgrass. Switchgrass has been shown to remain productive on poor soils with ethanol yields comparable to or greater than that of combined maize grain and stover (Varvel et al. 2008).

Pivot corners and small, complexly shaped dryland fields were considered more marginal than larger, more uniformly shaped dryland fields. Farming rowcrops on small, complexly shaped fields with increasingly large, modern farm equipment can be inconvenient and time consuming, and these fields could be better suited to raising less management intensive and perennial switchgrass stands (Mitchell et al. 2008). All pivot corners were considered small, as were dryland fields with areas less than the mean pivot corner area (3.7 hectares). Dryland fields with areas greater than 3.7 hectares, but less than the 25th percentile value for dryland field area (4.7 hectares), and with a shape index greater than the 75th percentile value for dryland field shape index (1.56), were considered small and complexly shaped.

Due to the rain shadow effect of the Rocky Mountains, mean annual precipitation increases from west to east throughout the Great Plains (Rickets et al. 1999), making the western half of the Rainwater Basin region drier than the eastern half (Pederson et al. 1989). Fields in areas with a mean annual precipitation of 63.5 centimeters or less were considered more marginal than fields in areas with a mean annual precipitation greater than 63.5 centimeters. Switchgrass is more drought tolerant and water use efficient than rowcrops (Vogel 2004; Kiniry et al. 2008), and therefore may be more feasible to produce than rowcrops on non-irrigated croplands subject to frequent drought.

Center-pivot irrigated and gravity irrigated fields were classified according to their likelihood of experiencing additional irrigation limitations in the future. Fields were assigned to a high risk or low risk classification, based on the Natural Resource District

(NRD) in which they were located. Fields in NRDs with histories of implementing moratoriums or stays on irrigation were classified as being at high risk for additional future irrigation limitations, whereas those without previously implemented moratoriums or stays were classified as being at low risk for future limitations (Kurtz 2007). If NRDs restrict irrigation in the future, switchgrass may be more feasible to produce than annual rowcrops, especially under the warmer and drier conditions that would be expected to force the implementation of irrigation limitations (IPCC 2007; McDonald et al. 2009).

Rowcrop fields classified as most marginal and suitable for conversion to switchgrass under present climatic conditions were pivot corners and small, complexly shaped dryland fields, located on soils in NRCS land capability classes 3, 4, 5 or 6 and in areas with annual average precipitation of 63.5 centimeters or less. Rowcrop fields classified as least marginal and unsuitable for conversion to switchgrass were gravity and pivot irrigated fields, and large, uniformly shaped dryland fields located on soils in NRCS land capability classes 1 or 2, in areas with annual average precipitation greater than 63.5 centimeters and at low risk of experiencing future irrigation limitations. Remaining croplands were placed into intermediate marginality classes according to the number of marginal criteria they satisfied (Table 1).

LANDUSE CHANGE SCENARIOS

I developed three scenarios for assessing potential impacts of future biofuel-based landuse change on Rainwater Basin grassland bird populations. These scenarios encompass a wide range of potential futures for agricultural lands, with major drivers of future change being climate, irrigation limitations, commodity markets and ethanol demand. Interactions between these drivers could determine the future enrollment of marginally productive agricultural lands in rowcrops or bioenergy switchgrass, which in turn will affect the quantity and quality of wildlife habitat.

The Limited Change Scenario assumes minimal climatic changes without any additional irrigation limitations and an increased demand for cellulosic ethanol. This scenario establishes the baseline for biofuel-based landuse change under current climate and policy. In the Limited Change Scenario, agricultural fields converted to switchgrass consist primarily of small, complexly shaped dryland fields and pivot corners located on marginal soils and in areas with mean annual precipitation of 63.5 centimeters or less. Under this scenario, 53,672 rowcrop hectares are converted to switchgrass.

Under the Modest Change Scenario, modest climatic changes occur and are accompanied by irrigation limitations in water stressed regions, along with a greater cellulosic ethanol demand than in the Limited Change Scenario. Under this scenario, the converted proportions of rowcrop fields in marginality classes composed of pivot corners and small, complexly shaped dryland fields located on poor soils were 25% greater than those in the Limited Change Scenario (Table 1). Additionally, between 25% and 75% of fields in selected marginality classes composed of larger dryland fields located on poorer soils and irrigated fields located on poorer soils and in drier areas at higher risk for irrigation limitations are converted from rowcrops to switchgrass. More gravity irrigated fields were converted than pivot irrigated fields, due to the greater water used efficiency of center-pivot irrigation systems. A total of 121,141 rowcrop hectares are converted to switchgrass in this scenario.

The Extreme Change Scenario projects more extreme climatic changes that are accompanied by widespread irrigation limitations and a high demand for cellulosic ethanol. Between 50% and 100% of all pivot corners and dryland fields were converted to switchgrass, depending on soil type, field size and shape and mean annual precipitation (Table 1). The proportion of some gravity irrigated and pivot irrigated fields in areas at higher risk for additional future irrigation limitations converted to switchgrass was 25% greater than in the Modest Change Scenario. More gravity irrigated fields were converted than pivot irrigated fields, due to the greater water used efficiency of center-pivot irrigation systems. Under the Extreme Change Scenario, 208,827 rowcrop hectares are converted to switchgrass.

CROPLAND CONVERSION

The geostatistical analyst extension in ArcGIS was used to assign percentages of rowcrop fields in different marginality classes to be converted to switchgrass under the three landuse change scenarios. I used a conservative approach to determine rowcrop fields that could be converted to switchgrass under the Limited Change Scenario, and increased conversion percentages under the following two scenarios. Although many fields possess marginal characteristics, not all fields are expected to be converted to switchgrass, because conventional crop production on marginal soils can provide justifiable economic returns in some years. I assigned greater conversion percentages to classes satisfying more marginal criteria. However, only classes satisfying all marginal criteria had 100% of fields converted to switchgrass under the Limited Change Scenario. Classes satisfying fewer marginal criteria were assigned lower switchgrass conversion percentages of 75, 50, 25 or 0 (Table 1). Fields not assigned to switchgrass conversion remained in rowcrop production. Conversion percentages increased under the Modest Change and Extreme Change Scenarios, which assumed greater climate change, more irrigation limitations and greater cellulosic ethanol demand.

LANDCOVER AREA

Total switchgrass area shapefiles from the three landuse change scenarios and the total current cropland shapefile were converted from vector to raster format with the polygon to raster tool, reclassified into single classes, and input into the program Fragstats to calculate total area. To minimize errors in area calculations due to the conversion of shapefiles to raster format, remaining rowcrop areas following conversion to switchgrass under each landuse change scenario were calculated by subtracting converted switchgrass area from current total rowcrop area. CRP grassland, non-CRP grassland and wet meadow areas were obtained directly from the 40 kilometer ethanol plant service area Rainwater Basin landcover. CRP grassland, non-CRP grassland and wet meadow areas were calculated by multiplying the number of raster cells occupied by each landcover type by 900 m^2, the area covered by a single raster cell.

GRASSLAND BIRD ABUNDANCE

A customized version of the HABS (Hierarchical All Birds Strategy) model (PLJV 2007) was used to predict current abundances for a suite of Rainwater Basin grassland bird species within 40 kilometer ethanol plant service areas, in addition to changes in abundance under each of the three landuse change scenarios. HABS is a hierarchically organized database that links bird conservation regions (BCRs) within the

PLJV (Playa Lakes Joint Venture) with different landuse associations. The PLJV covers nearly 777,000 km^2 in portions of seven Great Plains states. In Nebraska, PLJV BCRs consist of prairie potholes, shortgrass prairie, mixedgrass prairie and tallgrass prairie (PLJV 2007). The Rainwater Basin region lies almost entirely within the mixedgrass prairie region of Nebraska (BCR 19 – Nebraska), and all HABS model runs in this study are conducted for BCR 19 – Nebraska.

Landuse associations are general forms of landuse within PLJV BCRs. For different landuse associations, various habitat conditions exist. Maize is an example of a habitat conditions housed within the cropland landuse association. With the assistance of the Nebraska Bird Partnership, I customized a bioenergy switchgrass habitat condition within the cropland landuse association. Existing scientific literature was used to populate individual habitat conditions with bird density estimates for the breeding or non-breeding seasons (Table 2). Within HABS, the breeding season abundance estimate for an individual species in a single landuse condition was calculated by multiplying the following factors: the bird density estimate within the habitat condition, total number of hectares enrolled in the habitat condition, the proportion of hectares comprising the habitat condition available as habitat for the species, the proportion of hectares comprising the habitat condition considered suitable habitat for the species, and a proportion indicating how minimum area requirements may limit the utilization of the habitat condition by the species (PLJV 2007). Bird abundances from each habitat condition are then summed to generate an abundance estimate for the species across the landscape. In this study, I considered all habitat conditions 100% available and 100% suitable for all species, and did not factor in minimum area requirements. These HABS model simplifications were appropriate for developing a consistent approach for assessing species responses under different landuse change scenarios.

I used HABS to predict overall abundances and changes in abundance for the following avian species under three scenarios of landuse change: bobolink (*Dolichonyx oryzivorus*), dickcissel (*Spiza americana*), eastern kingbird (*Tyrannus tyrannus*), field sparrow (*Spizella pusilla*), grasshopper sparrow (*Ammodramus savannarum*), meadowlark (*Sturnella spp.*), ring-necked pheasant (*Phasianus colchicus*), sedge wren (*Cistothorus plantensis*) and upland sandpiper (*Bartramia longicauda*). No distinction was made between eastern meadowlarks (*Sturnella magna*) and western meadowlarks (*Sturnella neglecta*), because meadowlark density estimates were obtained from Murray & Best (2003), who did not differentiate between species in their survey. Some studies used to populate other landuse conditions did differentiate between meadowlark species (Faanes & Lingle 1995; Kim et al. 2008). In these instances, western meadowlark densities were used, since they were considered the more common of the meadowlark species in the Rainwater Basin region (Rosenberg 2004).

Not all grassland bird density estimates utilized for predicting present and future in this study were collected in the study area. Grassland bird surveys in switchgrass stands managed as bioenergy crops are limited, and no surveys have been conducted in the Rainwater Basin or State of Nebraska to date. All switchgrass bird density estimates utilized in this study are from Murray and Best (2003), who surveyed grassland birds in bioenergy switchgrass stands in the Chariton Valley Region of southern Iowa, U.S.A. The Chariton Valley is a landscape characterized by rolling topography, farming of annual rowcrops, and prevalence of native and restored grasslands. This landscape differs from the Rainwater Basin, primarily in the greater proportion of the Chariton Valley landscape that remains in some form of grassland (Murray & Best 2003). Similarly, density estimates for bobolinks and sedge wrens in annual rowcrops were obtained from Johnson and Igl (1995), who conducted their study in North Dakota, where CRP grassland comprised a considerable portion of the landscape. In addition to changes in landcover type, differences in landscape factors and regional climate could drive differences in grassland bird abundances between landscapes.

Lower and upper grassland bird density estimates for each species in each habitat condition were calculated by multiplying the mean observed density for each species in each habitat condition by the standard error of the mean density estimate for the considered species from Murray and Best (2003), and then subtracting and adding the result from the mean density estimate. Murray and Best (2003) standard errors were used for all species in all habitat conditions because some studies from which density estimates were obtained did not include standard errors. Lower and upper density estimates for each species in each habitat condition were each input into lower and upper copies of the HABS model, respectively. As with mean abundance estimates, lower and upper confidence intervals for individual species abundances under different landuse change scenarios were calculated by multiplying lower and upper density estimates for each habitat condition by the number of hectares enrolled in the respective habitat condition under the considered scenario, and then summing the appropriate lower or upper abundance estimates from all habitat conditions. Percent changes in bird abundance for each scenario were determined by calculating the difference between current mean abundance and the newly generated mean abundance under the scenario, dividing the difference by the current bird abundance, and multiplying the result by 100. Percent changes in abundance were useful for assessing and comparing the potentials of different landuse change scenarios to impact bird abundances.

RESULTS

Within the 40 kilometer ethanol plant service areas of the Rainwater Basin, rowcrops dominate landuse, occupying 1,010,180 hectares, or 74% of total land area. Non-CRP grassland covers 188,930 hectares; wet meadows 13,718 hectares; and CRP grassland 2,583 hectares. Together, these agriculturally and economically important landcover classes account for approximately 90% of the total 1,357,850 Rainwater Basin hectares within the 40 kilometer ethanol plant service areas. Grasshopper sparrows, bobolinks, dickcissels and meadowlarks were predicted to be the most abundant species in HABS models (Table 3).

Under the Limited Change Scenario, which assumed no additional climatic changes or irrigation limitations, there were 53,672 marginally productive rowcrop hectares converted to switchgrass, with 956,509 hectares remaining in rowcrop production, 2,583 hectares in CRP grasslands, 188,930 hectares in non-CRP grasslands, and 13,718 hectares in wet meadows. In this scenario, the conversion of rowcrops to switchgrass positively impacted a variety of grassland bird species, most notably sedge wrens, which exhibited a 34–55% increase in abundance (Table 4). Grasshopper sparrows, ring-necked pheasants, meadowlarks and dickcissels also increased, but less than sedge wrens (Figure 6). Field sparrows, eastern kingbirds and upland sandpipers increased to even lesser degrees, and bobolink abundance decreased slightly (Table 4).

In the Modest Change Scenario, with modest climatic change and some limited irrigation limitations, there were 121,141 marginally productive rowcrop hectares converted to switchgrass, and 889,039 hectares remained in rowcrop production, 2,583 hectares in CRP grassland, 188,930 hectares in non-CRP grassland, and 13,718 hectares in wet meadow grassland. In this scenario, the magnitude of avian responses was greater than in the Limited Change Scenario, due to the additional 67,470 hectares being converted to switchgrass. Sedge wrens had the greatest percent change in abundance, followed by grasshopper sparrows, ring-necked pheasants, meadowlarks, dickcissels, eastern kingbirds and upland sandpipers (Figure 7). Bobolinks exhibited a slightly negative percent change in abundance (Table 4).

Under the Extreme Change Scenario, with extreme climatic changes and more widespread irrigation limitations, there were 208,827 marginally productive rowcrop hectares converted switchgrass, leaving 801,354 hectares in rowcrop production, 2,583 hectares in CRP grassland, 188,930 hectares in non-CRP grass, and 13,718 hectares in wet meadow grassland. Sedge wren abundance increased between 135% and 213%, grasshopper sparrow and ring-necked pheasant increases ranged between 50% and 100%, and meadowlarks and dickcissels increased between 30% and 50%. Eastern kingbirds increased by 17%, while upland sandpipers and bobolinks showed slightly positive and negative responses, respectively (Table 4).

DISCUSSION

Predicted avian responses to landuse change scenarios varied by species; however, the overall impact of converting rowcrops to switchgrass on the grassland bird community was positive. Under the Limited Change Scenario, which does not factor in climatic changes or irrigation limitations, over 53,600 hectares of marginally productive cropland were identified. These hectares are currently in intensive rowcrop agriculture, but may be better suited to other forms of landuse. The enrollment of these hectares in CRP or other conservation programs could produce the greatest ecological and environmental benefits, but farmers generally engage in landuses that secure the greatest profit, and presently high grain market prices encourage rowcrop production. However, increasing cellulosic ethanol demand and government support for cellulosic ethanol production, in addition to changes in climate, could make the conversion of marginally productive croplands to switchgrass more appealing. Switchgrass stands are expected to support greater densities of birds than rowcrops; and therefore, the conversion of rowcrops to switchgrass represents an economically feasible form of landuse change that could benefit farmers and avifauna.

Species that responded most positively to the conversion of rowcrops to switchgrass were sedge wrens, grasshopper sparrows, ring-necked pheasants and meadowlarks (Figure 8). Dickcissels, eastern kingbirds, field sparrows and upland sandpipers also increased, but to lesser degrees. Only bobolinks responded negatively to the conversion to switchgrass, and percent decreases in abundance were slight (Table 4). Individual species habitat preferences are important for determining responses to switchgrass implementation, and the floral and structural diversity of switchgrass stands will depend heavily on management (Murray & Best 2003).

The timing of annual switchgrass harvest could influence switchgrass stand vegetative structure (Murray & Best 2003). If switchgrass haying is conducted after the first killing frost, stands will not regrow over winter, and residual cover may be short and sparse during the following year's bird breeding season, benefitting species like grasshopper sparrows (Skinner 1975; Murray & Best 2003; Johnsgard 2009). However, if switchgrass harvest is conducted at anthesis in late summer or early autumn, switchgrass stands will have time to regrow following harvest, and vegetation may be tall and dense the following year, benefitting sedge wrens and other species that prefer tall, dense vegetative structure (Skinner 1975; Delisle & Savidge 1997; Renfrew & Ribic 2002; Johnsgard 2009). Switchgrass regrowth following harvest could also be important for early nesting and resident species like ring-necked pheasants, which rely on the presence of residual vegetation for winter cover and nest construction the following season (Haensly et al. 1987; Delisle & Savidge 1997; Murray & Best 2003).

Fertilizer and herbicide applications will likely influence avian use of switchgrass stands. Annual fertilizer application could cause stands to grow taller and denser than they would naturally, potentially amplifying or offsetting the effects of switchgrass harvest timing on vegetative structure and avian species. Fertilizer application could positively impact species preferring tall, dense vegetation structure, but is not likely to benefit species associated with short, sparse grassland habitat. Herbicide application could decrease the floral diversity present in switchgrass stands, which might negatively impact species like dickcissels, which prefer patchy vegetation with tall forb species for nesting, perching and singing (Skinner 1975; Delisle & Savidge 1997; Johnsgard 2009). Meadowlarks also prefer vegetative habitat with elevated singing perches (Weins 1969; Skinner 1975; Renfrew & Ribic 2002), which may not be as readily available in switchgrass monocultures as in native or restored grasslands. In the future, improved switchgrass hybrids could also increase the height and density of switchgrass stands (Vogel & Mitchell 2008), which may be beneficial or detrimental to different avian species.

Bioenergy switchgrass may benefit grassland birds most in intensively cultivated landscapes like the Rainwater Basin, since the greatest potential for change in these landscapes exists in the conversion of rowcrops to switchgrass. However, landscapes with higher proportions of more marginally productive agricultural lands already enrolled in grazing or the CRP might be negatively impacted by the conversion to switchgrass if the adoption of switchgrass as a bioenergy crop results in the conversion of high diversity grassland to switchgrass stands. Species responses were amplified from the Limited Change Scenario to the Modest Change Scenario and from the Modest Change Scenario to the Extreme Change Scenario, due to the increasing number of rowcrop hectares converted to switchgrass under scenarios that assumed greater intensities of climate change, irrigation limitation and cellulosic ethanol demand. The limited responses to landuse change scenarios exhibited by upland sandpipers, field sparrows and eastern kingbirds could indicate their reliance on habitats other than those found in annual rowcrops or switchgrass stands. Conserving and restoring CRP grassland, non-CRP grassland and wet meadows might be most important for these species. Limited bobolink responses resulted from the similarity between bobolink density estimates in rowcrops and switchgrass stands (Table 3).

In this study, landuse change intensity was positively correlated with the magnitude of future climate change; however, the ways in which climate change might directly impact grassland birds is not addressed. Furthermore, species minimum area requirements, sensitivity to habitat fragmentation, proximity to water sources in lowland areas and other local and landscape metrics that may influence avian utilization of agricultural lands are not accounted for. Results should be interpreted carefully and generally in this context. The continuation of the CRP, in coordination with the conversion of marginally productive croplands to switchgrass, could benefit grassland bird populations. However, if both marginally productive croplands and CRP grasslands are converted to switchgrass, the habitat improvements associated with the conversion of rowcrops to switchgrass might be offset by the conversion of CRP grasslands to switchgrass. Focusing future management efforts on preserving restored grasslands and converting rowcrops to switchgrass or CRP grassland could strongly benefit grassland bird populations in agricultural regions.

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TABLES AND FIGURES

Table 1: List of 24 Rainwater Basin rowcrop field marginality classes and percentages of marginality classes converted to switchgrass under three landuse change scenarios. Conversion percentages of 0, 25, 50, 75 or 100 were assigned to marginality classes, with higher conversion percentages being assigned to classes satisfying more marginal criteria. The intensity of climatic change and accompanying irrigation limitations increases from the Limited Change Scenario to the Modest Change Scenario and from the Modest Change Scenario to the Extreme Change Scenario.

Landuse classification	**Limited**	**Modest**	**Extreme**
Pivot corners + poor soils + dry area	100%	100%	100%
Pivot corners + poor soils + wet area	75%	100%	100%
Pivot corners + good soils + dry area	75%	100%	100%
Pivot corners + good soils + wet area	50%	75%	100%
Small dryland fields + poor soils + dry area	100%	100%	100%
Small dryland fields + poor soils + wet area	75%	100%	100%
Small dryland fields + good soils + dry area	75%	100%	100%
Small dryland fields + good soils + wet area	50%	75%	100%
Large dryland fields + poor soils + dry area	25%	50%	75%
Large dryland fields + poor soils + wet area	0%	25%	50%
Large dryland fields + good soils + dry area	0%	25%	50%
Large dryland fields + good soils + wet area	0%	0%	25%
Gravity + poor soils + dry area + high risk of irrigation limitations	0%	50%	75%
Gravity + poor soils + wet area + high risk of irrigation limitations	0%	25%	50%
Gravity + good soils + dry area + high risk of irrigation limitations	0%	25%	50%
Gravity + good soils + wet area + high risk of irrigation limitations	0%	0%	25%
Gravity + poor soils + wet area + low risk of irrigation limitations	0%	0%	25%
Gravity + good soils + wet area + low risk of irrigation limitations	0%	0%	0%

Table 1 continued.

Landuse classification	**Limited**	**Modest**	**Extreme**
Pivots + poor soils + dry area + high risk of irrigation limitations	0%	25%	25%
Pivots + poor soils + wet area + high risk of irrigation limitations	0%	0%	25%
Pivots + good soils + dry area + high risk of irrigation limitations	0%	0%	0%
Pivots + good soils + wet area + high risk of irrigation limitations	0%	0%	0%
Pivots + poor soils + dry area + low risk of irrigation limitations	0%	0%	0%
Pivots + good soils + wet area + low risk of irrigation limitations	0%	0%	0%

Table 2: Grassland bird densities in bioenergy switchgrass, rowcrops, CRP grassland, non-CRP grassland and wet meadows input into the HABS model and used to predict current Rainwater Basin grassland bird abundances and changes in abundance under three landuse change scenarios involving the conversion of rowcrops to bioenergy switchgrass.

Name	**Landcover type**	**Reference**	**Density (birds/hectare)**
Bobolink	Switchgrass	Murray & Best 2003	0.0240
Dickcissel	Switchgrass	Murray & Best 2003	0.0751
Eastern kingbird	Switchgrass	Murray & Best 2003	0.0200
Field sparrow	Switchgrass	Murray & Best 2003	0.0089
Grasshopper sparrow	Switchgrass	Murray & Best 2003	0.3509
Meadowlark	Switchgrass	Murray & Best 2003	0.0499
Ring-necked pheasant	Switchgrass	Murray & Best 2003	0.0309
Sedge wren	Switchgrass	Murray & Best 2003	0.0739
Upland sandpiper	Switchgrass	Murray & Best 2003	0.0069
Bobolink	Rowcrops	Johnson & Igl 1995	0.0245
Dickcissel	Rowcrops	Best et al. 1997	0.0079
Eastern kingbird	Rowcrops	Best et al. 1997	0.0079
Field sparrow	Rowcrops	Best et al. 1997	0.0020
Grasshopper sparrow	Rowcrops	Best et al. 1997	0.0040
Meadowlark	Rowcrops	Best et al. 1997	0.0040
Ring-necked pheasant	Rowcrops	Best et al. 1997	0.0059
Sedge wren	Rowcrops	Johnson & Igl 1995	0.0000
Upland sandpiper	Rowcrops	Best et al. 1997	0.0040
Bobolink	CRP grassland	Delisle & Savidge 1997	0.1295
Dickcissel	CRP grassland	Delisle & Savidge 1997	1.6741
Eastern kingbird	CRP grassland	Delisle 1995	0.0146
Field sparrow	CRP grassland	Delisle 1995	0.0054
Grasshopper sparrow	CRP grassland	Delisle & Savidge 1997	0.5211
Meadowlark	CRP grassland	Delisle & Savidge 1997	0.0694

Table 2 continued.

Name	**Landcover type**	**Reference**	**Density (birds/hectare)**
Ring-necked	CRP grassland	Delisle & Savidge 1997	0.0591
Sedge wren	CRP grassland	Delisle & Savidge 1997	0.1376
Upland sandpiper	CRP grassland	Delisle 1995	0.0030
Bobolink	Non-CRP grassland	Faanes & Lingle 1995	0.0541
Dickcissel	Non-CRP grassland	Faanes & Lingle 1995	0.0640
Eastern kingbird	Non-CRP grassland	Faanes & Lingle 1995	0.0299
Field sparrow	Non-CRP grassland	Faanes & Lingle 1995	0.0200
Grasshopper	Non-CRP grassland	Faanes & Lingle 1995	0.3600
Meadowlark	Non-CRP grassland	Faanes & Lingle 1995	0.3800
Ring-necked	Non-CRP grassland	Faanes & Lingle 1995	0.0040
Sedge wren	Non-CRP grassland	Utrup & Davis 2007	0.0334
Upland sandpiper	Non-CRP grassland	Faanes & Lingle 1995	0.0400
Bobolink	Wet meadow	Kim et al. 2008	1.0645
Dickcissel	Wet meadow	Kim et al. 2008	0.4302
Eastern kingbird	Wet meadow	Faanes & Lingle 1995	0.0739
Field sparrow	Wet meadow	Faanes & Lingle 1995	0.0000
Grasshopper	Wet meadow	Kim et al. 2008	0.1843
Meadowlark	Wet meadow	Kim et al. 2008	0.2219
Ring-necked	Wet meadow	Kim et al. 2008	0.0000
Sedge wren	Wet meadow	Kim et al. 2008	0.0418
Upland sandpiper	Wet meadow	Kim et al. 2008	0.2511

Table 3: Predicted lower and upper confidence interval bounds for current Rainwater Basin grassland bird abundances within 40 kilometer ethanol plant service areas.

Species	**Abundance (lower)**	**Abundance (upper)**
Bobolink	32,754.87	66,993.04
Dickcissel	16,181.19	66,645.93
Eastern kingbird	7,479.99	19,655.15
Field sparrow	3,038.40	10,597.91
Grasshopper sparrow	55,059.01	175,601.05
Meadowlark	15,255.94	34,968.18
Ring-necked pheasant	129.57	17,702.48
Sedge wren	1,703.17	39,967.46
Upland sandpiper	10,005.51	21,015.31

Table 4: Predicted lower and upper confidence interval bounds for percent changes in Rainwater Basin grassland bird abundances within 40 kilometer ethanol plant service areas under three biofuel-based landuse change scenarios. The Limited Change Scenario assumes no climatic changes or irrigation limitations, the Modest Change Scenario assumes some climate changes and accompanying irrigation limitations, and the Extreme Change Scenario assumes extreme climatic change and widespread irrigation limitations.

Species	Scenario	Percent change (lower)	Percent change (upper)
Bobolink	Limited Change	-0.05%	-0.05%
Dickcissel	Limited Change	8.01%	11.90%
Eastern kingbird	Limited Change	4.42%	4.42%
Field sparrow	Limited Change	4.58%	6.41%
Grasshopper sparrow	Limited Change	19.01%	24.54%
Meadowlark	Limited Change	9.61%	12.67%
Ring-necked pheasant	Limited Change	17.13%	19.44%
Sedge wren	Limited Change	34.85%	54.84%
Upland sandpiper	Limited Change	0.71%	1.06%
Bobolink	Modest Change	-0.12%	-0.12%
Dickcissel	Modest Change	18.08%	26.87%
Eastern kingbird	Modest Change	9.99%	9.99%
Field sparrow	Modest Change	10.34%	14.47%
Grasshopper sparrow	Modest Change	42.92%	55.38%
Meadowlark	Modest Change	21.68%	28.60%
Ring-necked pheasant	Modest Change	38.67%	43.88%
Sedge wren	Modest Change	78.66%	123.78%
Upland sandpiper	Modest Change	1.60%	2.39%
Bobolink	Extreme Change	-0.21%	-0.21%
Dickcissel	Extreme Change	31.16%	46.31%

Table 4 continued.

Species	Scenario	Percent change (lower)	Percent change (upper)
Eastern kingbird	Extreme Change	17.21%	17.21%
Field sparrow	Extreme Change	17.82%	24.94%
Grasshopper sparrow	Extreme Change	73.98%	95.47%
Meadowlark	Extreme Change	37.37%	49.30%
Ring-necked pheasant	Extreme Change	66.65%	75.64%
Sedge wren	Extreme Change	135.59%	213.38%
Upland sandpiper	Extreme Change	2.75%	4.13%

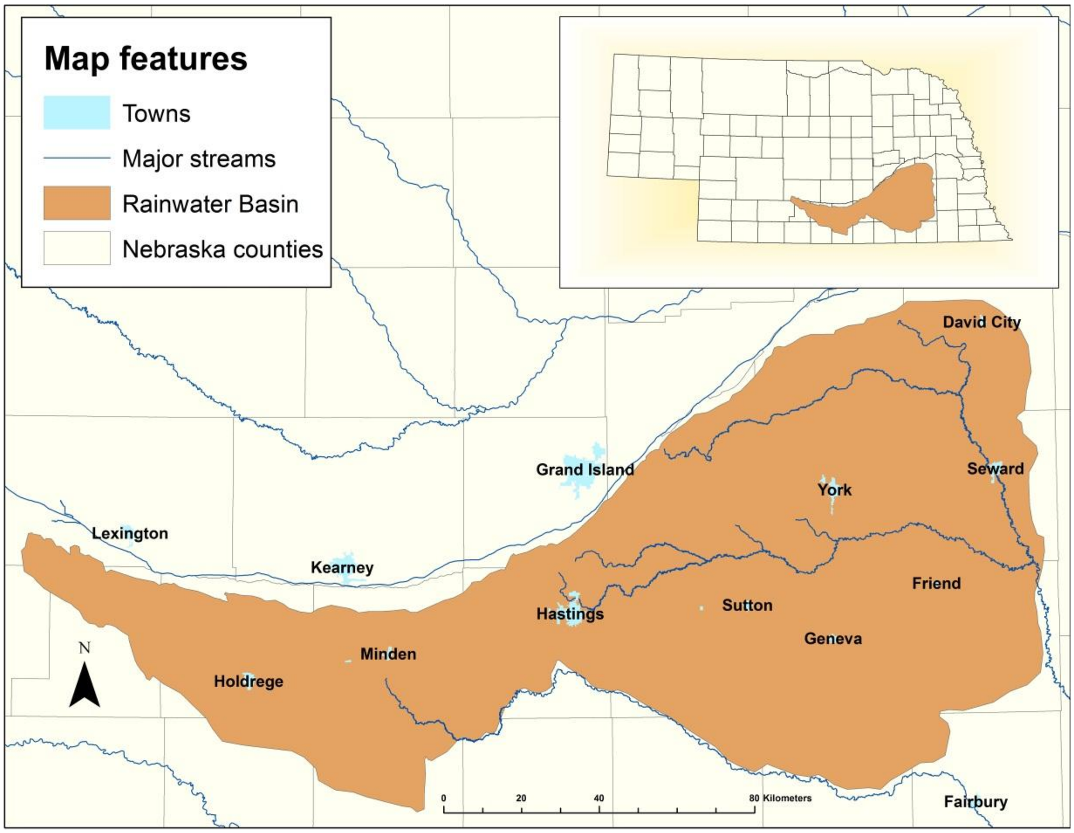

Figure 1: Location of the Rainwater Basin in south-central Nebraska, U.S.A, displaying Nebraska counties, major towns and rivers.

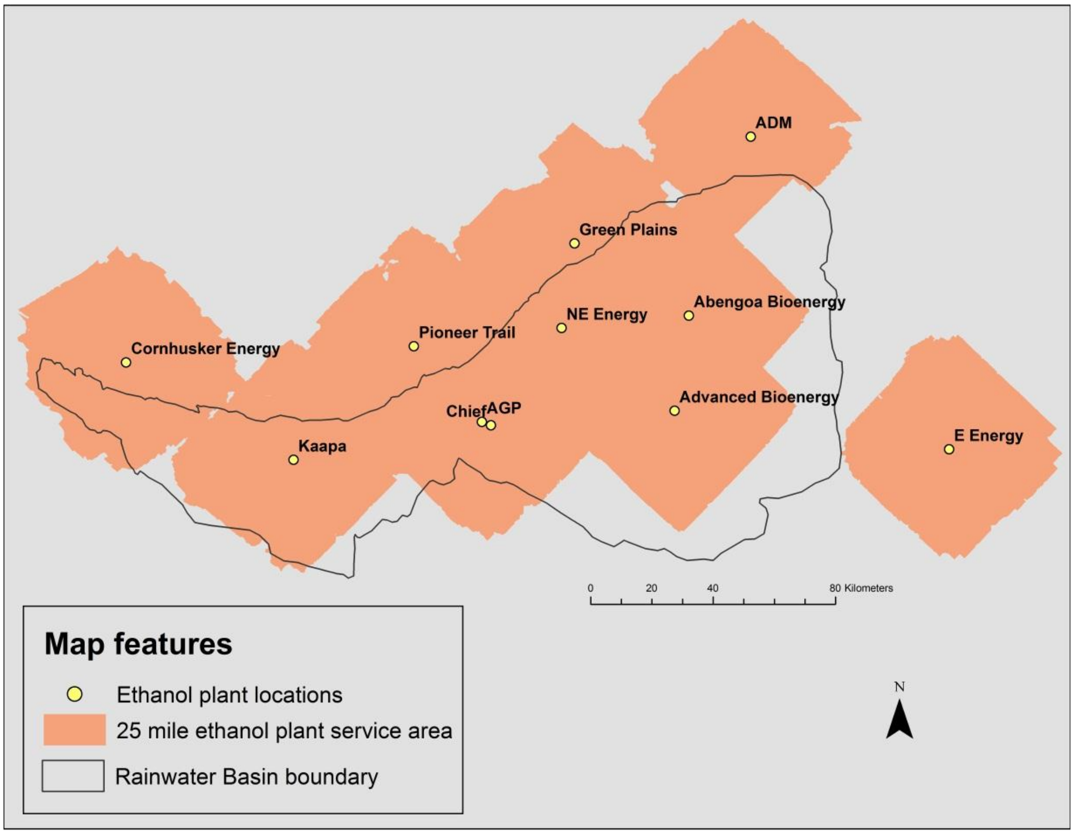

Figure 2: Locations and 40 kilometer network service areas of existing starch-based ethanol plants currently servicing the Rainwater Basin region.

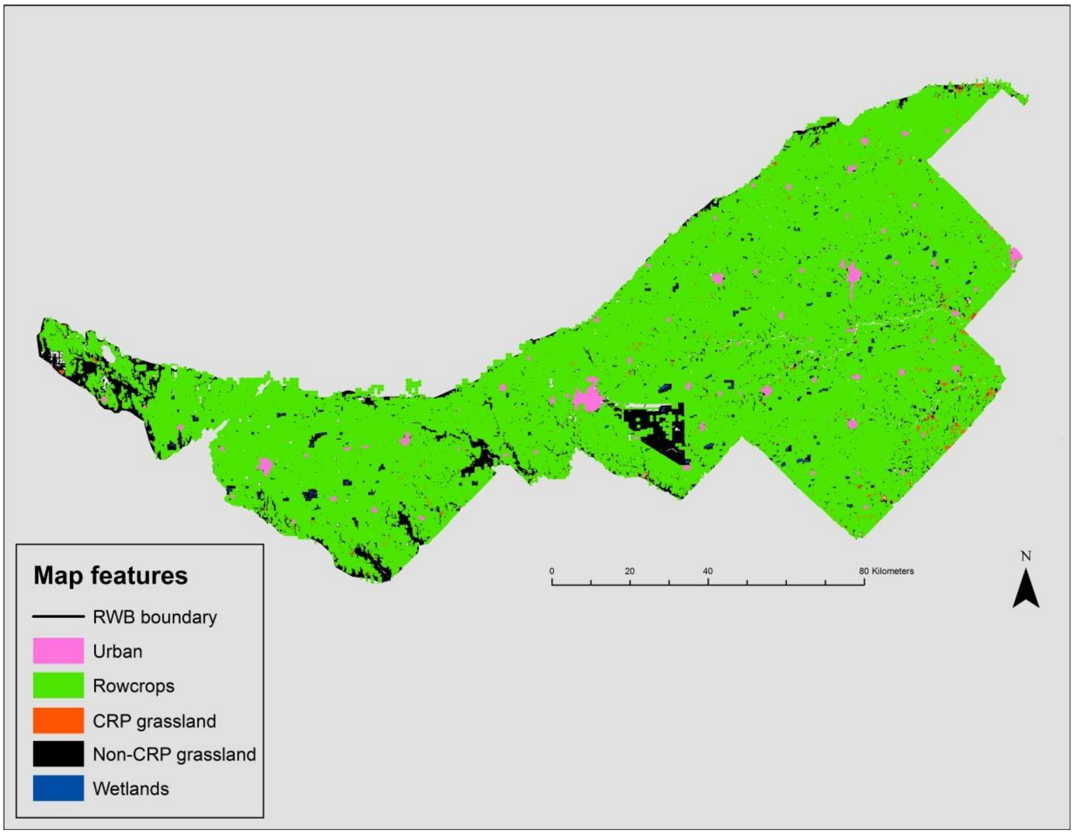

Figure 3: Major Rainwater Basin landcover classes within 40 kilometer road network service areas of existing starch-based ethanol plants. Cropland area is the aggregation of all irrigated and non-irrigated rowcrop fields from Rainwater Basin Joint Venture (RWBJV) 2006 agricultural irrigation type data. Urban areas were derived from Nebraska Department of Natural Resources political boundaries data, and grasslands and wetlands were extracted from 2010 RWBJV landcover.

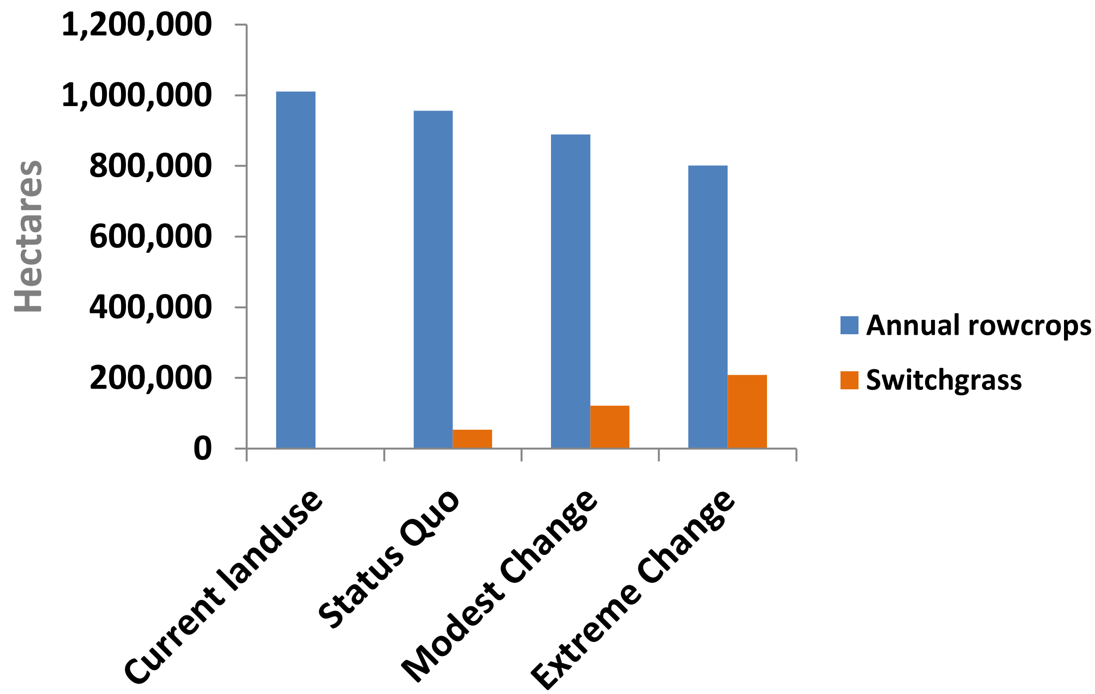

Figure 4: Number of Rainwater Basin hectares within 40 kilometer ethanol plant service areas enrolled in rowcrop and switchgrass production under current landuse and three landuse change scenarios. The intensity of climatic change and accompanying irrigation limitations increases from the Limited Change Scenario to the Modest Change Scenario and from the Modest Change Scenario to the Extreme Change Scenario.

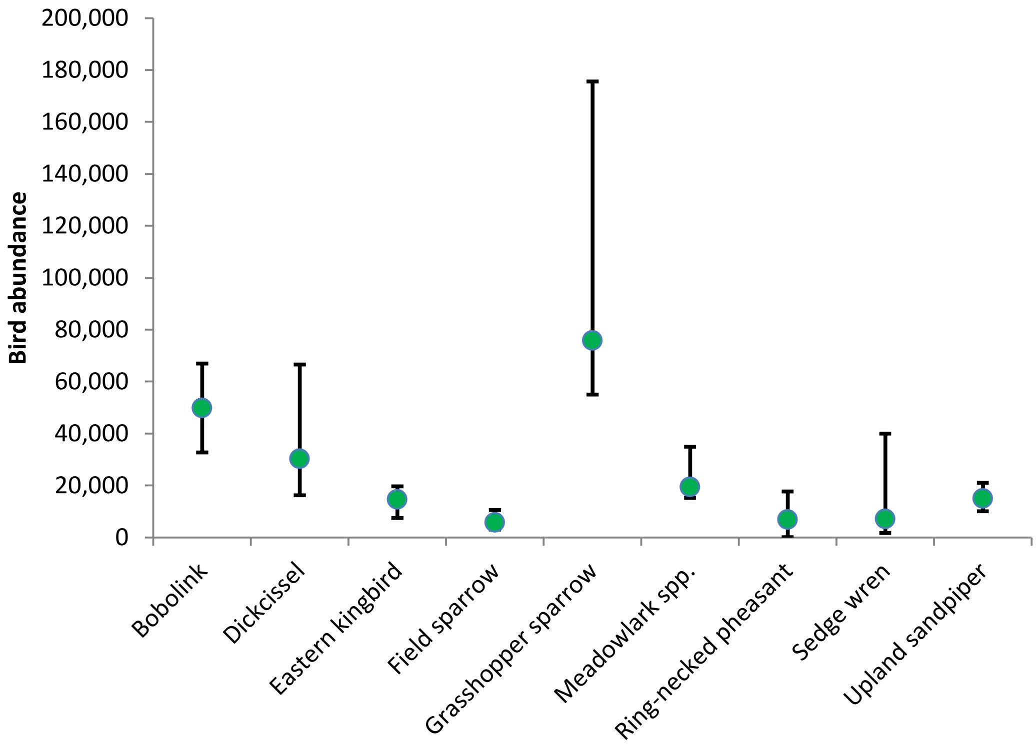

Figure 5: Current predicted Rainwater Basin grassland bird abundances within 40 km of existing starch–based ethanol plants. Green dots represent mean abundance estimates and vertical bars represent the confidence intervals on abundance estimates. Grasshopper sparrows, bobolinks and dickcissels were predicted as the most abundant species.

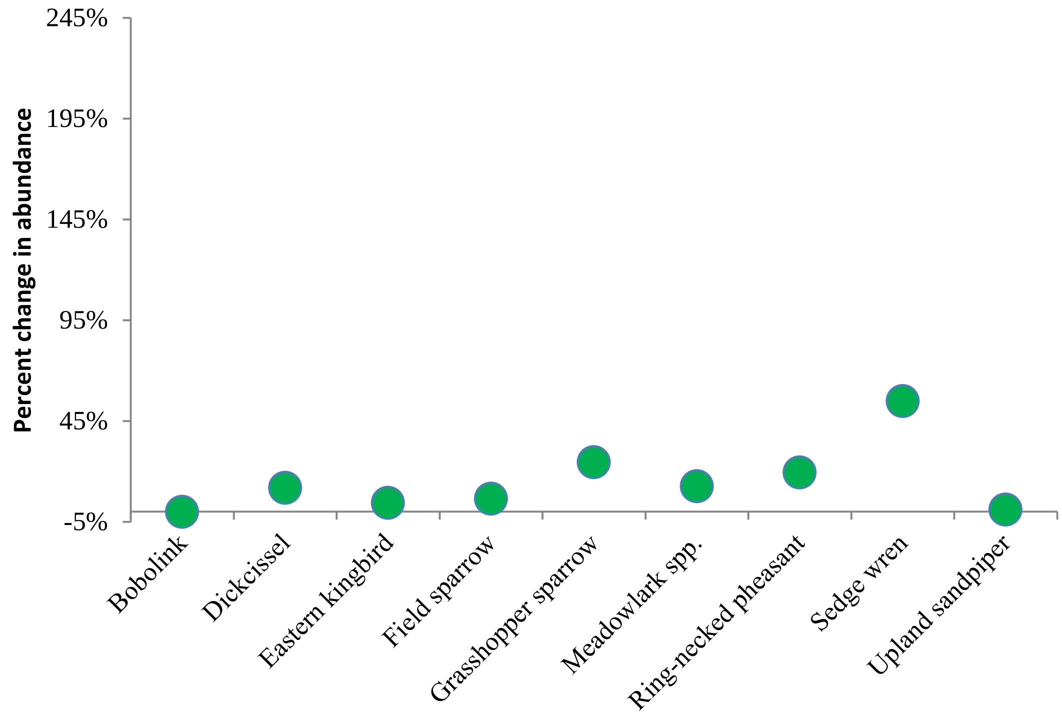

Figure 6: Percent changes in mean grassland bird abundances following the conversion of 53,672 rowcrop hectares to switchgrass under the Limited Change Scenario, which assumes minimal future climatic changes and irrigation limitations.

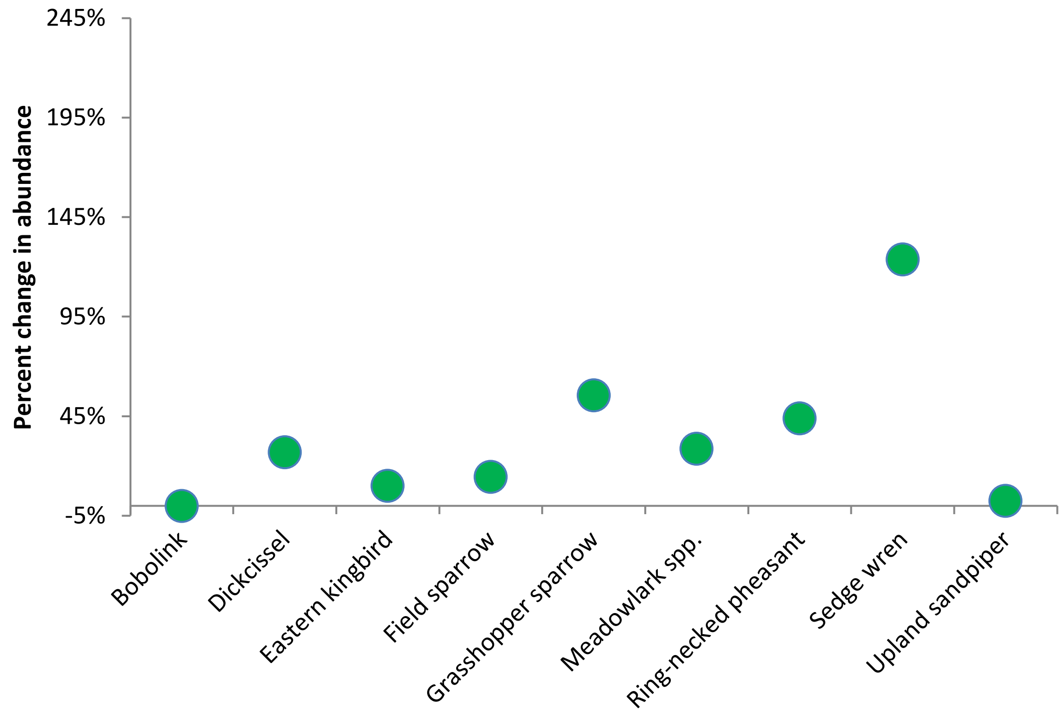

Figure 7: Percent changes in mean grassland bird abundances following the conversion of 121,141 rowcrop hectares to switchgrass under the Modest Change Scenario, which assumes a moderate degree of future climatic changes and irrigation limitations.

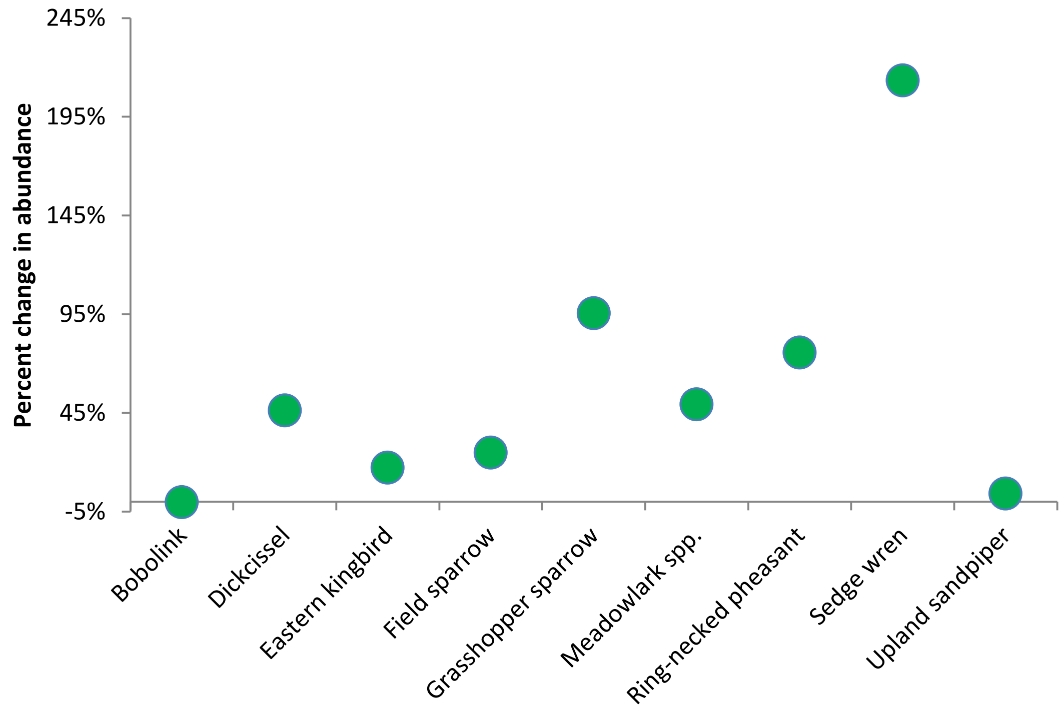

Figure 8: Percent changes in mean grassland bird abundances following the conversion of 208,827 rowcrop hectares to switchgrass under the Extreme Change Scenario, which assumes a high intensity of future climatic changes and widespread irrigation limitations.

CHAPTER 4: RAINWATER BASIN GRASSLAND BIRD RESPONSES TO SCENARIOS OF CHANGE IN CONSERVATION RESERVE PROGRAM GRASSLAND AREA

ABSTRACT

Since 19th Century European settlement, the conversion of native prairie to agriculture has significantly decreased North American grassland bird habitat, but the enrollment of marginally productive croplands in the Conservation Reserve Program (CRP) has compensated for some of the loss. In the future, perennial biofuel crops could further transform agricultural and prairie landscapes. Switchgrass (*Panicum virgatum*) is an alternative biofuel feedstock that may be environmentally and economically superior to maize (*Zea mays*) grain for ethanol production on marginally productive agricultural lands. It is unclear how future conversions between rowcrops, switchgrass and CRP grassland on marginally productive lands might impact grassland bird populations. To explore potential impacts, I developed three agricultural landuse change scenarios for the Rainwater Basin region of Nebraska, USA, driven by potential future cellulosic ethanol demand, grain market prices and continuation of the CRP. For each scenario, I generated spatially explicit maps of agricultural landcover and calculated changes in area of landcover classes. Changes in area were input into the Hierarchical All Birds Strategy (HABS) model to predict changes in avian abundances. Abundances of six species decreased following the conversion of CRP grassland to switchgrass, whereas eight species responded negatively to the replacement of CRP grassland with rowcrops. Alternatively, eight species exhibited positive responses to the conversion of rowcrops to

CRP grassland. Dickcissels, sedge wrens, ring–necked pheasants and grasshopper sparrows responded most negatively to losses in CRP grassland area and most positively to additions to it. CRP provides crucial habitat for grassland bird populations and converting CRP grassland to alternative forms of agricultural landuse could be detrimental to grassland birds, especially in intensively cultivated landscapes with little remaining high diversity grassland.

INTRODUCTION

Since 19th Century European settlement, intensive agricultural production has replaced native grasslands in the North American Great Plains (Weaver 1968; Barker & Whitman 1988; Knopf 1994). Approximately 99% of the Northern Great Plains is either farmed or grazed by livestock (Forrest et al. 2004), and remaining grasslands are fragmented by woody vegetation (Grant et al. 2004), croplands and roads (White et al. 2000). Major crops grown in the Great Plains include maize (*Zea mays*), soybeans (*Glycine max*), wheat (*Triticum aestivum*), milo (*Sorghum bicolor*) and alfalfa (*Medicago sativa*) (Knopf 1994; Mitchell et al. 2010). North American mixedgrass prairie is an endangered ecosystem type and tallgrass prairie is critically endangered (Ricketts et al. 1999; Samson et al. 2004). In addition to prairie remnants, North American grasslands include rangelands, grassland buffers along water bodies and roads, and Conservation Reserve Program (CRP) grasslands (Delisle & Savidge 1997; Utrup & Davis 2007). The CRP provides landowners with monetary incentives for removing highly erodible croplands from rowcrop production and seeding them with conservation plantings, which in addition to reducing soil erosion, benefits water resources and wildlife (Ribaudo 1989; Dunn et al. 1993; USDA – NRCS 2012). In the future, food and bioenergy demands could drive additional landuse conversions among rowcrops, bioenergy crops, and conservation plantings (Tilman et al. 2002).

Grassland birds have been negatively impacted by reductions and fragmentation of native grasslands over the past two centuries (Herkert 1994; Samson & Knopf 1994; White et al. 2000; Grant et al. 2004). Grassland birds have experienced greater, faster and more widespread declines than any other ecological guild on the North American continent (Herkert et al. 1993; Knopf 1994; Sauer et al. 2011). Today, remnant and restored grasslands provide grassland birds with crucial feeding and breeding habitat (Johnson & Schwartz 1993; Johnson & Igl 1995; Ramirez-Yanez et al. 2011).

Demand for clean, renewable energy has resulted in recent, large scale efforts to utilize maize grain for ethanol production (Schnoor et al. 2008). Despite extensive development of the starch-based ethanol industry, ethanol production from maize grain remains controversial, due to uncertainties over its net energy production, water use efficiency, ability to reduce atmospheric greenhouse gas emissions, and competition with food production for landuse (Berndes 2008; Searchinger et al. 2008; Tilman et al. 2009; Dale et al. 2010). This has led to an increasing promotion of the benefits of second generation biofuels (Tilman et al. 2009). One second generation alternative biofuel feedstock proposed for large scale cellulosic ethanol production in Great Plains agricultural landscapes is bioenergy switchgrass (*Panicum virgatum*) (Mitchell et al. 2012).

Switchgrass is a warm-season, perennial, C4 grass species, native to the Great Plains (Vogel 2004; Kaul et al. 2006). Switchgrass has been the subject of long-term agronomic research (Schmer et al. 2008; Mitchell et al. 2012), and in recent years, has been promoted as a bioenergy crop (Sanderson et al. 2004; Vogel 2004; Mitchell et al. 2008). Simple sugars from switchgrass cell walls can be fermented to produce cellulosic ethanol (Dein et al. 2006). Switchgrass thrives in rain fed systems east of the 100th Meridian (Vogel 2004; Kaul et al. 2006) where non-irrigated (dryland) farming can be conducted in most years (Mitchell et al. 2010).

Switchgrass is an alternative biofuel feedstock that could be produced on a large scale in agricultural landscapes, with environmental and economic benefits (Dale et al. 2010; Perrin et al. 2008; Mitchell et al. 2010). Economically, switchgrass is a relatively drought tolerant crop (Vogel 2004; Sarath et al. 2008), produces large quantities of biomass on marginally productive lands (Vogel 2004; Schmer et al. 2008), requires less water and chemical inputs than annual rowcrops, requires less intensive management than annual rowcrops, can be managed and harvested using traditional farm machinery (Mitchell et al. 2010), and could help diversify farm income (Sanderson et al. 2004; Gopalakrishnan et al. 2009; Mitchell et al. 2010). Switchgrass is also net energy positive (Schmer et al. 2008). Environmentally, switchgrass is a near carbon neutral fuel source (McLaughlin et al. 2002; Fargione et al. 2008) that releases less carbon into the atmosphere than rowcrop cultivation (Adler et al. 2007) and sequesters carbon in prairie soils (McLaughlin et al. 2002; Vogel 2004; Mitchell et al. 2010). Perennial grasses like switchgrass are common components of CRP conservation plantings and have been promoted for reducing soil erosion and protecting water resources (McLaughlin & Kszos 2005). Although switchgrass is not likely to replace rowcrops on productive soils or irrigated fields, it could replace rowcrops on non-irrigated, marginally productive lands, in addition to CRP and other native grasslands. (Mitchell et al. 2010)

It is unclear how the conversion of marginally productive agricultural lands to switchgrass might affect grassland bird populations, because most studies addressing the impacts of switchgrass monocultures on wildlife are conducted in CRP switchgrass plantings, which are managed less intensively and are more structurally and florally diverse than bioenergy switchgrass stands (McCoy et al. 2001; Gardiner et al. 2010).

Structurally and florally heterogeneous grasslands provide quality habitat for multiple grassland bird species (Delisle & Savidge 1997), while more homogeneous grasslands may benefit only select species (Murray & Best 2003). Annual rowcrop fields are intensively managed, low diversity plant communities that typically support very low grassland bird densities (Johnson & Igl 1995; Best et al. 1997). Switchgrass stands may provide birds with habitat that annual rowcrops do not, because the conversion of rowcrops to switchgrass involves introducing a native perennial grass back into a highly fragmented agricultural landscape (Robertson et al. 2010). However, switchgrass stands could be detrimental to grassland birds if they replace CRP enrolled grasslands or native prairie polycultures, both of which are more structurally and florally diverse than switchgrass stands (Murray & Best 2003; Gardiner et al. 2010).

Scenarios are structured accounts of possible future events (Peterson et al. 2003). Scenario planning is appropriate in situations where there is a high level of uncertainty over uncontrollable future events and their impacts (Peterson et al. 2003; Williams et al. 2009). The development of the cellulosic ethanol industry and subsequent adoption of switchgrass as a bioenergy crop are highly speculative, as are future commodity prices and continued funding of the CRP. In the future, food and bioenergy production are likely to compete with regreening efforts aimed at benefitting wildlife and the environment (Tilman et al. 2009), and scenario planning allows for the comparison of alternative futures in agricultural landscapes.

It is unclear how different grassland bird species might respond to future agricultural landuse changes. To explore avian population responses, I developed three agricultural landuse change scenarios for the Rainwater Basin region of south-central

Nebraska, U.S.A., driven by potential future cellulosic ethanol demand, grain market prices, and continuation of the CRP. For each scenario, I generated spatially explicit landcover maps and calculated changes in area for major landcover classes within 40 km road network service areas of existing ethanol plants. Changes in area were input into a customized version of the Hierarchical All Birds Strategy (HABS) model (PLJV 2007) to predict changes in abundance for a suite of grassland bird species under different landuse change scenarios. Changes in abundance were compared between scenarios to assess which forms of potential future agricultural landuse change may benefit or hinder individual grassland bird species most.

METHODS

STUDY AREA

The Rainwater Basin is a major watershed of the Greater Platte River Basins (USGS 2009), covering 15,800 km^2 in all or portions of 21 south-central Nebraska counties (LaGrange 2005) (figure 1). This study was conducted within the 40 km service areas of the 11 ethanol plants currently servicing the Rainwater Basin region. In this intensively farmed area, irrigation and dryland farming are the dominant landuses. Groundwater for irrigation is obtained from the underlying Ogallala Aquifer (McGuire 2011), and the majority of the agricultural landscape is utilized for maize and soybean (*Glycine max*) production, although small grain farming and cattle ranching are also conducted on smaller scales (Gilbert 1989; Bishop & Vrtiska 2008). The region is also an important stopover location for migrating waterfowl and shorebirds. Thousands of shallow, rain–fed wetlands dot the agricultural landscape, providing critical wetland habitat for birds traveling through a narrow stretch of the Central Flyway migration route (Gersib et al. 1991; RWBJV 1994; Bishop & Vrtiska 2008). An estimated 7 – 14 million North American ducks and geese utilize Rainwater Basin wetland habitat annually, in addition to various shorebird species (Farmer & Parent 1999; Lagrange 2005).

DATA SOURCES

Agricultural irrigation type and complete landcover Geographic Information System (GIS) data for the Rainwater Basin region were provided by the Rainwater Basin Joint Venture (http://www.rwbjv.org)**,** and road, stream and political boundary GIS data were downloaded from the Nebraska Department of Natural Resources **(**http://dnr.ne.gov/databank/spat.html**).** Geographic coordinates of ethanol plants servicing the Rainwater Basin were collected from Google Earth satellite imagery, and plant locations were digitized in ArcGIS. All GIS data layers were projected in the North American Datum 1983 Universal Transverse Mercator Zone 14 North coordinate system.

ETHANOL PLANT SERVICE AREAS

Agricultural lands in close proximity to ethanol plants are more likely to experience biofuel-based landuse change in the future than lands located farther from ethanol plants, due to the availability of starch-based ethanol conversion infrastructure that could be modified for cellulosic ethanol production. I used the Network Analyst extension in ArcGIS to generate 40 km network service areas for 11 ethanol plants currently servicing the Rainwater Basin, using all roads within the State of Nebraska as travel corridors. Forty kilometers is the approximate maximum distance producers are willing to transport grain or feedstocks to biorefineries for processing (Khanna et al. 2008; Mitchell et al. 2012). I used the 40 km service area boundaries to restrict the Rainwater Basin landcover to service area boundaries prior to analysis.

LANDCOVER CLASSES

I identified four major agriculturally or ecologically important landcover classes that together account for over 89% of total Rainwater Basin landcover within the ethanol plant service areas. These landcover classes are rowcrops, CRP enrolled grasslands, non-CRP grasslands and wet meadows. Woodlands and developed areas accounted for the majority of remaining land area; however neither are directly utilized for rowcrop production or grassland bird habitat, and therefore were omitted from further analysis.

Rainwater Basin grasslands consist of CRP enrolled grasslands, non-CRP grasslands and wet meadows. CRP grasslands are highly erodible croplands that have been removed from production and seeded with nonnative (CP1) or native (CP2) conservation plantings (King & Savidge 1995; USDA – NRCS 2012). CP1 plantings typically consist of smooth brome (*Bromus inermis*) and legumes, whereas native plantings are composed primarily of native tallgrass species like big bluestem (*Andropogon gerardii*), little bluestem (*Schizachyrium scoparium*), switchgrass and Indiangrass (*Sorghastrum nutans*) (Clark et al. 1993; Delisle & Savidge 1997). In this study, no differentiation was made between nonnative and native grassland conservation plantings because it was difficult to differentiate between planting with existing GIS data. Non-CRP grassland includes remnant grasslands, cattle pastures, restored grasslands not affiliated with the CRP, grass buffers surrounding restored wetland sites, and grass linings of road ditches and canals (Utrup & Davis 2007; Ramirez-Yanez et al. 2011). Wet meadows are commonly flooded and hayed riparian grasslands dominated by sedges, rushes and other native mixedgrass prairie species (Currier 1989).

LANDUSE CHANGE SCENARIOS

I developed three scenarios for assessing potential impacts of future landuse change on Rainwater Basin grassland bird populations. These scenarios encompass a range of potential futures for agricultural lands, with major drivers of future change being ethanol demand, commodity markets, and continuation of the CRP. Interactions between these drivers could determine the future enrollment of marginally productive agricultural lands in rowcrops, bioenergy switchgrass, or the CRP, which in turn could affect the quantity and quality of wildlife habitat.

Under the CRP to Switchgrass Scenario, increased cellulosic ethanol demand, high grain market prices and decreased CRP funding resulted in the conversion of all 2,583 ha of Rainwater Basin CRP grassland within the 40 km ethanol plant service areas to switchgrass, while presently farmed rowcrop fields remained in rowcrop production. CRP grassland currently comprises 0.19% of landcover within the study area, making the conversion of only CRP grassland to switchgrass for cellulosic ethanol production infeasible from an ethanol production standpoint. Assuming a switchgrass biomass yield of 11 Mg/ha (Schmer et al. 2008) and a cellulosic ethanol conversion efficiency of 329 l/Mg (Varvel et al. 2008), CRP grassland land area alone could produce only 5% of the 566,990 Mg of biomass required to supply just one cellulosic ethanol plant with an annual ethanol production capacity of 189,270,589 l (Mitchell et al. 2012). However, this scenario affords the opportunity to assess the potential ecological impacts involved in converting CRP grasslands to switchgrass, without factoring in the impacts associated with the conversion of greater rowcrop areas to switchgrass.

The CRP to Rowcrops Scenario addresses the future management of marginally productive agricultural lands under decreased cellulosic ethanol demand, high market grain prices and decreased CRP funding. Under these conditions, all 2,583 ha of Rainwater Basin CRP grassland within the 40 km ethanol plant service areas are converted to annual rowcrop production, with presently farmed rowcrop fields remaining in rowcrop production. If funding for the CRP decreases or is eliminated, the cellulosic ethanol industry fails to develop and grain market prices remain high, farmers will have little incentive for enrolling marginally productive croplands in alternative biofuel feedstocks or conservation plantings, and may convert CRP enrolled hectares back to rowcrop production as CRP contracts expire.

The Rowcrops to CRP Scenario assumes decreased cellulosic ethanol demand, low market grain prices and increased CRP funding. In this scenario, the CRP program is expanded and gains popularity in the Rainwater Basin, resulting in a doubling of currently enrolled CRP hectares. If the cellulosic ethanol industry fails to develop, grain market prices decrease, and financial support for conservation programs aimed at benefitting ecosystems and the environment increases, enrolling additional marginally productive lands in the CRP may become more appealing to farmers. Doubling CRP grassland area in the study area through the replacement of 2,583 rowcrop hectares with CRP grassland increased the number of CRP enrolled grassland hectares to 5,166.

LANDCOVER AREA

A GIS shapefile representing total current cropland was converted from vector to raster format with the polygon to raster tool, reclassified into a single class and input into the program Fragstats to calculate total area. CRP grassland, non-CRP grassland and wet meadow grassland areas were obtained directly from the 40 m ethanol plant service area Rainwater Basin landcover layer. Landcover areas were calculated by multiplying the number of raster cells occupied by each landcover type by 900 m^2, the area covered by a single raster cell.

GRASSLAND BIRD ABUNDANCE

A customized version of the HABS (Hierarchical All Birds Strategy) model (PLJV 2007) was used to predict current abundances for a suite of Rainwater Basin grassland bird species within 40 km ethanol plant service areas, in addition to changes in abundance under each of the three landuse change scenarios. HABS is a hierarchically organized database that links bird conservation regions (BCRs) within the PLJV (Playa Lakes Joint Venture) with different landuse associations. The PLJV covers nearly 777,000 km^2 in portions of seven Great Plains states. In Nebraska, PLJV BCRs consist of prairie potholes, shortgrass prairie, mixedgrass prairie and tallgrass prairie (PLJV 2007). The Rainwater Basin region lies almost entirely within the mixedgrass prairie region of Nebraska (BCR 19 – Nebraska), and all HABS model runs in this study are conducted for BCR 19 – Nebraska.

Landuse associations are general forms of landuse within PLJV BCRs. For different landuse associations, various habitat conditions exist. Maize is an example of a habitat conditions housed within the cropland landuse association. With the assistance of the Nebraska Bird Partnership, I customized a bioenergy switchgrass habitat condition within the cropland landuse association. Existing scientific literature was used to populate individual habitat conditions with bird density estimates for the breeding or non-breeding seasons (Table 1). Within HABS, the breeding season abundance estimate for an individual species in a single landuse condition was calculated by multiplying: the bird density estimate within the habitat condition; total number of hectares enrolled in the habitat condition; the proportion of hectares comprising the habitat condition available as habitat for the species; the proportion of hectares comprising the habitat condition considered suitable habitat for the species; and a proportion indicating how minimum area requirements may limit the utilization of the habitat condition by the species (PLJV 2007). Bird abundances from all habitat conditions were summed to generate a total abundance estimate for the species across the landscape. In this study, I considered all habitat conditions 100% available and 100% suitable for all species, and did not factor in minimum area requirements. These HABS model simplifications were appropriate for developing a consistent approach for assessing species responses under different landuse change scenarios.

I used HABS to predict overall abundances and changes in abundance for the following grassland bird species under the three scenarios of landuse change: bobolink (*Dolichonyx oryzivorus*), dickcissel (*Spiza americana*), eastern kingbird (*Tyrannus tyrannus*), field sparrow (*Spizella pusilla*), grasshopper sparrow (*Ammodramus savannarum*), meadowlark (*Sturnella spp.*), ring-necked pheasant (*Phasianus colchicus*), sedge wren (*Cistothorus plantensis*) and upland sandpiper (*Bartramia longicauda*). No distinction was made between eastern meadowlarks (*Sturnella magna*) and western meadowlarks (*Sturnella neglecta*), because meadowlark density estimates were obtained from Murray & Best (2003), who did not differentiate between species in their survey. Some studies used to populate other landuse conditions did differentiate between meadowlark species (Faanes & Lingle 1995; Kim et al. 2008). In these instances, western meadowlark densities were used because they were considered the more common of the meadowlark species throughout PLJV BCR-19, which encompasses the Rainwater Basin region (Rosenberg 2004).

Grassland bird surveys in switchgrass stands managed as bioenergy crops are limited, and no surveys have been conducted in the Rainwater Basin or State of Nebraska to date. All switchgrass bird density estimates utilized in this study are from Murray and Best (2003), who surveyed grassland birds in bioenergy switchgrass stands in the Chariton Valley Region of southern Iowa, U.S.A. The Chariton Valley is characterized by rolling topography, annual rowcrops, and prevalence of native and restored grasslands. This landscape differs from the Rainwater Basin primarily in the greater proportion of the Chariton Valley landscape that remains in some form of grassland (Murray & Best 2003). Density estimates for bobolinks and sedge wrens in annual rowcrops were obtained from Johnson and Igl (1995), who conducted their study in North Dakota, where CRP grassland comprised a considerable portion of the landscape. In addition to changes in landcover type, differences in landscape factors and regional climate could drive differences in grassland bird abundances between landscapes.

Lower and upper grassland bird density estimates for each species in each habitat condition were calculated by multiplying the mean observed density for each species in each habitat condition by the standard error of the mean density estimate for the considered species from Murray and Best (2003), and then subtracting and adding the result from the mean density estimate. Murray and Best (2003) standard errors were used for all species in all habitat conditions because some studies from which density estimates were obtained did not include standard errors. Lower and upper density estimates for each species in each habitat condition were each input into lower and upper copies of the HABS model, respectively. As with mean abundance estimates, lower and upper confidence intervals for individual species abundances under different landuse change scenarios were calculated by multiplying lower and upper density estimates for each habitat condition by the number of hectares enrolled in the respective habitat condition under the considered scenario, and then summing the appropriate lower or upper abundance estimates from all habitat conditions. Percent changes in bird abundance for each scenario were determined by calculating the difference between current mean abundance and the newly generated mean abundance under the scenario, dividing the difference by the current bird abundance, and multiplying the result by 100. Percent changes in abundance were useful for assessing and comparing the potentials of different landuse change scenarios to impact bird abundances.

RESULTS

Within the 40 km ethanol plant service areas of the Rainwater Basin, rowcrops dominate landuse, occupying 1,010,180 ha, or 74% of total land area. Non-CRP grassland covers 188,930 ha; wet meadows 13,718 ha; and CRP grassland 2,583 ha. Together, these agriculturally and economically important landcover classes account for approximately 90% of the total 1,357,850 Rainwater Basin ha within the 40 km ethanol plant service areas. Grasshopper sparrows, bobolinks, dickcissels and meadowlarks were predicted to be the most abundant species in HABS models (Table 2).

All 2,583 ha of CRP grassland in the ethanol plants service area polygon were converted to switchgrass under the CRP to Switchgrass Scenario. Rowcrop enrolled hectares remained constant at 1,010,180, as did non-CRP grasslands (188,930 ha) and wet meadows (13,718 ha). Abundances of six out of nine bird species decreased. Percent changes in abundance were less than 10% for all species except dickcissels, which displayed a 13.6% decrease in abundance. Sedge wrens and ring-necked pheasants decreased by 2.3% and 1.1% respectively, and grasshopper sparrows, bobolinks and meadowlarks decreased < 1%. Field sparrows, eastern kingbirds and upland sandpipers exhibited positive responses to the replacement of CRP grassland with switchgrass, although responses were slight (Table 3).

In the CRP to Rowcrops Scenario, all 2,583 currently enrolled CRP grassland hectares within the ethanol plants service area polygon were converted back to rowcrops, increasing the rowcrop area to 1,012,763 ha. Non-CRP grasslands and wet meadows remained constant at 188,930 ha and 13,718 ha, respectively, and no hectares were enrolled in bioenergy switchgrass production. Dickcissels decreased by 14.0% – 14.2%, sedge wrens by 3.9% – 4.9%, ring-necked pheasants by 1.9% – 2.0% and grasshopper sparrows by 1.5% – 1.8%. Meadowlarks, bobolinks, eastern kingbirds and field sparrows all decreased by less than 1%, and upland sandpipers increased between 0.00% and 0.02% (Table 3).

Under the Rowcrops to CRP Scenario, 2,583 hectares of marginally productive rowcrop hectares were removed from crop production, enrolled in the CRP and seeded with conservation grassland plantings. This conversion reduced the number of rowcrop enrolled hectares to 1,007,597 and increased the number of CRP grassland hectares to 5,166. 188,930 ha remained in non-CRP grasslands, 13,718 ha in wet meadows, and no hectares in bioenergy switchgrass production. Species responses in this scenario mirrored those observed under the CRP to Rowcrops Scenario. Dickcissels increased by 14.0% – 14.2%, sedge wrens by 3.9% – 4.9%, ring-necked pheasants by 1.9% – 2.0% and grasshopper sparrows by 1.5% – 1.8%. Meadowlarks, bobolinks, eastern kingbirds and field sparrows all increased < 1%, and upland sandpipers decreased between 0.00% and 0.02% (Table 3).

DISCUSSION

The overall impact of converting CRP grassland to switchgrass or rowcrops on the grassland bird community was negative, whereas converting rowcrops to CRP grassland yielded positive responses. Although converting CRP grassland to switchgrass is predicted to negatively impact grassland birds, returning CRP grassland to rowcrop production could be even more detrimental. Rowcrops are generally unsuitable for grassland birds and switchgrass stands may be moderately suitable when compared with CRP grassland. These results illustrate the ecological importance of CRP in agricultural landscapes and the dangers that could result from replacing it with alternative agricultural landuses that promote the growth of monocultures.

CRP grasslands are typically more florally and structurally diverse than rowcrops or switchgrass stands, and therefore could satisfy the habitat preferences of a variety of species. CRP grassland may be most important for species like dickcissels, which prefer patchy vegetation with tall forb species for nesting, perching and singing (Skinner 1975; Bryan & Best 1991; Delisle & Savidge 1997; Johnsgard 2009). Meadowlarks also prefer vegetative habitat with elevated singing perches (Weins 1969; Skinner 1975; Herkert 1994; Renfrew & Ribic 2002), which may be more readily available in CRP grassland than switchgrass or rowcrops. CRP grasslands are generally not grazed, hayed or mowed (USDA – NRCS 2012), and therefore could benefit sedge wrens and other species that thrive in tall, dense vegetation (Skinner 1975; Bryan & Best 1991; Delisle & Savidge 1997; Johnsgard 2009). Ring-necked pheasants are one of the earliest nesting grassland bird species, and rely on the presence of residual vegetation from the previous year for winter cover (Delisle & Savidge 1997) and constructing nests (Haensly et al. 1987). CRP grassland provides good nesting habitat for early nesting species like ring-necked pheasants (Clark & Bogenschutz 1999), whereas the usefulness of switchgrass for early nesting habitat may be dependent on stand regrowth following harvest (Murray & Best 2003). Although switchgrass stands hayed after the first killing frost could provide the short, sparse vegetative structure preferred by grasshopper sparrows and upland sandpipers (Murray & Best 2003), the additional reliance of these species on large grassland expanses (Helzer & Jelinski 1999) may be better provided in CRP grassland if only small, marginally productive rowcrops field portions are converted to switchgrass. The limited responses to landuse change scenarios exhibited by bobolinks, upland sandpipers, field sparrows and eastern kingbirds indicates their reliance on habitats other than those found in annual rowcrops or switchgrass stands. Conserving and restoring CRP grassland, non-CRP grassland and wet meadows might be most important for these species.

CRP grassland represents only 0.19% of the Rainwater Basin landscape. Despite its ecological and environmental value, the CRP may have difficulty competing economically with rowcrops in the future, given presently high commodity demand and prices that encourage farmers to raise rowcrops. The continuation of the CRP, in coordination with the conversion of marginally productive croplands to switchgrass could benefit grassland bird populations. However, if both marginally productive croplands and CRP grasslands are converted to switchgrass, the habitat improvements associated with the conversion of rowcrops to switchgrass could be offset by the conversion of CRP grasslands to switchgrass.

The impacts of future agricultural landuse change on Rainwater Basin grassland bird populations will depend on which landcover types are replaced, the alternative forms of landuse are adopted, and how intensely landuse change occurs. Species minimum area requirements, sensitivity to habitat fragmentation, and other local and landscape metrics that may influence avian utilization of different habitat types are not accounted for in this analysis. In this context, results should be interpreted carefully and generally. In highly cultivated landscapes like the Rainwater Basin, CRP grassland is limited, but crucial avian habitat. Losses in CRP grassland area could negatively impact dickcissels, sedge wrens, ring-necked pheasants and grasshopper sparrows. In less intensively cultivated landscapes, it is likely that greater land areas remain in some form of grassland. In these landscapes, a greater potential for converting CRP grassland to alternative landuses exists. Although the conversion of CRP grassland to switchgrass or rowcrops could decrease abundances of multiple grassland bird species, the conversion to switchgrass is likely to be less detrimental. Maintaining the CRP program is important for the continued provision of quality grassland bird habitat, especially in agricultural landscapes where high diversity grassland area is limited.

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TABLES AND FIGURES

Table 1: Grassland bird densities in bioenergy switchgrass, rowcrops, CRP grassland, non-CRP grassland and wet meadows input into the HABS model and used to predict current Rainwater Basin grassland bird abundances and changes in abundance under three agricultural landuse change scenarios.

Name	Landcover type	Reference	Density (birds/hectare)
Bobolink	Switchgrass	Murray & Best 2003	0.0240
Dickcissel	Switchgrass	Murray & Best 2003	0.0751
Eastern kingbird	Switchgrass	Murray & Best 2003	0.0200
Field sparrow	Switchgrass	Murray & Best 2003	0.0089
Grasshopper sparrow	Switchgrass	Murray & Best 2003	0.3509
Meadowlark	Switchgrass	Murray & Best 2003	0.0499
Ring-necked pheasant	Switchgrass	Murray & Best 2003	0.0309
Sedge wren	Switchgrass	Murray & Best 2003	0.0739
Upland sandpiper	Switchgrass	Murray & Best 2003	0.0069
Bobolink	Rowcrops	Johnson & Igl 1995	0.0245
Dickcissel	Rowcrops	Best et al. 1997	0.0079
Eastern kingbird	Rowcrops	Best et al. 1997	0.0079
Field sparrow	Rowcrops	Best et al. 1997	0.0020
Grasshopper sparrow	Rowcrops	Best et al. 1997	0.0040
Meadowlark	Rowcrops	Best et al. 1997	0.0040
Ring-necked pheasant	Rowcrops	Best et al. 1997	0.0059
Sedge wren	Rowcrops	Johnson & Igl 1995	0.0000
Upland sandpiper	Rowcrops	Best et al. 1997	0.0040
Bobolink	CRP grassland	Delisle & Savidge 1997	0.1295
Dickcissel	CRP grassland	Delisle & Savidge 1997	1.6741
Eastern kingbird	CRP grassland	Delisle 1995	0.0146

Table 1 continued.

Name	Landcover type	Reference	Density (birds/hectare)
Field sparrow	CRP grassland	Delisle 1995	0.0054
Grasshopper sparrow	CRP grassland	Delisle & Savidge 1997	0.5211
Meadowlark	CRP grassland	Delisle & Savidge 1997	0.0694
Ring-necked pheasant	CRP grassland	Delisle & Savidge 1997	0.0591
Sedge wren	CRP grassland	Delisle & Savidge 1997	0.1376
Upland sandpiper	CRP grassland	Delisle 1995	0.0030
Bobolink	Non-CRP grassland	Faanes & Lingle 1995	0.0541
Dickcissel	Non-CRP grassland	Faanes & Lingle 1995	0.0640
Eastern kingbird	Non-CRP grassland	Faanes & Lingle 1995	0.0299
Field sparrow	Non-CRP grassland	Faanes & Lingle 1995	0.0200
Grasshopper sparrow	Non-CRP grassland	Faanes & Lingle 1995	0.3600
Meadowlark	Non-CRP grassland	Faanes & Lingle 1995	0.3800
Ring-necked pheasant	Non-CRP grassland	Faanes & Lingle 1995	0.0040
Sedge wren	Non-CRP grassland	Utrup & Davis 2007	0.0334
Upland sandpiper	Non-CRP grassland	Faanes & Lingle 1995	0.0400
Bobolink	Wet meadow	Kim et al. 2008	1.0645
Dickcissel	Wet meadow	Kim et al. 2008	0.4302
Eastern kingbird	Wet meadow	Faanes & Lingle 1995	0.0739
Field sparrow	Wet meadow	Faanes & Lingle 1995	0.0000
Grasshopper sparrow	Wet meadow	Kim et al. 2008	0.1843
Meadowlark	Wet meadow	Kim et al. 2008	0.2219
Ring-necked pheasant	Wet meadow	Kim et al. 2008	0.0000
Sedge wren	Wet meadow	Kim et al. 2008	0.0418
Upland sandpiper	Wet meadow	Kim et al. 2008	0.2511

Table 2: Predicted lower and upper confidence interval bounds for current Rainwater Basin grassland bird abundances within 40 kilometer ethanol plant service areas.

Species	**Abundance (lower)**	**Abundance (upper)**
Bobolink	32,754.87	66,993.04
Dickcissel	16,181.19	66,645.93
Eastern kingbird	7,479.99	19,655.15
Field sparrow	3,038.40	10,597.91
Grasshopper sparrow	55,059.01	175,601.05
Meadowlark	15,255.94	34,968.18
Ring-necked pheasant	129.57	17,702.48
Sedge wren	1,703.17	39,967.46
Upland sandpiper	10,005.51	21,015.31

Table 3: Predicted lower and upper confidence interval bounds for percent changes in Rainwater Basin grassland bird abundances within 40 kilometer ethanol plant service areas under three landuse change scenarios. Under the CRP to Switchgrass Scenario, all 2,583 hectares of CRP grassland are converted to switchgrass. All 2,583 hectares of CRP grassland are converted to rowcrops under the CRP to Rowcrops Scenario. In the Rowcrops to CRP Scenario, 2,583 rowcrop hectares are converted to CRP grassland, increasing CRP grassland area to 5,166 hectares.

Species	**Scenario**	**Percent change (lower)**	**Percent change (upper)**
Bobolink	CRP to Switchgrass	-0.55%	-0.55%
Dickcissel	CRP to Switchgrass	-13.63%	-13.63%
Eastern kingbird	CRP to Switchgrass	0.10%	0.10%
Field sparrow	CRP to Switchgrass	0.15%	0.15%
Grasshopper sparrow	CRP to Switchgrass	-0.58%	-0.58%
Meadowlark	CRP to Switchgrass	-0.26%	-0.26%
Ring-necked pheasant	CRP to Switchgrass	-1.06%	-1.06%
Sedge wren	CRP to Switchgrass	-2.28%	-2.28%
Upland sandpiper	CRP to Switchgrass	0.03%	0.07%
Bobolink	CRP to Rowcrops	-0.54%	-0.54%
Dickcissel	CRP to Rowcrops	-14.01%	-14.20%
Eastern kingbird	CRP to Rowcrops	-0.12%	-0.12%
Field sparrow	CRP to Rowcrops	-0.07%	-0.15%
Grasshopper sparrow	CRP to Rowcrops	-1.49%	-1.76%
Meadowlark	CRP to Rowcrops	-0.72%	-0.87%
Ring-necked pheasant	CRP to Rowcrops	-1.88%	-1.99%
Sedge wren	CRP to Rowcrops	-3.95%	-4.92%
Upland sandpiper	CRP to Rowcrops	0.00%	0.02%
Bobolink	Rowcrops to CRP	0.54%	0.54%
Dickcissel	Rowcrops to CRP	14.01%	14.20%

Table 3 continued.

Species	Scenario	Percent change (lower)	Percent change (upper)
Eastern kingbird	Rowcrops to CRP	0.12%	0.12%
Field sparrow	Rowcrops to CRP	0.07%	0.15%
Grasshopper sparrow	Rowcrops to CRP	1.49%	1.76%
Meadowlark	Rowcrops to CRP	0.72%	0.87%
Ring-necked pheasant	Rowcrops to CRP	1.88%	1.99%
Sedge wren	Rowcrops to CRP	3.95%	4.92%
Upland sandpiper	Rowcrops to CRP	0.00%	-0.02%

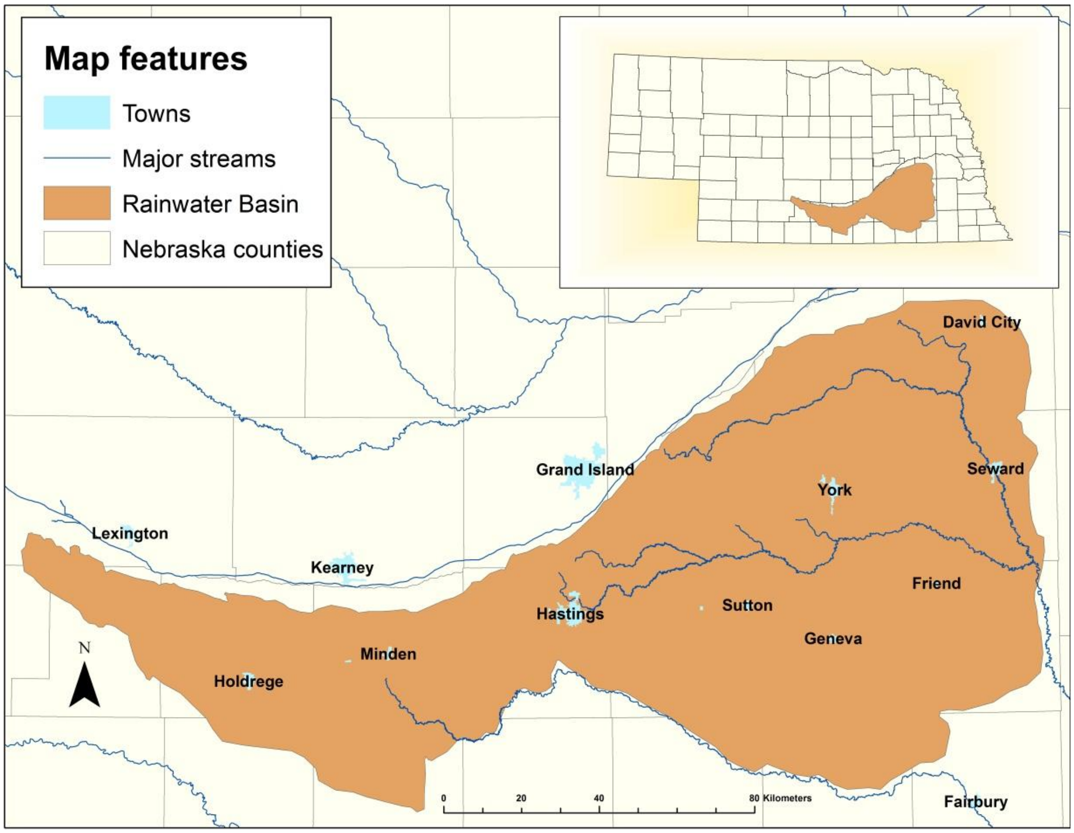

Figure 1: Location of the Rainwater Basin in south-central Nebraska, U.S.A, displaying Nebraska counties, major urban areas and rivers.

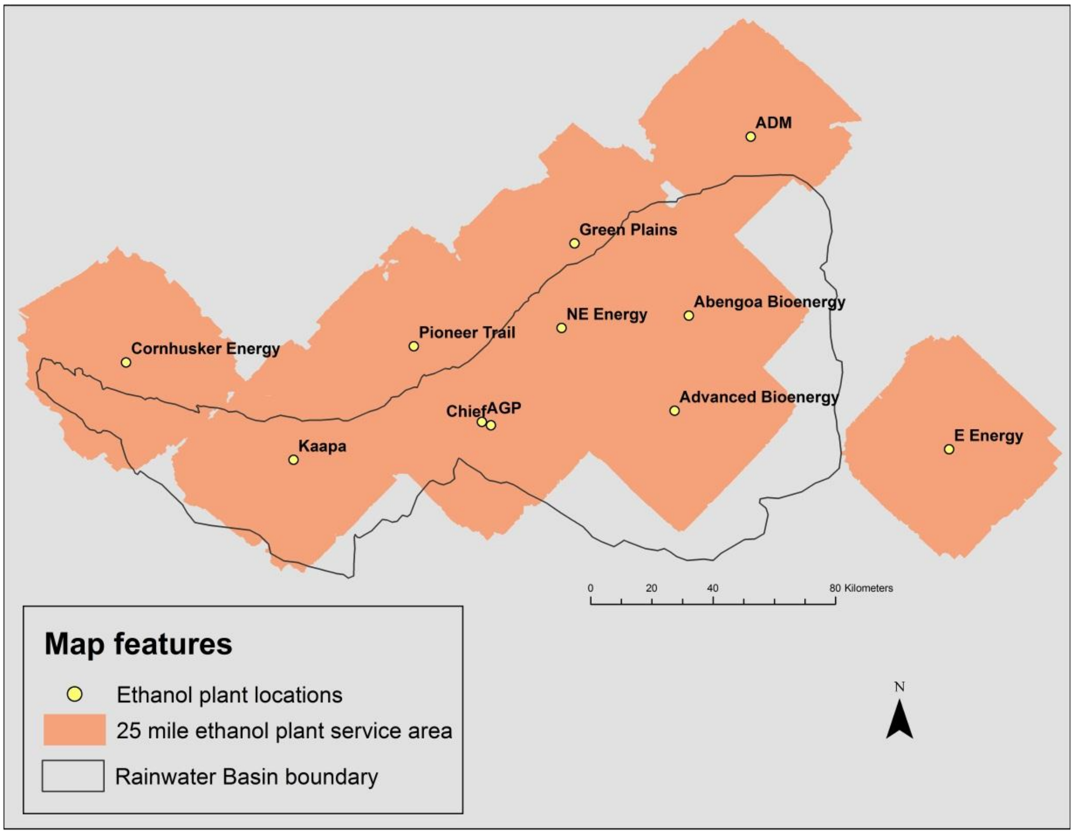

Figure 2: Locations and 40 km network service areas of existing starch-based ethanol plants currently servicing the Rainwater Basin region.

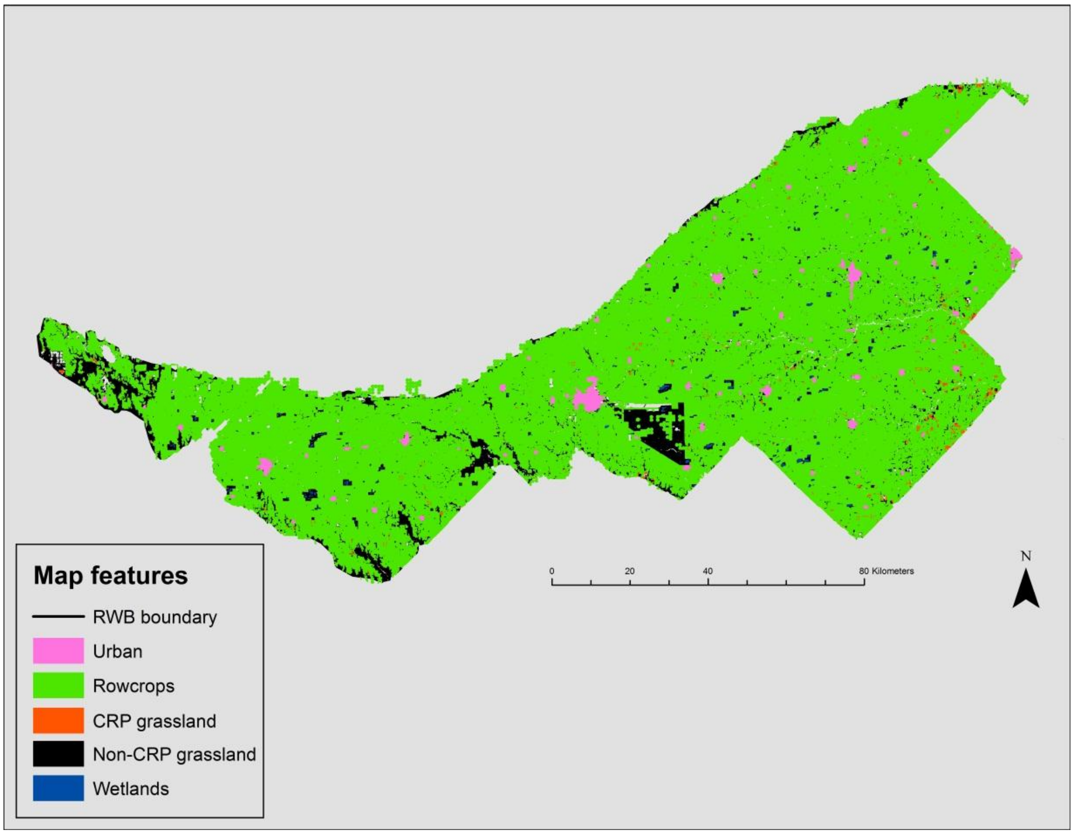

Figure 3: Major Rainwater Basin landcover classes within 40 km road network service areas of existing starch–based ethanol plants. Cropland area is the aggregation of all irrigated and non-irrigated rowcrop fields from Rainwater Basin Joint Venture (RWBJV) agricultural irrigation type data. Urban areas were derived from Nebraska Department of Natural Resources political boundaries data, and grasslands and wetlands were extracted from 2010 RWBJV landcover.

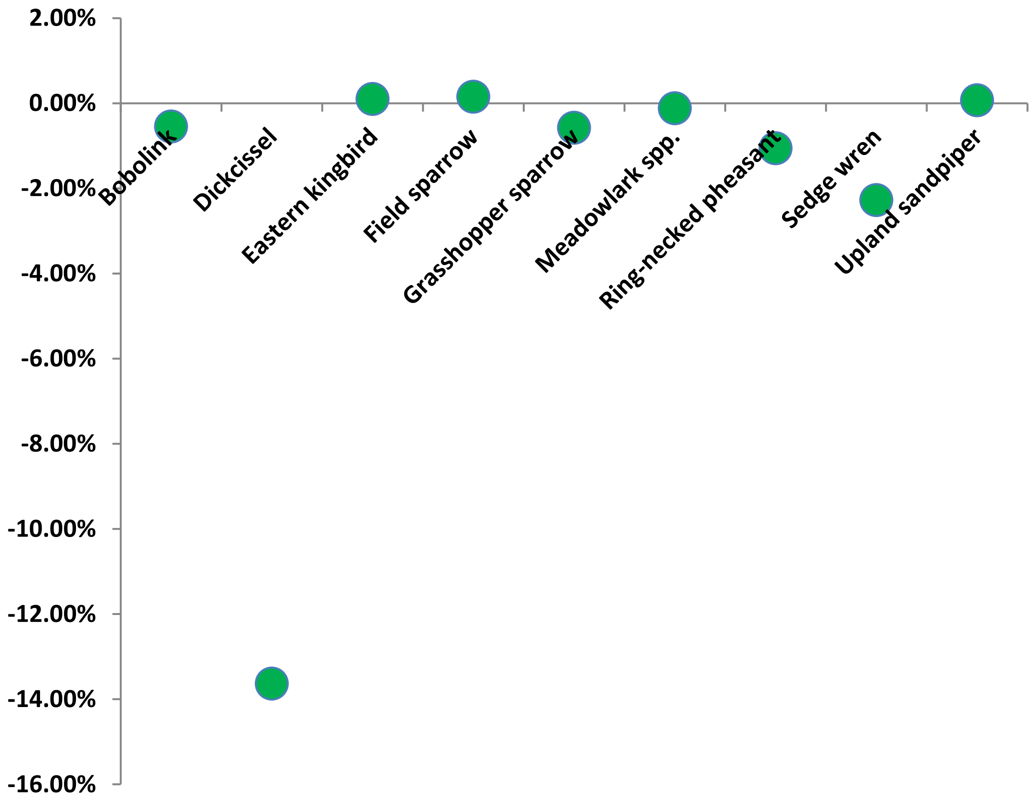

Figure 4: Mean percent changes in abundance for nine grassland bird species under the CRP to Switchgrass Scenario, in which all 2,583 ha of CRP grassland within the 40 km service areas of existing starch–based ethanol plants in the Rainwater Basin are converted to switchgrass.

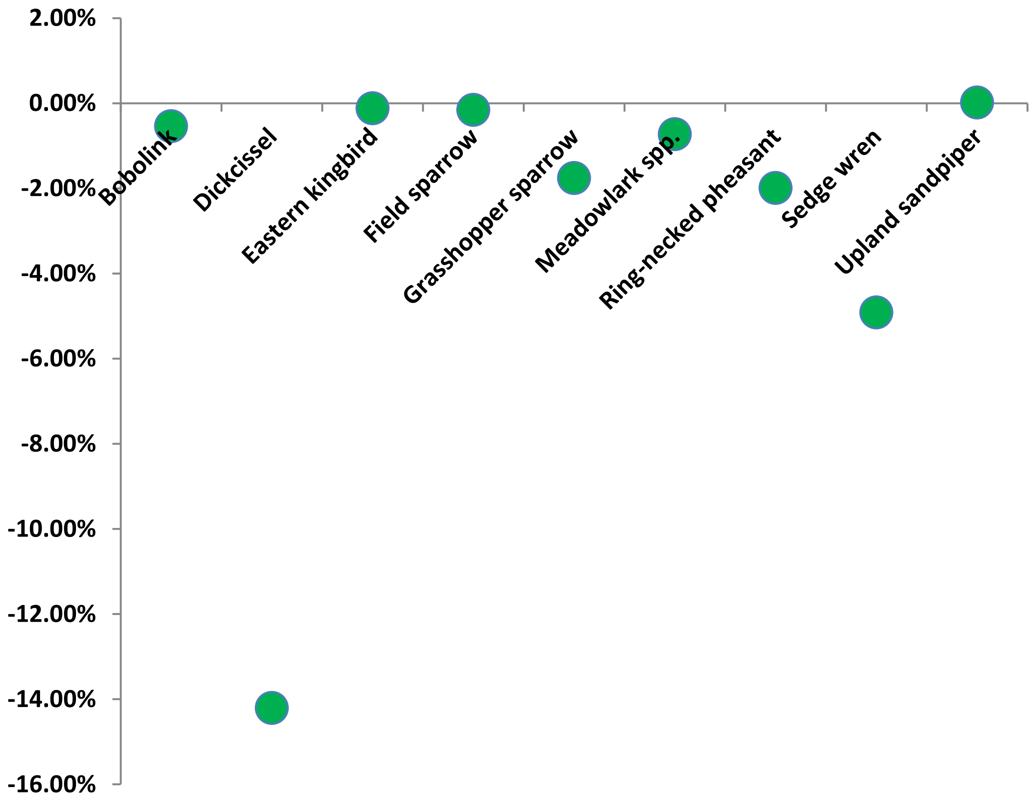

Figure 5: Mean percent changes in abundance for nine grassland bird species under the CRP to Rowcrops Scenario, in which all 2,583 ha of CRP grassland within the 40 km service areas of existing starch–based ethanol plants in the Rainwater Basin are converted to rowcrops.

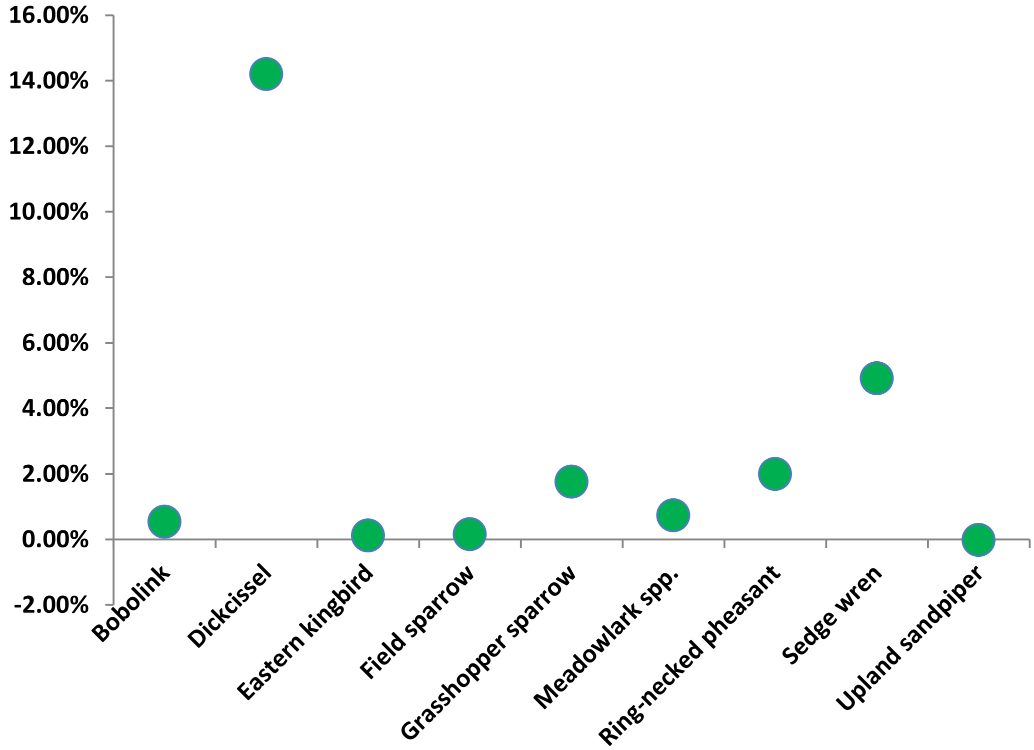

Figure 6: Mean percent changes in abundance for nine grassland bird species under the Rowcrops to CRP Scenario, where CRP grassland area within the 40 km service areas of existing starch–based ethanol plants in the Rainwater Basin is doubled through the conversion of 2,583 ha of rowcrops to CRP grassland, increasing the area of CRP grassland to 5,166 ha.

CHAPTER 5: POTENTIAL FUTURE IMPACTS OF CLIMATIC CHANGE AND BIOFUEL PRODUCTION ON RAINWATER BASIN AGRICULTURAL GROUNDWATER USE

ABSTRACT

Since 19th Century European settlement, much of the North American Great Plains landscape has been transformed from native prairie to agriculture. This transformation was aided by the availability of ample groundwater from the High Plains Aquifer system for irrigation. Climatic changes and biofuel crops could further transform agricultural and prairie landscapes and impact groundwater use. Switchgrass (*Panicum virgatum*) is a relatively drought tolerant alternative biofuel feedstock that may be environmentally and economically superior to maize (*Zea mays*) grain for ethanol production. Non-irrigated, small, marginally productive rowcrop fields are most likely to be converted to switchgrass in the future; however, irrigated fields on poor agricultural soils and in drier areas where irrigation limitations have been imposed in the past may also become suitable for raising switchgrass under novel climatic conditions and agricultural policies. To investigate potential changes to water use, I developed three agricultural landuse change scenarios for the Rainwater Basin region of Nebraska, USA, driven by potential future climatic changes, irrigation limitations, grain market prices and ethanol demand. For each scenario, I generated spatially explicit maps of rowcrop and switchgrass distributions and identified registered groundwater irrigation wells on rowcrop fields. Average Nebraska well pumping hours and individual well pumping capacities were used to determine how the conversion of marginally productive irrigated

rowcrop fields to switchgrass in the Rainwater Basin might impact annual irrigation groundwater withdrawals from the High Plains Aquifer. Under landuse change scenarios, annual groundwater irrigation withdrawals decreased by 2.6% – 5.6% for the entire Rainwater Basin, or 9.6% – 19.1% in Natural Resource Districts where irrigation limitations have been previously implemented. Under novel climatic conditions, the adoption of switchgrass as a bioenergy crop could contribute to water conservation goals.

INTRODUCTION

Since 19th Century European settlement, the prairie landscape of the North American Great Plains has been converted to intensive agricultural production (Samson & Knopf 1994; Forrest et al. 2004). Global food and bioenergy demands are rising and driving additional landuse conversions (Tilman et al. 2002; Fargione et al. 2008). Currently, approximately 99% of the Northern Great Plains is either farmed or grazed by livestock (Forrest et al. 2004). In areas with fertile soils, agricultural lands are used for the production of maize (*Zea mays*), soybeans (*Glycine max*), wheat (*Triticum aestivum*), milo (*Sorghum bicolor*), alfalfa (*Medicago sativa*) and cotton (*Gossypium spp.*) (Musick et al. 1990; Guru & Horne 2000; Mitchell et al. 2010). Dryland farming is conducted east of the 100th meridian (Mitchell et al. 2010), but rowcrops are also irrigated with groundwater from the underlying High Plains Aquifer system (HPA 1982; Peterson et al. 2003b; McGuire 2011). Irrigation increases agricultural yields when moisture availability limits crop productivity (Musick et al. 1990; Weinhold et al. 1995).

The High Plains Aquifer is the largest freshwater aquifer system in the U.S.A., supplying 30% of total groundwater irrigation withdrawals for the country (Sophocleous 2005). The aquifer underlies portions of 8 Great Plains states: Colorado, Kansas, Nebraska, New Mexico, Oklahoma, South Dakota, Texas and Wyoming (McGuire 2011), and is composed of saturated silt, sand and gravel between the water table and aquifer floor (Peterson et al. 2003b). The heart of the aquifer lies beneath extensive remnant grasslands in the Nebraska Sandhills region, where neither rowcrop production nor irrigation are common (Peterson et al. 2003b). The saturated thickness of the aquifer generally decreases from north to south (Peterson et al. 2003b), and water table levels, groundwater withdrawals and groundwater recharge rates vary between locations and times of the year (Rosenberg et al. 1999; Guru & Horne 2000; Sophocleous 2005). Since the mid-20th Century, equilibrium in water table levels has been upset by large-scale groundwater irrigation withdrawals (Sophocleous 2005). Withdrawals are greater during the growing season, when more irrigation water is extracted, and recharge is greater outside of the growing season, when plant water use and evapotranspiration decrease (Rosenberg et al. 1999; Sophocleous 2005). In cooler years with above average precipitation, water table levels may stabilize or rise as a result of increased recharge and decreased withdrawals; whereas levels tend to diminish in drier, warmer years when withdrawals increases and recharge decreases (Rosenberg et al. 1999).

Following World War II, large-scale groundwater irrigation from the High Plains Aquifer was embraced throughout the Great Plains, increasing agricultural productivity, but lowering water table levels (Keech & Dreeszen 1959; HPA 1982; Rosenberg et al. 1999; Peterson et al. 2003b). Decreases in water table levels and saturated thickness of the aquifer have been most pronounced in the southern Great Plains (Musick et al. 1990; Guru & Horne 2000; Sophocleous 2005). Although improvements in irrigation system efficiencies have helped slow and sometimes stabilize water table level declines (Rosenberg et al. 1999), the aquifer system is still vulnerable to overuse during extended droughts or when groundwater demand increases (Guru & Horne 2000; Ojima & Lackett 2002; Peterson et al. 2003b; Sophocleous 2005).

Global climate models predict that the Great Plains region will continue to experience climatic changes throughout the 21st Century (IPCC 2007; Karl et al. 2009). In its 4th report, the Intergovernmental Panel on Climate Change (2007) projected maximum and minimum temperature increases, winter precipitation increases and summer precipitation decreases for the Northern Great Plains during the 21st Century. Average annual temperature increases of 0.6 – 2.2 degrees Celsius are projected by 2020, and increases of 1.1 – 7.2 degrees Celsius are projected by 2090 (Karl et al. 2009). Although both maximum and minimum temperatures are projected to rise, minimum temperatures are expected to increase more (IPCC 2007). Precipitation changes are less certain than temperature changes (Ojima & Lackett 2002); however, a higher proportion of precipitation is expected to fall in fewer major storm events than does presently (IPCC 2007). These changes could directly and indirectly impact societies, economies, and the environment (Karl et al. 2009). Extreme climatic changes could result in additional irrigation limitations being implemented in water stressed regions (McDonald et al. 2009), which in turn could make rowcrop production less economically feasible in certain areas.

Demand for clean, renewable energy has resulted in recent, large scale efforts to utilize maize grain for ethanol production (Schnoor et al. 2008). Despite extensive development of the starch-based ethanol industry, ethanol production from maize grain remains controversial, due to uncertainties over its net energy production, water use efficiency, ability to reduce atmospheric greenhouse gas emissions and concentrations, and competition with food production for landuse (Berndes 2008; Searchinger et al. 2008; Tilman et al. 2009; Dale et al. 2010). Meanwhile, the benefits of second generation biofuels are increasingly promoted (Tilman et al. 2009). One alternative biofuel feedstock proposed for large scale cellulosic ethanol production in Great Plains agricultural landscapes is bioenergy switchgrass (*Panicum virgatum*) (Mitchell et al. 2012).

Switchgrass is a warm-season, perennial, C4 grass species, native to the Great Plains (Vogel 2004; Kaul et al. 2006). Switchgrass has been the subject of long-term agronomic research (Schmer et al. 2008; Mitchell et al. 2012), and in recent years has been promoted as a bioenergy crop (Sanderson et al. 2004; Vogel 2004; Mitchell et al. 2008). Simple sugars from switchgrass cell walls can be fermented to produce cellulosic ethanol (Dein et al. 2006). Switchgrass thrives in rain fed systems east of the 100th Meridian (Vogel 2004; Kaul et al. 2006) where non-irrigated (dryland) farming can be conducted in most years (Mitchell et al. 2010).

Switchgrass is an alternative biofuel feedstock that could be produced on a large scale in agricultural landscapes, with environmental and economic benefits (Perrin et al. 2008; Dale et al. 2010; Mitchell et al. 2010). Economically, switchgrass is a relatively drought tolerant crop (Vogel 2004; Sarath et al. 2008), produces large quantities of biomass on marginally productive lands (Vogel 2004; Schmer et al. 2008), requires less water and chemical inputs than annual rowcrops, requires less intensive management than annual rowcrops, can be managed and harvested using traditional farm machinery (Mitchell et al. 2010), and could help diversify farmer income (Sanderson et al. 2004; Gopalakrishnan et al. 2009; Mitchell et al. 2010). Switchgrass is also net energy positive (Schmer et al. 2008). Environmentally, switchgrass is a near carbon neutral fuel source (McLaughlin et al. 2002; Fargione et al. 2008) that releases less carbon into the atmosphere than rowcrop cultivation (Adler et al. 2007) and sequesters carbon in prairie soils (McLaughlin et al. 2002; Vogel 2004; Mitchell et al. 2010). Perennial grasses like switchgrass are common components of CRP conservation plantings and have been promoted for reducing soil erosion and protecting water resources (McLaughlin & Kszos 2005).

Efforts to mitigate the effects of climate change through reductions in global greenhouse gas concentrations could generate additional interest in cellulosic ethanol production. Facilitating the growth of more perennial vegetation in practical ways is one suggested method for reducing atmospheric carbon dioxide levels (Harris et al. 2006). The extensive root systems of perennial grasses are capable of sequestering carbon in soils where cultivation has depleted carbon reserves (Mitchell et al. 2010). In addition, switchgrass requires less management than rowcrops, decreasing carbon releases from prairie sod and farm machinery (Schmer et al. 2008, Mitchell et al. 2010).

Scenarios are structured accounts of possible future events (Peterson et al. 2003a). Scenario planning is appropriate in situations where there is a high level of uncertainty over uncontrollable future events and their impacts (Peterson et al. 2003a; Williams et al. 2009). The impacts of climate change are largely uncertain; but there is scientific consensus that climate change will proceed, regardless of changes in landuse or greenhouse gas emissions (IPCC 2007; McDonald et al. 2009). Similarly, agricultural policy adjustments responding to climate change or other factors are speculative, but likely (Olesen & Bindi 2002). Both climatic changes and policy adjustments could become future drivers of landuse change in agricultural landscapes, thereby influencing agricultural groundwater use.

It is unclear how future agricultural landuse changes might affect annual groundwater withdrawals in the Great Plains. To understand possible effects, I developed three biofuel-based agricultural landuse change scenarios for the Rainwater Basin region of south-central Nebraska, U.S.A., each driven by potential future climatic changes, irrigation limitations, commodity prices and cellulosic ethanol demand. I employed a conservative approach to determine which marginally productive rowcrop fields that might be converted to switchgrass in the future. For each scenario, I generated spatially explicit landcover maps and identified irrigated rowcrop fields converted from rowcrops to switchgrass. Average well pumping hours for the State of Nebraska and water pumping capacities of individual registered groundwater irrigation wells were used to calculate average annual groundwater withdrawals and potential annual changes in withdrawals under proposed landuse change scenarios. Changes in annual groundwater withdrawals were compared between landuse change scenarios to assess how the adoption of switchgrass as a bioenergy crop might aid groundwater conservation efforts in water stressed regions under projected future climatic conditions.

METHODS

STUDY AREA

The Rainwater Basin is a major watershed of the Greater Platte River Basins (USGS 2009), covering 15,800 km^2 in all or portions of 21 south-central Nebraska counties (LaGrange 2005) (Figure 1). This study was conducted within the 40 km service areas of 11 ethanol plants currently servicing the Rainwater Basin region. In this intensively farmed area, irrigation and dryland farming are the dominant landuses. Groundwater for irrigation is pumped from the underlying High Plains Aquifer system (McGuire 2011), and the majority of the agricultural landscape is utilized for maize and soybean (*Glycine max*) production, although small grain farming and cattle ranching are also conducted on smaller scales (Gilbert 1989; Bishop & Vrtiska 2008). Hundreds of remnant and restored rain–fed wetlands also dot the agriculturally dominated landscape, providing wildlife habitat, environmental services and recreational opportunities (Lagrange 2005).

DATA SOURCES

Agricultural irrigation data for the Rainwater Basin region were provided by the Rainwater Basin Joint Venture (http://www.rwbjv.org). The Soil Survey Geographic database (SSURGO) was downloaded from the U.S. Department of Agriculture (USDA) – Natural Resource Conservation Service (NRCS) Soil Data Mart (http://soildatamart.nrcs.usda.gov); road, stream and political boundary GIS data were downloaded from the Nebraska Department of Natural Resources (http://dnr.ne.gov/databank/spat.html); and mean annual precipitation GIS data were downloaded online from the USDA – NRCS (http://datagateway.nrcs.usda.gov/GDGOrder.aspx). Geographic coordinates of ethanol plants servicing the Rainwater Basin were collected from Google Earth satellite imagery, and plant locations were digitized in ArcGIS. All GIS data layers were projected in the North American Datum 1983 Universal Transverse Mercator Zone 14 North coordinate system.

ETHANOL PLANT SERVICE AREAS

Agricultural lands in close proximity to ethanol plants are more likely to experience biofuel-based landuse change in the future than lands located farther from ethanol plants, due to the availability of starch-based ethanol conversion infrastructure that could be modified for cellulosic ethanol production (Mitchell et al. 2012). I used the Network Analyst extension in ArcGIS to generate 40 km network service areas for 11 ethanol plants currently servicing the Rainwater Basin, using all roads within the State of Nebraska as travel corridors. Forty kilometers is the approximate maximum distance producers are willing to transport grain or feedstocks to biorefineries for processing (Khanna et al. 2008; Mitchell et al. 2012). I used the 40 km service area boundaries to restrict the Rainwater Basin landcover and agricultural irrigation type GIS data layers to service area boundaries for the development of scenarios.

AGRICULTURAL IRRIGATION TYPES

Irrigation system types were used to distinguish between different rowcrop fields. Irrigation system types were center-pivots, pivot corners, gravity irrigation and dryland fields. A center-pivot is a large sprinkler system typically anchored at a field center point and connected to a groundwater well. Groundwater is pumped through a pipe extending from the center point to the least distant field perimeter, with multiple two–wheeled moving towers supporting the pipe along its extent. As the center-pivot moves in a circular motion around the field, sprinklers connected to the pipe release water to the soil surface. Pivot corners result from irrigating square shaped properties with circular center-pivot irrigation systems. Since center-pivots fail to move across pivot corners, the corners are not supplied with water. Several means of irrigating pivot corners exist, including center-pivot corner systems or lateral irrigation pipes; however, farmers in the Rainwater Basin commonly raise crops on pivot corners without irrigation. Gravity irrigation consists of a temporary lateral irrigation pipe extending along the field edge with the highest altitude and perpendicular to the direction of crop rows. Water is released from the pipe into furrows between crop rows and is moved by gravity toward the opposite end of the field. A dryland field is not irrigated by any means. In years with adequate growing season precipitation, dryland field and pivot corner grain yields are comparable with irrigated croplands; however, in drier years, they tend to yield less.

MARGINALLY PRODUCTIVE CROPLANDS

Criteria making rowcrop fields marginally productive and suitable for conversion to switchgrass were based on agricultural suitability of underlying soils, field size and shape complexity, mean annual precipitation, and likelihood of experiencing additional irrigation limitations in the future. Each field was assigned to 1 of 24 marginality classes, based on the number of criteria it satisfied (Table 1). The more criteria a field satisfied the more marginal the cropland for rowcrop production and the more suitable for conversion to switchgrass.

Dryland fields and pivot corners were considered more marginal and suitable for conversion to switchgrass than center-pivot or gravity irrigated fields, due to the lack of irrigation systems on dryland fields and pivot corners and because switchgrass is more drought tolerant and water use efficient than rowcrops (Kiniry et al. 2008). USDA – NRCS land capability classes were used to determine the agricultural suitability of field soils. Soils in classes 1 and 2 are most suitable for agriculture, whereas soils in classes 7 and 8 are completely unsuitable. Soils in classes 3, 4, 5 and 6 can be described as marginally productive, and may be better suited to less intensive forms of agricultural landuse, which could include annual haying of perennial grasses like switchgrass. Switchgrass has been shown to remain productive on poor soils with ethanol yields comparable to or greater than that of combined maize grain and stover (Varvel et al. 2008).

Pivot corners and small, complexly shaped dryland fields were considered more marginal than larger, more uniformly shaped dryland fields. Farming rowcrops on small, complexly shaped fields with increasingly large, modern farm equipment can be inconvenient and time consuming, and these fields could be better suited to raising less management intensive and perennial switchgrass stands (Mitchell et al. 2008). All pivot corners were considered small, as were dryland fields with areas less than the mean pivot corner area (3.7 ha). Dryland fields with areas greater than 3.7 ha, but less than the 25th percentile value for dryland field area (4.7 ha), and with a shape index greater than the 75th percentile value for dryland field shape index (1.56), were considered small and complexly shaped.

Due to the rain shadow effect of the Rocky Mountains, mean annual precipitation increases from west to east throughout the Great Plains (Rickets et al. 1999). The central portion of the Rainwater Basin has a mean annual precipitation of 63.5 cm (NE–DNR 2010) and serves as an approximate divider between the drier western portion and wetter eastern portion of the Rainwater Basin (Pederson et al. 1989). Fields in areas with a mean annual precipitation of 63.5 cm or less were considered more marginal than fields in areas with a mean annual precipitation greater than 63.5 cm. Switchgrass is more drought tolerant and water use efficient than rowcrops (Vogel 2004; Kiniry et al. 2008), and therefore may be more feasible to produce than rowcrops on non-irrigated croplands subject to frequent drought.

Center-pivot irrigated and gravity irrigated fields were classified according to their likelihood of experiencing additional irrigation limitations in the future. Fields were assigned to a high risk or low risk classification, based on the Natural Resource District (NRD) in which they were located. Fields in NRDs with histories of implementing moratoriums or stays on irrigation were classified as being at high risk for additional future irrigation limitations, whereas those without previously implemented moratoriums or stays were classified as being at low risk for future limitations (Kurtz 2007). If NRDs restrict irrigation in the future, switchgrass may be more feasible to produce than annual rowcrops, especially under the warmer and drier conditions that would be expected to force the implementation of irrigation limitations (IPCC 2007; McDonald et al. 2009).

Rowcrop fields classified as most marginal and suitable for conversion to switchgrass under present climatic conditions were pivot corners and small, complexly shaped dryland fields, located on soils in NRCS land capability classes 3, 4, 5 or 6 and in areas with annual average precipitation of 63.5 centimeters or less. Rowcrop fields classified as least marginal and unsuitable for conversion to switchgrass were gravity and pivot irrigated fields, and large, uniformly shaped dryland fields located on soils in NRCS land capability classes 1 or 2, in areas with annual average precipitation greater than 63.5 centimeters and at low risk of experiencing future irrigation limitations.

Remaining croplands were placed into intermediate marginality classes according to the number of marginal criteria they satisfied (Table 1).

IRRIGATION LIMITATIONS

In 2004, the Nebraska Unicameral passed the Nebraska Ground Water Management and Protection Act (Dunnigan et al. 2010). This act called for the integrated management of surface and groundwater resources and annual evaluation of the long-term future availability of hydrologically connected water supplies for surface and groundwater users (NE–DNR 2007; Dunnigan et al. 2010). In areas where surface water and groundwater are connected, excessive groundwater withdrawals can negatively reduce stream flows and lead to disparities in water availability between surface and groundwater users (Dunnigan et al. 2010). In Nebraska, hydrologically connected water resources are considered fully appropriated when water withdrawal rates threaten the continued future availability of stream flow levels necessary for supporting continued surface water or groundwater use, or when reduced stream flow levels cause Nebraska to come into noncompliance with an interstate agreement (NE–DNR 2007). In the event that water resources are determined to be fully or over–appropriated, individual NRDs are granted the authority to place stays or limitations on wells and acres in considered areas (NE–DNR 2007).

The following 7 NRDs currently service the Rainwater Basin region: Lower Republican, Tri-Basin, Central Platte, Little Blue, Upper Big Blue, Lower Big Blue and Lower Platte North. As of July 1, 2008, water resources in portions of the Central Platte, Tri-Basin and Lower Republican NRDs, all of which service the Rainwater Basin region, were considered fully appropriated (Dunnigan et al. 2010). In the future, climatic changes could increase crop water use and decrease groundwater recharge in the Great Plains (Rosenberg et al. 1999; Ojima & Lackett 2002; McDonald et al. 2009), and NRDs where irrigation limitations have been imposed in the past may be more likely to impose additional limitations in the future.

Following the designation of water resources in a river basin, subbasin or reach as fully appropriated or over–appropriated, NRDs are required to develop an integrated management plan for reducing water withdrawals (NE–DNR 2007). 20% reductions in groundwater withdrawals have been suggested and implemented in several Nebraska NRDs servicing areas where water resources are over–appropriated (Supalla 2010; Hilger 2010; Middle Republican NRD 2011). In some cases, integrated management plans could provide farmers with incentives for reducing water withdrawals (NE–DNR 2007). In the future, converting rowcrop fields to switchgrass could be an alternative but economically profitable form of landuse adopted to reduce groundwater withdrawals in response to climatic changes. Rowcrop fields on poor soils and in dry areas where additional irrigation limitations are likely to be enacted could be better suited to growing switchgrass, which has been shown to remain more productive than rowcrops on poor soils and in dry conditions (Kiniry et al. 2008; Schmer et al. 2008).

LANDUSE CHANGE SCENARIOS

I developed three scenarios for assessing potential impacts of future biofuel-based landuse change on Rainwater Basin groundwater use. These scenarios encompass a wide range of potential futures for agricultural lands, with major drivers of future change being climate, irrigation limitations, commodity markets and ethanol demand. Interactions between these drivers could determine the future enrollment of marginally productive agricultural lands in rowcrops or bioenergy switchgrass, which in turn will affect groundwater withdrawals. A conservative approach was employed to determine rowcrop fields that could be converted to switchgrass under the Limited Change Scenario, and increased conversion percentages under the Modest Change and Extreme Change Scenarios, with assumed greater climatic changes, more irrigation limitations and increased ethanol demand. Although many fields possess marginal characteristics, not all fields are expected to be converted to switchgrass, because conventional crop production on marginal soils can provide justifiable economic returns in some years. I assigned greater conversion percentages to classes satisfying more marginal criteria. However, only classes satisfying all marginal criteria had 100% of fields converted to switchgrass. Classes satisfying fewer marginal criteria were assigned lower switchgrass conversion percentages of 75, 50, 25 or 0 (Table 1). Fields not assigned to switchgrass conversion remained in rowcrop production.

The Limited Change Scenario assumes minimal climatic changes without any additional irrigation limitations and an increased demand for cellulosic ethanol. This scenario establishes the baseline for biofuel-based landuse change under current climate and policy. In the Limited Change Scenario, agricultural fields converted to switchgrass consist primarily of small, complexly shaped dryland fields and pivot corners located on marginal soils and in areas with mean annual precipitation of 63.5 centimeters or less. Under this scenario, 53,672 rowcrop hectares are converted to switchgrass.

Under the Modest Change Scenario, modest climatic changes occur and are accompanied by irrigation limitations in water stressed regions, along with a greater cellulosic ethanol demand than in the Limited Change Scenario. Under this scenario, the converted proportions of rowcrop fields in marginality classes composed of pivot corners and small, complexly shaped dryland fields located on poor soils are 25% greater than those in the Limited Change Scenario (Table 1). Additionally, between 25% and 75% of fields in marginality classes composed of larger dryland fields located on poorer soils and irrigated fields located on poorer soils and in drier areas at higher risk for irrigation limitations are converted from rowcrops to switchgrass. More gravity irrigated fields were converted than pivot irrigated fields, due to the greater water used efficiency of center-pivot irrigation systems. A total of 121,141 rowcrop hectares are converted to switchgrass in this scenario.

The Extreme Change Scenario projects more extreme climatic changes that are accompanied by widespread irrigation limitations and a high demand for cellulosic ethanol. Between 50% and 100% of all pivot corners and dryland fields were converted to switchgrass, with greater percentages of classes that satisfied more marginal criteria converted (Table 1). The proportion of some gravity irrigated and pivot irrigated fields in areas at higher risk for additional future irrigation limitations converted to switchgrass was 25% greater than in the Modest Change Scenario. More gravity irrigated fields were converted than pivot irrigated fields, due to the greater water used efficiency of center-pivot irrigation systems. The greater degree of climatic change assumed under the Extreme Change Scenario also resulted in the conversion of 25% of gravity irrigated fields located on poor soils and in drier areas that have not imposed irrigation limitations in the past to switchgrass. Under the Extreme Change Scenario, 208,827 rowcrop hectares are converted to switchgrass.

The geostatistical analyst extension in ArcGIS was used to assign percentages of rowcrop fields in different marginality classes to be converted to switchgrass under the three landuse change scenarios. Total switchgrass area under Scenarios 1, 2 and 3 and the total current cropland area were calculated in the program Fragstats. To minimize errors in area calculations due to the conversion of shapefiles to raster format, rowcrop areas under the three landuse change scenarios were calculated by subtracting converted switchgrass area from current rowcrop area.

GROUNDWATER WITHDRAWAL CALCULATIONS

I identified individual groundwater irrigation wells located on gravity or pivot irrigated fields. Mean annual groundwater withdrawals for individual groundwater wells were calculated by multiplying individual well pumping capacity (l/h) by 774 h, the average annual number of well pumping hours for the State of Nebraska (Kranz 2010). Mean annual groundwater withdrawals of all wells were summed to obtain an estimate of total annual groundwater withdrawal in the study area, and potential reductions in groundwater withdrawals were calculated by summing annual withdrawals of all wells located on fields that were converted from rowcrop to switchgrass production under landuse change scenarios.

RESULTS

There are currently 14,632 groundwater wells located on gravity or pivot irrigated rowcrop fields within 40 km ethanol plant service areas of the Rainwater Basin. Assuming each well pumps for the mean Nebraska well pumping time of 774 hrs (Kranz 2010) results in over 2.5 trillion l of groundwater being withdrawn from the High Plains Aquifer annually. Under the Limited Change Scenario, which assumed no additional climatic changes or irrigation limitations, there were 53,672 marginally productive rowcrop hectares converted to switchgrass. These converted hectares were all pivot corners and dryland fields. All groundwater wells in the study area continued to be utilized for rowcrop irrigation, and no reduction in groundwater withdrawal resulted from the conversion of marginally productive rowcrop fields to switchgrass (Table 2).

In the Modest Change Scenario, with modest climatic change and some limited irrigation limitations, there were 121,141 marginally productive rowcrop hectares converted to switchgrass. In addition to 25% - 100% of non-irrigated fields, 0% - 25% of irrigated rowcrop fields were converted to switchgrass, resulting in the cessation of groundwater pumping from 350 groundwater wells. Converted irrigated fields were primarily on poor soils and in drier areas that are likely to have additional irrigation limitations implemented in them under warmer and drier future climatic conditions. This level of conversion reduced annual groundwater withdrawals by more than 64.2 billion l, or approximately 2.6% of current estimated annual groundwater withdrawals in the study area (Table 2).

Under the Extreme Change Scenario, with assumed extreme climatic changes and more widespread irrigation limitations, there were 208,827 marginally productive rowcrop hectares converted switchgrass. 50% – 100% of non-irrigated rowcrop fields in were converted to switchgrass and 0% – 75% of irrigated fields were converted. 737 groundwater wells on irrigated rowcrop fields ceased groundwater pumping following the seeding of fields with switchgrass, reducing annual groundwater withdrawals by more than 139.1 billion l, or 5.6% of estimated annual groundwater withdrawals in the study area (Table 2).

In NRDs with a history of implementing irrigation limitations, there are 3,843 registered groundwater wells on irrigated rowcrop fields with a combined annual groundwater withdrawal potential of 671,539,624,144 l at an annual pumping time of 774 hours. All rowcrop fields converted to switchgrass under the Modest Change Scenario were in NRDs that had previously implemented irrigation limitations. The 64.2 billion l annual reduction in groundwater withdrawal under the Modest Change Scenario represents a 9.6% decrease in withdrawals in NRDs that have previously implemented irrigation limitations (Table 3). Under the Extreme Change Scenario, the conversion of rowcrop fields to switchgrass was not restricted to NRDs where irrigation limitations have been implemented in the past. Restricting switchgrass conversion to NRDs that have previously implemented limitations resulted in the cessation of pumping on 679 groundwater wells and reduced annual withdrawals by more than 128.5 billion l, or 19.1% of the current estimated withdrawals in the area (Table 3).

DISCUSSION

Effective groundwater conservation actions in the Great Plains will remain important for societies, the environment and agriculture in the future, especially under projected future climatic changes. The adoption of switchgrass as a bioenergy crop could create novel opportunities for groundwater conservation under potential future climatic and agricultural policy changes. Replacing irrigated rowcrops with switchgrass on fields with marginally productive, drier soils and in areas where irrigation limitations have been implemented in the past could conserve groundwater while providing an alternative source of income for farmers, thereby making individual farming operations more resilient to agricultural policy changes and variations in commodity prices.

Under the Modest Change Scenario, annual Rainwater Basin groundwater withdrawals decreased by 2.6% within 40 km ethanol plant service areas, whereas withdrawals decreased by 5.6% under the Extreme Change Scenario. While the conversion to switchgrass does not drastically reduce basin-wide groundwater withdrawals, withdrawal reductions in NRDs where additional irrigation limitations are most likely to be implemented were 9.6% under the Modest Change Scenario and 19.1% under the Extreme Change Scenario. These reductions are comparable to the 20% reduction goals recently identified by various Nebraska NRDs with over–appropriated water resources (Supalla 2010; Hilger 2010; Middle Republican NRD 2011).

Reduced precipitation and elevated evapotranspiration (ET) rates associated with climate change could increase crop water use requirements and well pumping time on rowcrop fields; thereby offsetting potential reductions in groundwater withdrawals. Furthermore, switchgrass stands may require some irrigation under warmer and drier conditions, especially as plant root systems develop during the establishment year. Nevertheless, replacing irrigated rowcrop fields with switchgrass could reduce groundwater withdrawals from what they would be if all fields remain in rowcrop production. Furthermore, if switchgrass stands are irrigated, groundwater withdrawals are likely to be less than what is required for rowcrop production.

Sustainable groundwater use is critical to continued agricultural productivity and sustainability in the Great Plains. Drought tolerant, locally adapted biofuel feedstocks like switchgrass could decrease the dependence of agricultural communities on groundwater irrigation for crop production. Potential groundwater withdrawal reductions under landuse change scenarios illustrate the ability of switchgrass to contribute to groundwater conservation goals in water–stressed agricultural landscapes.

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TABLES AND FIGURES

Table 1: List of 24 Rainwater Basin rowcrop field marginality classes and percentages of marginality classes converted to switchgrass under three landuse change scenarios. Conversion percentages of 0, 25, 50, 75 or 100 were assigned to marginality classes, according to the number of marginal criteria fields composing the class satisfied. The intensity of climatic change and accompanying irrigation limitations increases from the Limited Change Scenario to the Modest Change Scenario and from the Modest Change Scenario to the Extreme Change Scenario.

Landuse classification	**Limited**	**Modest Change**	**Extreme Change**
Pivot corners + poor soils + dry area	100%	100%	100%
Pivot corners + poor soils + wet area	75%	100%	100%
Pivot corners + good soils + dry area	75%	100%	100%
Pivot corners + good soils + wet area	50%	75%	100%
Small dryland fields + poor soils + dry area	100%	100%	100%
Small dryland fields + poor soils + wet area	75%	100%	100%
Small dryland fields + good soils + dry area	75%	100%	100%
Small dryland fields + good soils + wet area	50%	75%	100%
Large dryland fields + poor soils + dry area	25%	50%	75%
Large dryland fields + poor soils + wet area	0%	25%	50%
Large dryland fields + good soils + dry area	0%	25%	50%
Large dryland fields + good soils + wet area	0%	0%	25%
Gravity + poor soils + dry area + high risk of irrigation limitations	0%	50%	75%
Gravity + poor soils + wet area + high risk of irrigation limitations	0%	25%	50%
Gravity + good soils + dry area + high risk of irrigation limitations	0%	25%	50%
Gravity + good soils + wet area + high risk of irrigation limitations	0%	0%	25%
Gravity + poor soils + wet area + low risk of irrigation limitations	0%	0%	25%
Gravity + good soils + wet area + low risk of irrigation limitations	0%	0%	0%

Table 1 continued.

Landuse classification	Limited	Modest Change	Extreme Change
Pivots + poor soils + dry area + high risk of irrigation limitations	0%	25%	25%
Pivots + poor soils + wet area + high risk of irrigation limitations	0%	0%	25%
Pivots + good soils + dry area + high risk of irrigation limitations	0%	0%	0%
Pivots + good soils + wet area + high risk of irrigation limitations	0%	0%	0%
Pivots + poor soils + dry area + low risk of irrigation limitations	0%	0%	0%
Pivots + good soils + wet area + low risk of irrigation limitations	0%	0%	0%

Table 2: Potential annual groundwater withdrawal reductions and percent changes in withdrawals for the Rainwater Basin region within 40 kilometer ethanol plant service areas under the Limited Change, Modest Change and Extreme Change Scenarios. The intensity of climatic change, irrigation limitations and the number of rowcrop hectares converted to switchgrass increases from the Limited Change Scenario to the Modest Change Scenario and from the Modest Change Scenario to the Extreme Change Scenario.

Scenario	**Groundwater conserved**	**Percent change**
Limited Change	0 liters	0.0%
Moderate Change	64,215,102,696 liters	-2.6%
Extreme Change	139,172,305,576 liters	-5.6%

Table 3: Potential annual reduction in groundwater withdrawals and percent changes in withdrawals for Natural Resource Districts in 40 kilometer ethanol plant service areas of the Rainwater Basin region that have previously implemented limitations on irrigation under the Limited Change, Modest Change and Extreme Change Scenarios. The intensity of climatic change, irrigation limitations and the number of rowcrop hectares converted to switchgrass increases from the Limited Change Scenario to the Modest Change Scenario and from the Modest Change Scenario to the Extreme Change Scenario.

Scenario	**Groundwater conserved**	**Percent change**
Limited Change	0 liters	0.0%
Moderate Change	64,215,102,696 liters	-9.6%
Extreme Change	128,457,101,955 liters	-19.1%

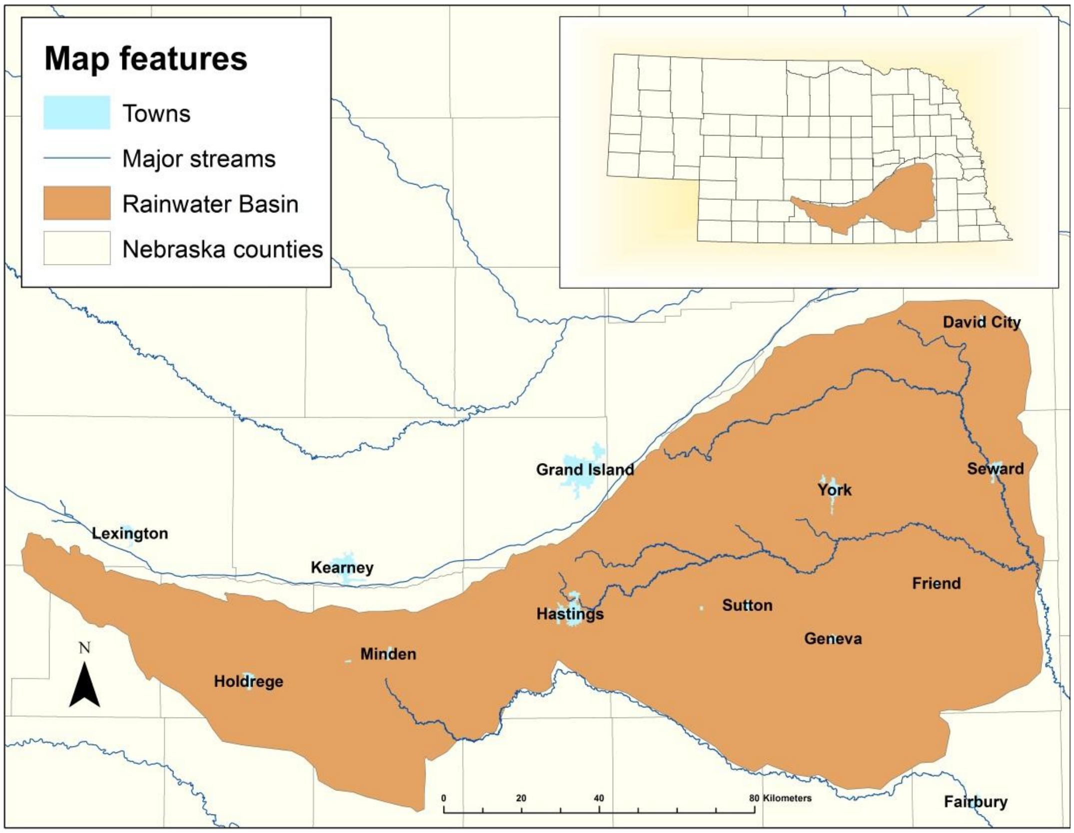

Figure 1: Location of the Rainwater Basin in south-central Nebraska, U.S.A, displaying Nebraska counties, major towns and rivers.

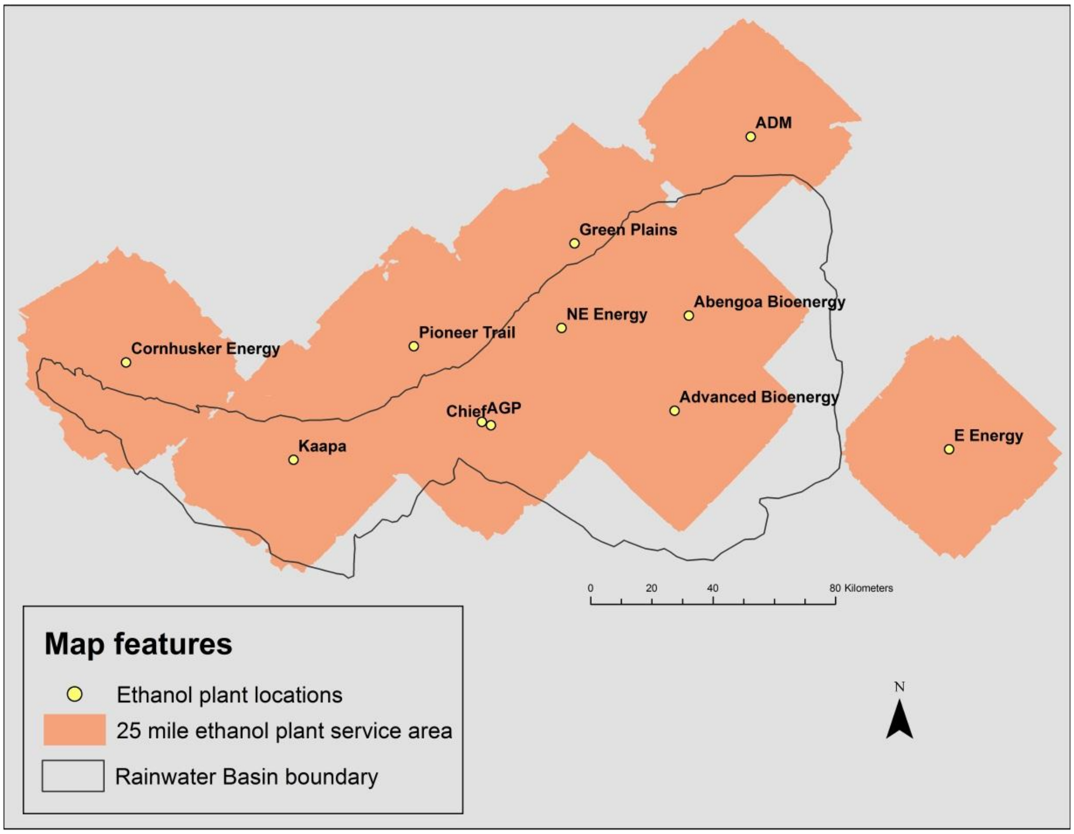

Figure 2: Locations and 40 kilometer network service areas of existing starch-based ethanol plants currently servicing the Rainwater Basin region.

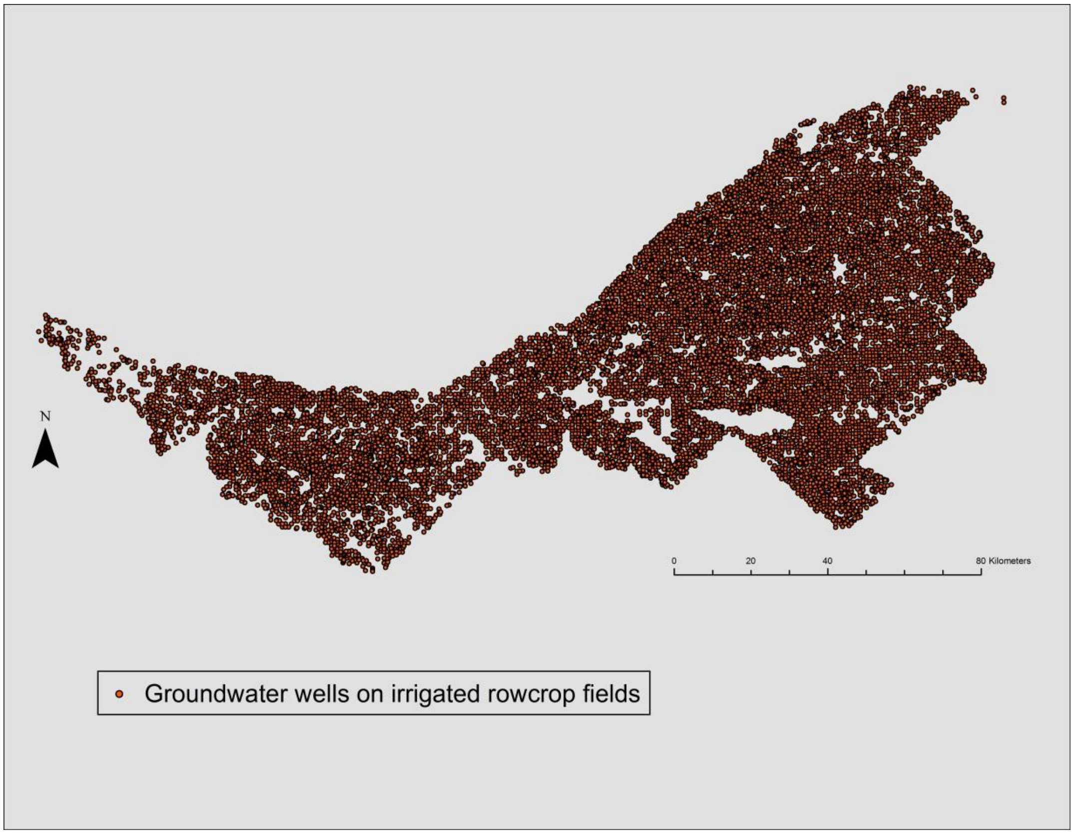

Figure 3: Registered Rainwater Basin groundwater wells located on irrigated rowcrop fields within 40 kilometer road network service areas of existing ethanol plants.

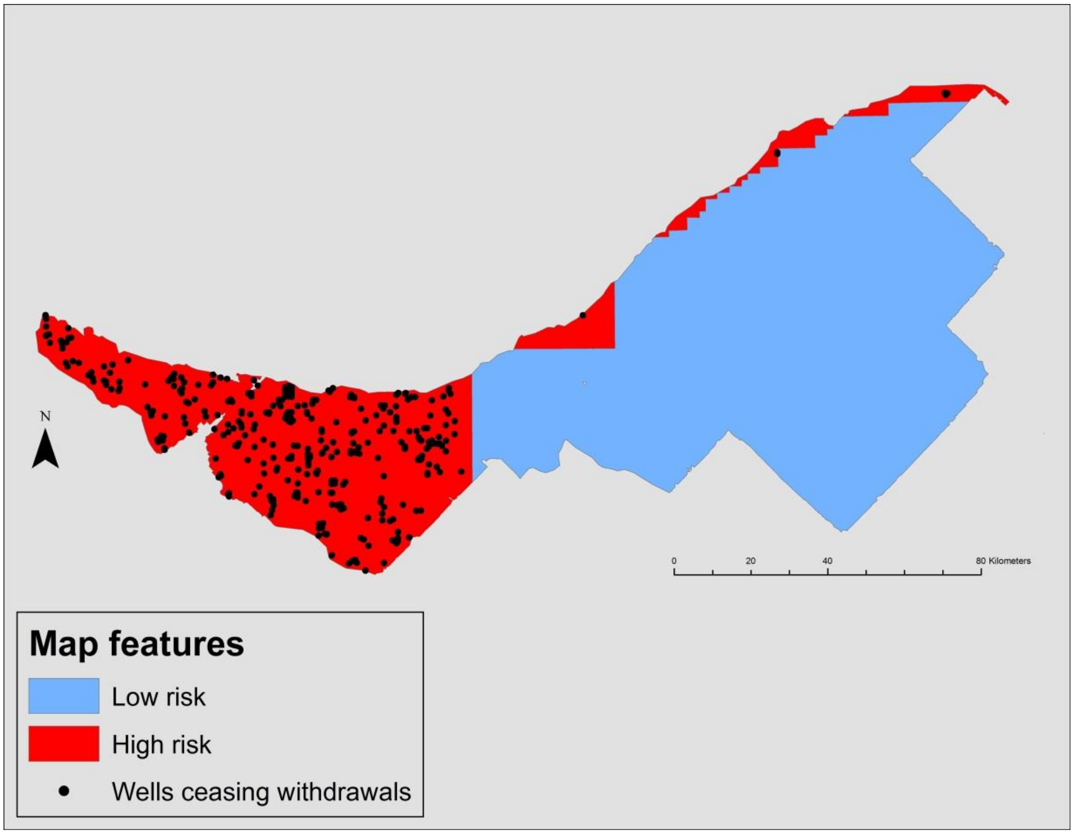

Figure 4: Registered Rainwater Basin groundwater wells located on irrigated rowcrop fields within 40 kilometer road network service areas that ceased pumping following conversion of the field to bioenergy switchgrass under the Modest Change Scenario, which assumes some climatic changes and additional irrigation limitations.

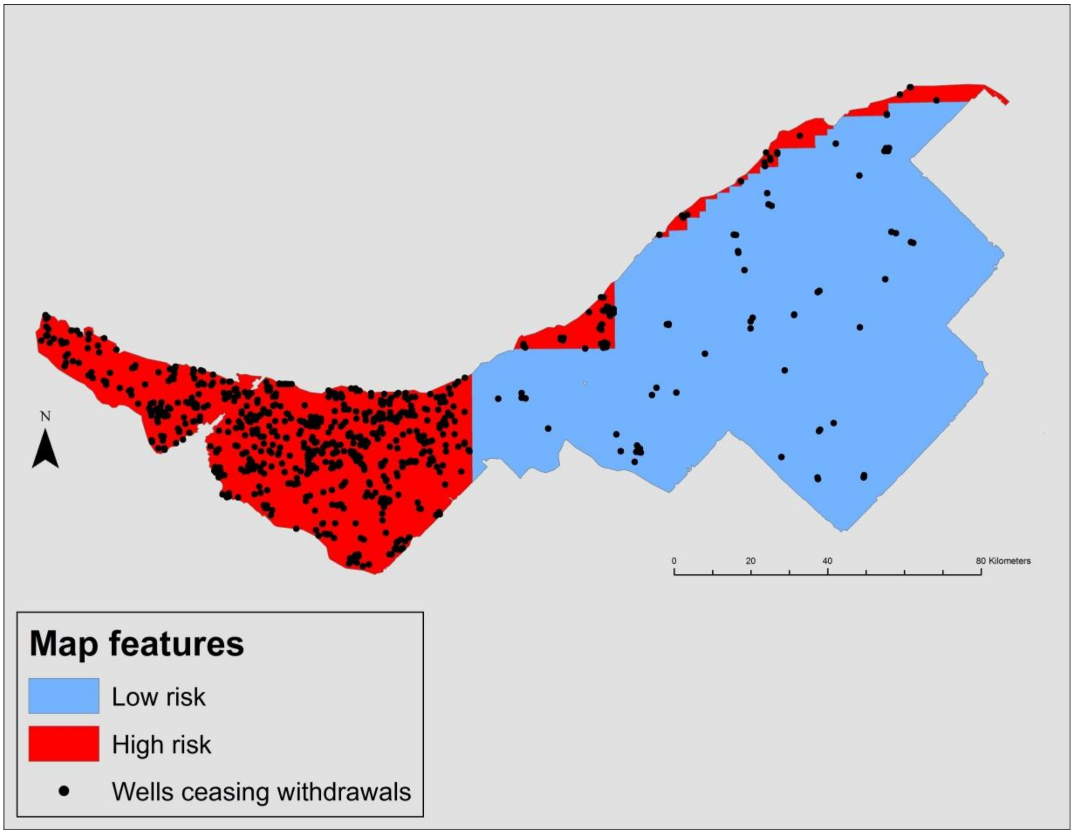

Figure 5: Registered Rainwater Basin groundwater wells located on irrigated rowcrop fields within 40 kilometer road network service areas that ceased pumping following conversion of the field to bioenergy switchgrass under the Extreme Change Scenario, which assumes extreme climatic changes and widespread irrigation limitations.

CHAPTER 6: PREDICTING VARIATION IN SPRINGTIME WETLAND OCCURRENCE AND FLOODED AREA IN NEBRASKA'S RAINWATER BASIN

ABSTRACT

The Rainwater Basin region of south-central Nebraska, U.S.A., is a critical stopover location for migratory waterfowl and shorebirds traveling along the Central Flyway migration route. Rainwater Basin wetlands serve as spring staging areas where birds rest, feed and pair before resuming northward migrations. Since 19th Century European settlement, approximately 90% of wetlands in the region have been destroyed through conversion to agriculture and remaining wetlands are degraded. Despite these losses, remnant and restored wetlands continue to provide critical stopover habitat to migratory waterfowl and shorebird populations. However, the ephemeral nature of the wetlands and localized nature of precipitation events causes the degree of springtime wetland inundation, and therefore the availability of stopover habitat, to vary between locations and years. Wetland inundation is believed to be driven by individual wetland characteristics, surrounding landuse and local weather events, but it is unclear which variables or combinations are most important. I used generalized linear mixed models in a multi–model inference framework to assess alternative models predicting variation in Rainwater Basin wetland occurrence and flooded area in 2004 and 2006 – 2009, according to local weather events, agricultural landuse practices and individual wetland characteristics. Rowcrop production, proximity to irrigation reuse pits, and increased wetland hydric footprint shape complexity negatively influenced wetland occurrence and flooded area. In general, greater autumn and winter precipitation totals increased the

probability of wetland occurrence and flooded area, whereas warmer autumn and winter maximum temperatures negatively influenced occurrence and flooded area. The effects of autumn precipitation and temperature were greater for wetland occurrence, and winter precipitation and temperature were more important for predicting flooded wetland area. Model predictions could help inform management actions aimed at providing adequate spring stopover habitat to migratory avifauna.

INTRODUCTION

Migration is a common life history strategy displayed across taxa (Dingle 1996). In a broad sense, migration entails moving between locations to secure better resources or conditions (Dingle 1996; Newton 2008). Despite its potential benefits, migration can be energetically expensive and dangerous, and tradeoffs between the benefits and costs associated with it impact the survival and reproductive success of migrating organisms (Lind & Cresswell 2006; Newton 2006, 2008). Additional stresses associated with rapidly changing climates and landscapes further complicate migratory timing and movements (Moeller et al. 2008; Fontaine et al. 2009), and affect the risks migratory species face.

Numerous avian species migrate semi-annually between southern wintering grounds and northern breeding grounds, with some traversing continents (Heglund & Skagen 2005; Newton 2008). Long–distance avian migrants are known to rely on the presence of quality breeding and wintering habitats (Robbins et al. 1989), but stopover habitat also influences their survival and reproductive success (Moore et al. 2005). Stopover habitat affords opportunities to rest and replenish energy reserves before resuming travel; and adequate caloric intake and rest at stopover locations helps promote improved body condition upon arrival at breeding grounds (Moore et al. 2005; Bishop & Vrtiska 2008). Conserving stopover habitat is therefore crucial to the continued viability of migratory bird populations, especially in altered landscapes where habitat is limited (Gibbs 2000).

The Rainwater Basin region of south-central Nebraska, U.S.A. is an important stopover location for migrating waterfowl and shorebirds. Shallow, rain–fed wetlands are spread across a landscape dominated by an agricultural matrix, providing critical wetland habitat for birds traveling through a bottleneck of the Central Flyway migration route (Gersib et al. 1991; RWBJV 1994; Bishop & Vrtiska 2008). Spring migration generally occurs between early February and mid–May, peaking in late February or early March (Bishop & Vrtiska 2008; Bishop 2010). At spring wetland staging sites, birds rest, pair, breed and build fat reserves necessary for the remaining journey and future reproduction by feeding on waste maize (*Zea mays*) grain, invertebrates and other vegetation in and around wetlands (Gersib et al. 1989; RWBJV 1994; Bishop & Vrtiska 2008). It is estimated that 7 – 14 million North American ducks and geese utilize Rainwater Basin wetland habitat annually, including 90% of continental white-fronted geese (*Anser albifrons*), 50% of continental mallards (*Anas platyrhynchos*), and 30% of continental northern pintails (*Anas acuta*) (Gersib et al. 1989; RWBJV 1994; Lagrange 2005). Various shorebird species also rely on Rainwater Basin wetlands for stopover habitat (Farmer & Parent 1999; Lagrange 2005).

Rainwater Basin wetlands are northeasterly to southwesterly oriented depressions created by eolian activity and lined with clay particles that retain water from precipitation and runoff events in surrounding closed watersheds (Smith 2003; Lagrange 2005). Soil survey maps from the early 20^{th} Century document the existence of as many as 1,000 major and 10,000 minor wetlands at the time of European settlement (Figure 2), less than 10% of which remain today (Figure 3) (Gersib 1991; Bishop & Vrtiska 2008). Technological advances and agricultural intensification throughout the 20^{th} Century led to wetland destruction and degradation through draining, development and conversion to agriculture (Gersib et al. 1989; Gersib 1991). Reductions in historical wetland area reduced the availability of stopover habitat and food resources for wetland dependent migratory birds (Bishop & Vrtiska 2008).

Wetlands are classified according to underlying hydric soils, which influence water retention and plant communities (Gersib et al. 1989; Gilbert 1989; RWBJV 1994). Massie soils underlie the deepest, semi-permanent wetlands which typically hold water year–round. Seasonal wetlands are underlain by Scott soils and inundated for the majority of the growing season, but do not generally pond water the entire year. Fillmore, Aquolls, Butler and Roscoe soils are associated with temporary wetlands, which are ephemeral in nature and only hold water for short time periods following major precipitation events. Temporary wetlands are often cultivated altering their hydrological and ecological structure (Gersib 1989; Gersib et al. 1989; RWBVJ 1994).

During wet periods, seasonal and temporary wetlands provide reliable habitat and nutrients for waterfowl and shorebirds, whereas only semi-permanent wetlands are dependable during drier periods (Gersib et al. 1989). Seasonal and temporary wetlands are shallower than semi-permanent wetlands, and because they warm faster and provide food sources earlier in the spring (Krapu 1974), are generally preferred by waterfowl when available (Kantrud & Stewart 1977). The utilization of different wetland types at different times of year and under different weather conditions illustrates the importance of the entire wetland complex for migratory avian species (Gersib et al. 1989).

The ephemeral nature of Rainwater Basin wetlands is associated with variation in flooded wetland area within and between years (Gersib et al. 1989). Springtime wetland stopover habitat is crucial for migratory waterfowl and shorebirds (Bishop & Vrtiska 2008). However, stopover habitat is generally less available in dry periods (Bishop

2010), and the detrimental impacts of drought on migratory waterfowl and shorebirds have likely been compounded by habitat loss associated with the transformation of the prairie landscape to agriculture (Gersib 1991; Bishop & Vrtiska 2008). In dry years, wildlife managers attempt to compensate for reduced wetland habitat availability by pumping groundwater into wetlands prior to avian migration (Bishop 2010). However, predicting the future availability of wetland habitat is difficult because of anthropogenic alterations to wetland hydrologic cycles and uncertainties over future weather events, and the relationships among hydrology and intrinsic and extrinsic variables affecting ponding.

Springtime wetland flooding is driven by various hydrologic factors, including precipitation, runoff, evapotranspiration (ET) and infiltration (Wilson 2010; Johnson et al. 2011). However, ways in which these factors manifest themselves through, and interact, with weather events, landscape alterations and individual wetland characteristics are not fully understood. Local weather events are important drivers of wetland inundation; however, weather patterns throughout the Great Plains are highly variable and difficult to predict (Weaver & Albertson 1956; Forrest et al. 2004). Wetlands typically fill with water following major precipitation events, when runoff is generated from rainfall or snowmelt (Lagrange 2005; Wilson 2010). Hydric wetland soils seal water in wetlands by preventing infiltration into the ground (Starks 1984; Gersib 1989). Springtime in the Rainwater Basin is characterized by snowmelt and increased thunderstorm activity (Keech & Dreeszen 1959; Kelly et al. 1985; Wilson 2010), which tend to fill wetlands. Less precipitation and warmer temperatures in summer months (Keech & Dreeszen 1959) dry wetlands via evapotranspiration and infiltration until they are refilled following precipitation events (Wilson 2010).

The pre-wetted condition of wetland soils can also influence wetland inundation by increasing or decreasing infiltration rates. During extended dry periods, desiccation cracks form in the dry clay pan of wetlands (Bagarello et al. 1999; Wilson 2010). These cracks break the seal of hydric soils and allow for rapid infiltration of water into the ground (Wilson 2010). Rapid infiltration into cracks can prevent major precipitation events and ensuing runoff from filling wetlands, and additional precipitation and runoff may be necessary for wetland inundation once the initial precipitation event has sealed the cracks (Wilson 2010). Similarly, the presence of a frost layer in the winter and early spring can reduce infiltration rates by separating wetland water from dry, underlying soils layers (Wilson 2010). If a frost layer is present, wetlands may better retain water collected from winter and spring precipitation, runoff and snowmelt.

Rowcrop production decreases wetland size and water retention capabilities by increasing infiltration and siltation (Gersib 1989; Gilbert 1989). Rowcrop irrigation is often associated with landscape alterations like land leveling, wetland draining and the excavation of irrigation reuse pits, all of which negatively impact hydrologic cycles within watersheds (Gersib 1989; Gilbert 1989; Lagrange 2005). Irrigation reuse pits are typically situated at the lowest elevations on properties and concentrate excess irrigation runoff for future use in gravity irrigation systems (Smith 2003; Lagrange 2005). Although they generally retain water throughout the year, irrigation reuse pits provide fewer benefits to wildlife than natural wetlands and reduce water availability in watersheds by catching precipitation runoff that might otherwise fill wetlands (Haukos & Smith 2003).

Successfully predicting the occurrence and flooded area in Rainwater Basin wetlands at peak spring bird migration could assist managers in providing adequate stopover habitat to migratory waterfowl and shorebirds in years when wetland inundation is reduced. In this chapter, I utilize generalized linear mixed models in a multi–model inference framework to sift amongst competing predictive models explaining annual variation in springtime Rainwater Basin wetland occurrence and flooded area. Models were put at risk with data from 2004 and 2006–2009 and included variables capturing local weather events, agricultural landuse practices and individual wetland characteristics. Generalized linear mixed models were used to assess variability in wetland occurrence (presence/absence) and linear mixed models were utilized to examine variation in the flooded area of wetlands that did occur.

METHODS

STUDY AREA

The Rainwater Basin is a watershed of the Greater Platte River Basins (USGS 2009), covering 15,800 km^2 in all or portions of 21 south–central Nebraska counties (LaGrange 2005) (Figure 1). In this intensively farmed region, the majority of the agricultural landscape is utilized for maize and soybean (*Glycine max*) production, although small grain farming and cattle ranching are also conducted (Gilbert 1989; Bishop & Vrtiska 2008). Irrigation and dryland farming are both common, although irrigated fields generally produce better yields than dryland fields. Hundreds of remnant and restored rain–fed wetlands also occur, providing wildlife habitat, environmental services and recreational opportunities (Lagrange 2005).

DATA SOURCES

Annual Habitat Survey (AHS) data from 2004 and 2006 – 2009 was provided by the Rainwater Basin Joint Venture (RWBJV). AHS data is collected annually at peak spring bird migration in late February or early March via aerial photography and is used to quantify wetland inundation throughout the Rainwater Basin (Bishop 2010). The AHS has been conducted annually since 2004, with the exception of 2005. The RWBJV also provided the contemporary wetland hydric footprint, irrigation reuse pit location and agricultural irrigation type data. Contemporary wetlands are defined here as wetlands determined by RWBVJ staff to be functioning, based on 2004 – 2007 AHS data (RWBJV 2010). Weather data was downloaded in the form of .tiff raster images from the Yellowstone Ecological Research Center's Customized Online Aggregation and Summarization Tool for Environmental Rasters (COASTER) website (http://coasterdata.net/Default.aspx). Geographic coordinates were used to restrict requested weather raster images to the Rainwater Basin region.

PROJECTIONS AND TRANSFORMATIONS

COASTER weather data raster images were obtained in the Lambert azimuthal equal area (LAEA) projection, and a LAEA projection file was included with each requested COASTER raster image. All Rainwater Basin wetland and AHS data layers were obtained in the North American Datum 1983 Universal Transverse Mercator Zone 14 North (NAD 1983 UTM 14N) projection. Reprojecting weather raster images into NAD 1983 UTM 14N entailed changing the projection of each weather raster to ‘undefined’, projecting the rasters with the LAEA projection file that was provided with the weather raster files, creating a custom geographic transformation for converting between LAEA and NAD 1983 UTM 14N, and carrying out the reprojection.

AGRICULTURAL LANDCOVER

Some wetlands are located within rowcrop fields, while others occur inside remnant grasslands or conservation properties. I classified rowcrop fields in which wetlands were located according to irrigation system types, which were center-pivot irrigated, gravity irrigated or dryland. A center-pivot is a large sprinkler system typically anchored at a field center point and connected to a groundwater well. Groundwater is pumped through a pipe extending from the center point to the least distant field perimeter, with multiple two–wheeled moving towers supporting the pipe along its extent. As the center-pivot moves in a circular motion around the field, sprinklers connected to the pipe release water to the soil surface. Gravity irrigation consists of a temporary lateral irrigation pipe extending along the field edge with the highest altitude and perpendicular to the direction of crop rows. Water is released from the pipe into furrows between crop rows and is moved by gravity toward the opposite end of the field. Dryland fields are not irrigated by any means, and landscape modifications on dryland fields tend to be less severe than on gravity and center–pivot irrigated fields. Agricultural irrigation type data was converted from vector to raster format to facilitate data extraction at each wetland location.

Gravity irrigation systems are generally associated with landscape alterations such as draining, land leveling, and the excavation of irrigation reuse pits, due to the need for uniform and gentle slopes for moving water through crop rows. Wetland proximity to the nearest irrigation reuse pit was determined using Euclidian (straight–line) distance. Straight–line distance to the nearest irrigation pit was calculated for the entire Rainwater Basin landscape and output as continuous raster image. Although water does not typically move through watersheds in straight lines, Euclidian distance provides a simple means of determining the proximity of wetlands to irrigation reuse pits that may be influencing hydrologic cycles in watersheds.

WEATHER DATA

Weather data for the year previous to each AHS survey was divided into four time periods: April 1st – June 30th (spring), July 1st – September 30th (summer), October 1st – November 31st (autumn) and December 1st – March 31st (winter). Changes between time periods are specified at proximate dates when Rainwater Basin weather patterns generally shift. In spring, warming temperatures produce snowmelt and localized thunderstorms. Warmers temperatures and decreasing precipitation characterize summer months, and autumn is accompanied by falling temperatures and variable precipitation. Temperatures are coolest in winter and may be accompanied by rain, or snow and ice accumulation.

Precipitation

Both total precipitation and major precipitation events were used to assess the influence of precipitation within each season on springtime wetland inundation the following year. The format of the precipitation data did not allow for the assessment of individual major weather events, but instead consisted of the number of 24 hour periods in which precipitation exceeded a specified threshold value. Therefore, multiple precipitation events could be captured in single days, or single precipitation events could stretch across multiple days. Threshold values for determining major precipitation events were 50.8 millimeters (2 inches) for spring and summer, and 25.4 millimeters (1 inch) for autumn and winter. 50.8 millimeters is recognized as the approximate quantity of precipitation necessary to generate runoff that fills wetlands (Randy Stutheit, Nebraska Game and Parks Commission, personal communication). However, there were few autumn or winter days in any of the five study years when more than 50.8 millimeters of precipitation was received; therefore the threshold for major precipitation events during autumn or winter was specified at 25.4 millimeters. The format of precipitation data did not allow for differentiation between forms of precipitation, so it was unclear if winter and spring precipitation events represented rain or snowfall.

Temperature

Minimum and maximum temperature data were collected to assess the impact of temperatures on springtime wetland inundation. Temperature can affect ET and frost layers in wetland soils. Mean minimum and maximum temperatures were collected for each time period, in addition to the number of winter days that the maximum temperature was < 0 C and the days of the year when minimum temperatures were < 0 C. A greater number of winter days when the temperature was < 0 C may promote the development of a frost layer and reduce evaporation from wetlands. Similarly, early frosts may be associated with long and more severe winters with frost layers and less evaporation, and later warming dates may prevent the thawing of frost layers and preserve accumulated moisture as snow. Furthermore, interactions between precipitation and temperature could influence wetland inundation differently than individual variables might. The presence of precipitation in wetland soils could promote the formation of frost layers, and frost layers may retain water in wetlands from multiple precipitation events.

Vapor pressure deficit

Vapor pressure deficit (vpd) is a measurement used to quantify the drying power of air (YERC 2011). Vpd is calculated by subtracting the quantity of water in the air at a given temperature from the quantity of water necessary to fully saturate the air at that temperature (YERC 2011). Low vpd values indicate that the air is relatively saturated and that the potential for drying is minimal, whereas higher vpd values suggest there is potential for the air to absorb large quantities of additional moisture.

WETLAND CHARACTERISTICS

Hydric soil type and shape complexity were additional characteristics used to describe wetlands. Hydric soils influence infiltration of water in wetlands (Gersib et al. 1989; Wilson 2010). Hydric soils were grouped into six soil series, according to their water retention capabilities. Massie soils have the greatest water retention abilities, and are followed by Scott, Fillmore, Aquolls, Butler and Roscoe, respectively. The classification of Rainwater Basin wetlands into semi–permanent, seasonal or temporary wetland types is related to soil series, with semi–permanent wetlands underlain by more hydric soils than seasonal wetlands and seasonal wetlands underlain by more hydric soils than temporary wetlands. Wetland shape also influences infiltration. More complexly shaped wetlands generally have more surface area exposed to less hydric soils on wetland perimeters (Wilson 2010). I used the perimeter to area ratio of wetland hydric footprints to quantify wetland shape complexity.

DATA EXTRACTION AND COMPILATION

The extent of the AHS increased from 2004 – 2006 and from 2006 – 2007, and then remained constant from 2007 – 2009. The 2004 AHS extent was enveloped by the AHS extents from 2006 – 2009; therefore, only contemporary wetlands within the 2004 AHS extent were considered in this study. This restriction ensured that the same wetlands were surveyed in each of the five study years. All contemporary wetland hydric soil polygons within the restricted shapefile were converted to point features. Converting wetland polygons to point features specified a single geographic location for each wetland and facilitated the extraction of data for that location. The resulting contemporary wetlands point shapefile was copied four times, producing identical shapefiles composed of individual wetland identification numbers and wetland types. Scripts composed in Python (http://www.python.org) were used to insert and populate fields in each of the shapefile attribute tables that specified a specific study year. Additional Python scripts were used to extract weather, agricultural irrigation type, hydric soil series type and proximity to irrigation reuse pit raster data from each contemporary wetland point location in each study year to the respective contemporary wetland points shapefile attribute table.

Wetland flooded area polygons from each year's AHS shapefile were also converted from polygon to point features. Python scripts were used to insert and populate fields in each AHS point shapefile attribute table that specified the occurrence (presence/absence) of wetlands in individual years. In the wetland occurrence field, a value of one signified that the hydric wetland footprint contained at least some water during the AHS survey in the year, whereas a value of zero signified that the wetland was dry. Because only wetlands that did occur are listed within AHS attribute tables, the wetland occurrence fields consisted entirely of presence values. Absence values were generated following the combination of AHS point shapefiles with the contemporary wetlands points shapefiles from respective years.

AHS point shapefiles from the study years were merged with contemporary wetland point shapefiles from respective years, using wetland identification as the common field between shapefiles. Wetland occurrence and flooded area were contained in the AHS point shapefiles, and survey year, weather data, hydric soil series type, wetland type, agricultural irrigation type and proximity to irrigation reuse pits were contained in the contemporary wetlands shapefile. The results of the merges were five final shapefiles listing the occurrence and flooded area of all contemporary wetlands within the 2004 AHS survey extent, in addition to various other characteristics at each wetland point location for each of the survey years. For analysis of wetland occurrence, the attribute tables of each shapefile were opened as database files in Microsoft Excel, combined into a single file and saved as a comma delimited (.csv) file. The analysis of flooded wetland area required only the flooded areas of wetlands that did occur be included in the database; therefore, all wetland footprints that did not pond any water were removed and the resulting file was saved in .csv format.

STATISTICAL ANALYSES

Multi–model inference

Multi–model inference is a useful approach for assessing the influence of a variety of predictor variables on a response variable (Anderson 2008). Unlike traditional hypothesis testing, which compares a null hypothesis with an alternative hypothesis to detect statistically significant differences in parameters, multi–model inference ranks multiple *a priori* hypotheses from best to worst, according to support given data (Anderson et al. 2000). I developed suites of *a priori* hypotheses for explaining variations in annual springtime wetland occurrence and flooded area. I relied on existing scientific literature and expert opinion regarding Rainwater Basin wetland hydrology to inform hypothesis development. Competing hypotheses, or models, were compared using Akaike's Information Criterion, corrected for small sample size (AICc) in Program R (R Development Core Team 2012). AICc is used to rank models according to their strength of evidence for explaining variation in the response variable (Burnham & Anderson 2004); however, including too many parameters in a model risks assigning residual variance to model parameters unduly, thereby overfitting the model (Anderson 2008). To promote parsimony, or a balance between underfitting and overfitting models, a penalty is applied during model ranking for each additional parameter included in a model (Burnham & Anderson 2004).

Model construction

Competing generalized linear mixed models and linear mixed models were used to explain variation in springtime wetland occurrence and flooded area, respectively. A mixed model consists of a fixed effects structure and random effects structure (Zuur et al. 2007). The fixed effects structure is comprised of fixed effects, or specific variables about which inferences wish to be made, whereas the random effects structure is composed of random effects, or variables by which the model intercept and fixed effects variables may vary. The incorporation of random effects into competing models helps explain additional variation in the dataset while conserving degrees of freedom that would otherwise be used to generate coefficient estimates for each level of each random effects variable. As a result, inferences about random effects variables can only be made for the entire population of the variable (Zuur et al. 2007). In this study, individual wetlands and years were considered random effects. Considering wetlands and years as random effects allowed the model intercept and weather related variables to vary between wetlands and/or years.

In the first step, a set of competing models was constructed *a priori* and compared to determine the optimum random effects structure for explaining wetland occurrence or flooded area. Each model possessed the same global fixed effects structure and a unique random effects structure. The model with the lowest AICc score and greatest AICc weight was determined to possess the optimum random effects structure. In the second step, another *a priori* set of competing models was constructed and compared, with each model containing unique fixed effects structures and the optimum random effects structure identified in the first step. This second model set included a null model and global model. The model with the lowest AICc score and greatest AICc weight was identified as the best supported model, and other models with a weight at least 10% that of the best supported model were included with the best supported model in the confidence set. This liberal cutoff for inclusion in the confidence set was similar to that suggested by Royall (1997). Coefficient estimates for parameters in the confidence set were averaged and combined into a final model for predicting springtime wetland occurrence or flooded area. Model averaged coefficient estimates were obtained by averaging a parameter coefficient with the coefficient estimates from other models containing the parameter of interest.

Wetland occurrence

Generalized linear mixed models were used to assess variation in contemporary wetland occurrence in 2004 and 2006 – 2009. The response variable, wetland occurrence, was binomially distributed, with each wetland being either present or absent each year. If any flooded area was detected in wetland hydric footprints during AHS surveys, the wetland was considered present; if no flooded area was detected, it was considered absent. 1,359 wetlands were surveyed in each of the five years, resulting in a total sample size of 6,795 wetlands. The assumptions of the linear model were tested by comparing the fitted values of the global model with the residuals of the global model. Correlations between explanatory variables were examined visually with plots and numerically with correlations. When two explanatory variables had a correlation greater than 0.5, the variable least correlated with wetland occurrence was removed from the analysis. Because the goal of this analysis was to develop a predictive model explaining the effects of wetland characteristics, surrounding agricultural landuse, and weather related events on wetland occurrence, these parameters were treated as fixed effects, whereas individual wetlands and years were treated as random effects. Treating wetlands and years as random effects conserved degrees of freedom and allowed for inferences to be made concerning the drivers of occurrence and flooded area for the entire population of Rainwater Basin wetlands in all years (Zuur et al. 2007). All non–categorical parameter values were centered at zero and scaled to improve model fit and facilitate comparisons between the coefficient estimates of different parameters. Values were scaled by subtracting mean parameter values from individual parameter values and dividing the difference by the standard deviation of the parameter mean.

The two–step model selection process previously described was used to identify the best supported model explaining variation in springtime wetland occurrence. In the first step, a set of 26 competing models was constructed *a priori* and compared to determine the optimum random effects structure for explaining wetland occurrence. In the second step, another *a priori* set of 29 competing models was constructed and tested to determine the optimum fixed effects structure.

Flooded wetland area

Linear mixed models were used to examine variation in springtime flooded area for contemporary wetlands in the five study years. Because only present wetlands have any flooded area, the response variable was bounded at zero on the lower end. To avoid the inflation of wetland flooded area with zero values, all wetland absences were removed from the dataset, reducing the sample size to 4,150 flooded wetlands during the five years. Correlations between explanatory variables were assessed with plots and absolute correlations, and variables that were strongly correlated with other variables but weakly correlated with flooded wetland area were removed from the analysis. Individual wetlands and years were treated as random effects and all other explanatory variables were treated as fixed effects. All non–categorical explanatory variables were centered at zero and scaled prior to analysis. The assumptions of the linear model were tested by plotting the fitted values of the global model against the residual values of the global model. Wetland flooded area was log10 transformed to satisfy the assumption of a normally distributed response variable in linear regression. The transformation normalized the wetland flooded area; however, some slight heterostochasticity was still evidenced in plots.

The two–step model selection process previously described was used to identify the best supported model explaining variation in springtime wetland occurrence. In the first step, a set of 26 competing models was constructed *a priori* and compared to determine the optimum random effects structure for explaining wetland occurrence. In the second step, another *a priori* set of 25 competing models was constructed and tested to determine the optimum fixed effects structure.

Management action deadlines

Although winter weather variables may be important drivers of springtime wetland occurrence and flooded area, managers may be required to take actions aimed at increasing the availability of wetland stopover habitat prior to the onset of winter. Under these circumstances, predictive models that omit the influences of winter weather parameters may provide the most useful information concerning wetland occurrence and flooded area the following spring. I developed additional predictive models explaining variation in wetland occurrence and flooded area that excluded winter parameters.

Wetland characteristics, surrounding agricultural landuse, and spring, summer and autumn weather related parameters were included in these models. The timeframe from which weather data was collected was April 1^{st} – November 30^{th}. The procedures for identifying the best supported models, constructing confidence sets and conducting model averaging were identical to those for the previously described models that did incorporate winter weather parameters.

The removal of winter weather parameters reduced the number of possible parameter combinations in model sets; and therefore, fewer competing models were tested in the analyses that omitted winter weather parameters. A total of 28 competing models in two different sets were constructed *a priori* and compared to select the optimum random effects and fixed effects structures for predicting wetland occurrence. To identify the best supported model for predicting wetland flooded area without winter weather parameters, a total of 40 competing models in two different sets were constructed *a priori* and compared to select the optimum random effects and fixed effects structures for making predictions.

RESULTS

WETLAND OCCURRENCE

Models including winter weather parameters

For predicting springtime wetland occurrence with winter weather parameters, the optimum random effects structure allowed the model intercept to vary between wetlands and the model intercept and mean winter vapor pressure deficit to vary between years (Table 1). The best supported model (wi = 0.43) was: Wetland occurrence = 2.17 * intercept + 0.95 * semi–permanent wetland – 1.19 * temporary wetland – 0.71 * center–pivot irrigation – 0.81 * dryland – 0.41 * gravity irrigation – 0.18 * Euclidian distance to nearest irrigation reuse pit – 1.47 * hydric footprint perimeter to area ratio + 0.38 * total summer precipitation + 1.14 * total autumn precipitation – 1.00 * mean autumn maximum temperature – 0.25 * mean winter vapor pressure deficit (Table 2).

Five models were included in the confidence set, one of which was the global model (Table 2). Model averaged coefficient estimates were generated for all parameters in the confidence set and were used to construct a final model (Table 3). The final model for predicting wetland occurrence was: Wetland occurrence = 2.14 * intercept + 0.95 * semi–permanent wetland – 1.19 * temporary wetlands – 0.77 * center–pivot irrigation – 0.80 * dryland – 0.41 * gravity irrigation – 0.77 * Euclidian distance to nearest irrigation reuse pits – 1.47 * hydric footprint perimeter to area ratio – 0.001 * number of spring days with more than 50.8 mm in precipitation + 0.37 * total summer precipitation + 1.13 * total autumn precipitation – 1.00 * mean autumn maximum temperature – 0.32 * mean winter vapor pressure deficit – 0.01 * first autumn/winter date with minimum temperature < 0 degrees Celsius.

Models excluding winter weather parameters

For predicting springtime wetland occurrence with winter weather parameters, the optimum random effects structure allowed the model intercept to vary between wetlands and the model intercept and total autumn precipitation to vary between years (Table 4). The best supported model (*wi* = 0.55) was the global model (Table 5). Two models were included in the confidence set and were used to create a final model for explaining springtime wetland occurrence without winter weather parameters (Table 6). The final model was: Wetland occurrence = 2.87 * intercept + 0.96 * semi–permanent – 0.98 * temporary wetland type – 0.70 * center–pivot irrigation – 0.77 * dryland – 0.42 * gravity irrigation – Euclidian distance to nearest irrigation reuse pit – 1.54 * hydric footprint perimeter to area ratio – 0.08 * number of spring days with > 50.8 mm in precipitation + 0.39 * total summer precipitation + 0.76 * total autumn precipitation – 1.20 * mean autumn maximum temperature.

FLOODED WETLAND AREA

Models including winter weather parameters

For predicting springtime wetland occurrence with winter weather parameters, the optimum random effects structure allowed the model intercept to vary between wetlands and the model intercept and total summer precipitation to vary between years (Table 7). The best supported model (*wi* = 0.43) was: Flooded wetland area = 0.07 * intercept + 0.70 * semi–permanent wetland type – 0.36 * temporary wetland type – 0.17 * center–pivot irrigation + 0.10 * dryland – 0.38 * gravity irrigation + 0.07 * Euclidian distance to nearest irrigation reuse pit – 0.14 * hydric footprint perimeter to area ratio + 0.01 * number of spring days with more than 50.8 mm of precipitation + 0.08 * total summer precipitation + 0.04 * mean autumn vapor pressure deficit + 0.11 * total winter precipitation + 0.23 * number of winter days with maximum temperature < 0 degrees Celsius + 0.06 * first winter/spring date with minimum temperature > 0 degrees Celsius (Table 8).

Four models were included in the model set, one of which was the global model (Table 8). Model averaged coefficient estimates were generated for all parameters in the confidence set and were used to construct a final model (Table 9). The final model was: Wetland flooded area = 0.07 * intercept + 0.70 * semi–permanent wetland type – 0.36 * temporary wetland type – 0.17 * center–pivot irrigation + 0.10 * dryland – 0.37 * gravity irrigation + 0.07 * Euclidian distance to nearest irrigation reuse pit – 0.14 * hydric footprint perimeter to area ratio + 0.07 * total summer precipitation + 0.11 total winter precipitation + 0.23 * number of winter days with maximum temperature < 0 degrees Celsius + 0.06 * first winter/spring data when minimum temperature > 0 degrees Celsius.

Models excluding winter weather parameters

For predicting springtime wetland occurrence without winter weather parameters, the best supported random effects structure allowed the model intercept to vary between wetlands and the model intercept and total summer precipitation to vary between years (Table 10). The best supported model (*wi* = 0.46) was: Wetland flooded area = 0.08 * intercept + 0.70 * semi–permanent wetland type – 0.37 * temporary wetland type – 0.17 * center–pivot irrigation + 0.10 * dryland – 0.36 * gravity irrigation + 0.07 * total summer precipitation – 0.09 * number of autumn days with more than 25.4 mm of precipitation – 0.16 * autumn vapor pressure deficit (Table 11).

Four models were included in the model set, one of which was the global model (Figure 11). Model averaged coefficient estimates were generated for all parameters in the confidence set and were used to construct a final model (Table 12). The final model was: Wetland flooded area = 0.08 * intercept + 0.70 * semi–permanent wetland type –

0.37 * temporary wetland type – 0.17 * center–pivot irrigation + 0.10 * dryland – 0.36 * gravity irrigation + 0.07 * Euclidian distance to nearest irrigation reuse pit – 0.13 * hydric footprint perimeter to area ratio – 0.01 * total number of spring days with more than 50.8 mm of precipitation + 0.07 * total summer precipitation – 0.09 * number of autumn days with more than 25.4 mm of precipitation + 0.02 * mean autumn minimum temperature – 0.16 * mean autumn vapor pressure deficit.

DISCUSSSION

The hydrology of Rainwater Basin wetlands is complex and influenced by wetland characteristics, anthropogenic landscape alterations and seasonal weather events. Although weather events produce the water that inundates wetlands, the ability of wetlands to retain water throughout the year and provide spring stopover habitat for migratory avifauna is influenced by both weather and non–weather factors. Anthropogenic alterations influence the water retaining capabilities of wetlands, and because the degree of alteration varies between wetlands, the occurrence and flooded area of wetlands with similar characteristics experiencing the same weather events may vary as well.

Semi–permanent wetlands more commonly occur and contain more flooded area than seasonal wetlands, which are more common than temporary wetlands; therefore, the restoration and conservation of semi–permanent and seasonal wetlands may be most beneficial to migratory bird populations, due to their increased likelihood of inundation during the spring months. Because many temporary wetlands are farmed in drier years and are often located in agricultural fields (Gersib et al. 1989), restoring them could be more difficult.

Surrounding agricultural landuse is also a driver of wetland occurrence and flooded area. In general, wetlands embedded in agricultural fields are likely hold water less often and contain less flooded area than wetlands surrounded by alternative landuses, likely due to landscape alterations associated with rowcrop production. One exception existed in that wetlands in dryland fields contained more flooded area than wetlands in alternative landuses. Landscape alterations tend to be less severe on dryland fields than on center–pivot or gravity irrigated fields; therefore, the hydrologic cycles of wetlands embedded within dryland fields may be more intact than those of wetlands in irrigated fields, and could allow wetlands within dryland fields to retain water for longer time periods. Wetlands with fewer and less intensive landscape alterations in their immediate vicinity are likely to serve as the most reliable sources of stopover habitat for migratory birds.

Wetland shape complexity had strongly negatively associated with wetland occurrence (Figure 5). Complexly shaped wetlands have more surface area contacting wetland edges, which are generally associated with less hydric soils than wetland interiors (Starks 1984). Thus, complexly shaped wetlands may lose more water through infiltration than compact wetlands and may occur less frequently and contain less water. In wetland restoration efforts, promoting more compact wetland shapes could improve future wetland water retention capabilities.

Increased summer precipitation was associated with more frequent wetland occurrence (Figure 5) and more flooded wetland area (Figure 7). The saturation of wetland soils during precipitation events could prevent the development of desiccation cracks in wetland soils and seal cracks that formed in previous dry periods. Subsequent precipitation events may be more likely to fill wetlands if desiccation cracks are not present. Furthermore, more precipitation could be associated with cooler summer temperatures, which would reduce wetland water loss through ET and slow the formation of desiccation cracks.

More autumn precipitation greatly increased the likelihood of springtime wetland occurrence (Figure 6), whereas greater mean autumn maximum temperatures strongly decreased the frequency of wetland occurrence (Figure 6). Autumn precipitation and temperature could be important for determining the condition of wetland soils prior to the onset of winter. If wetland soils are saturated during periods with freezing maximum temperatures, a frost layer could develop and be maintained throughout winter, providing that freezing temperatures persist. Frost layers could promote the retention of water in wetlands following winter precipitation and snowmelt by reducing infiltration rates (Wilson 2010). Wetter and cooler autumns could also be associated with wetter and cooler winters, which could decrease evaporation rates and increase the likelihood of a frost layer developing; thereby promoting wetland water retention and occurrence the following spring.

Greater winter precipitation and number of days when the maximum temperature never rose above freezing were strong drivers of wetland flooding (Figure 8), perhaps because they promote runoff from precipitation events and snowmelt. Winter precipitation and freezing temperatures could also be paramount for development and persistence of frost layers in wetland soils. More cold winter days could preserve winter precipitation as snow throughout the winter; thereby increasing runoff from spring snowmelt. Alternatively, greater mean winter vapor pressure deficits tended to decrease wetland occurrence (Figure 6). Increased vapor pressure deficit values are associated with greater temperatures and less precipitation. During the winter months, these factors may increase evaporation and inhibit the formation of frost layers in wetland soils.

Models for predicting wetland occurrence that excluded winter weather parameters yielded similar results to those that did not include them, but models predicting flooded wetland area were less similar. Two of the strongest drivers of wetland occurrence in both model types were total autumn precipitation and mean autumn maximum temperature. However, the best supported model for predicting wetland occurrence without winter weather parameters was the global model. This suggests that more parameters could have been incorporated to explain variation in wetland occurrence, and that excluded winter weather parameters were important.

Total winter precipitation and the number of winter days when the maximum temperature never rose above freezing were both influential for determining flooded wetland area. When winter weather variables were omitted, weather related parameter estimates tended to be lower, and the best supported predictive model did not fit the data as well as the best supported model that did incorporate winter weather parameters. Model results illustrated the importance of winter weather parameters for explaining spring wetland occurrence and flooded area. Instead of using models that exclude winter weather parameters, mean or expected winter weather variables could be input into models prior to the onset of winter to predict wetland flooded area. Scenario planning could also be utilized as a tool for considering the impacts of a wide range of potential winter weather patterns on flooded area.

Several weather variables were included in model confidence sets, but failed to have a strong influence on springtime wetland occurrence or flooded area. Major spring precipitation events had only slightly positive or negative influences on wetland occurrence and flooded area. This could be due to the long amounts of time that pass between the occurrence of major precipitation events and wetland inundation the following spring. The first autumn/winter date when the minimum temperature was < 0 degrees Celsius and the first winter/spring date when the minimum temperature was > 0 degrees Celsius were relatively weak drivers of wetland occurrence (Figure 6) and flooded area (Figure 8). Although these variables could be used to signify the onset of colder or warmer temperatures, mean maximum/minimum autumn and winter temperatures, and the number of winter days when the temperature was < 0 degrees Celsius were better predictors of wetland occurrence and flooded area (Figure 8).

The predictive models developed in this chapter provide additional insights into the role a variety of factors play in determining springtime wetland inundation. In general, more autumn and winter precipitation increases wetland occurrence and flooded area, whereas greater temperatures during these seasons decrease wetland occurrence and flooded area. In future studies, the roles that interactions between predictor variables play in determining wetland inundation could be explored. Interactions between precipitation and temperature could be important for determining the presence of a frost layer in wetland soils.

Future studies may assess how wetland occurrence and flooded area could impact food availability for migratory waterfowl and shorebirds. Different wetland types support different vegetative communities, some of which provide more food calories to birds than others (Bishop & Vrtiska 2008). Similarly, invertebrate densities could differ according to wetland type and surrounding landuse. Using model predictions to estimate food availability for migratory avifauna could help inform management decisions aimed at providing adequate habitat and food resources to migratory avifauna.

Wetland occurrence throughout the year could also be important for determining the functional connectivity of isolated wetlands, especially in agricultural matrices that may not be easily traversed by terrestrial organisms. Wetland occurrence may be more important for promoting functional connectivity than flooded area, especially if organisms use small, ephemeral wetlands as stepping stones when moving between larger wetlands. Because herpetofauna and other wetland dependent organisms are likely to be most active in late spring or early summer, determining the drivers of wetland occurrence during these time periods could be important for assessing the degree of functional connectivity.

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TABLES AND FIGURES

Table 1: Results of information theoretic model selection identifying the optimum random effects structure for predicting annual springtime variation in Rainwater Basin, Nebraska, wetland occurrence. A set of 26 competing models compared different random effects structures while holding the global fixed effects structure constant. Random effects structures allow the model intercept and/or fixed effects parameters to vary between individual wetlands and/or years. Winter weather parameters were included in the model set. The best supported model was determined to have the optimum random effects structure, and its AICc weight is listed in bold.

Model	**K**[a]	**AICc**[b]	**ΔAICc**[c]	***wi***[d]
GlobFix[e] + (Int[f] | Wet[g]) + (Int | Year[h]) + (MeanWintVpd[i] | Year)	18	6,088.83	0.00	**1.00**
GlobFix + (Int | Wet) + (Int | Year) + (TotSumPrecip[j] | Year)	18	6,100.17	11.34	0.00
GlobFix + (Int | Wet) + (Int | Year) + (MeanFallTmax[k] | Year)	18	6,135.62	46.79	0.00
GlobFix + (Int | Wet) + (Int | Year) + (YrFreezTmin[l] | Year)	18	6,161.11	72.28	0.00
GlobFix + (Int | Wet) + (YrFreezTmin | Wet) + (Int | Year)	18	6,165.25	76.42	0.00
GlobFix + (Int | Wet) + (TotSumPrecip | Wet) + (Int | Year)	18	6,165.45	76.62	0.00
GlobFix + (Int | Wet) + (Int | Wetland) + (Int | Year)	16	6,165.47	76.64	0.00
GlobFix + (Int | Wet) + (Int | Year) + (MajSpringPrecip[m] | Year)	18	6,166.57	77.73	0.00
GlobFix + (Int | Wet) + (MeanWintVpd | Wet) + (Int | Year)	18	6168.11	79.28	0.00
GlobFix + (Int | Wet) + (MeanFallTmax | Wet) + (Int | Year)	18	6,169.28	80.45	0.00
GlobFix + (Int | Wet) + (MajSpringPrecip | Wet) + (Int | Year)	18	6,169.28	80.45	0.00
GlobFix + (Int | Wet) + (TotFallPrecip[n] | Wet) + (Int | Year)	18	6,169.47	80.64	0.00
GlobFix + (Int | Wet) + (TotSumPrecip | Wet)	17	6,202.40	113.57	0.00

Table 1: Continued.

Model	**K**[a]	**AICc**[b]	**ΔAICc**[c]	***wi***[d]
GlobFix + (Int | Wet)	15	6,211.53	122.70	0.00
GlobFix + (Int | Wet) + (YrFreezTmin | Wet)	17	6,212.36	123.53	0.00
GlobFix + (Int | Wet) + (MajSpringPrecip | Wet)	17	6,214.34	125.51	0.00
GlobFix + (Int | Wet) + (MeanWintVpd | Wet)	17	6,215.29	126.46	0.00
GlobFix + (Int | Wet) + (MeanFallTmax | Wet)	17	6,215.54	126.71	0.00
GlobFix + (Int | Wet) + (TotFallPrecip | Wet)	17	6,215.55	126.71	0.00
GlobFix + (Int | Year) + (TotFallPrecip | Year)	17	7,029.83	941.00	0.00
GlobFix + (Int | Year) + (MeanWintVpd |Year)	17	7,052.97	964.14	0.00
GlobFix + (Int | Year) + (TotSumPrecip | Year)	17	7,063.98	975.15	0.00
GlobFix + (Int | Year) + (MeanFallTmax | Year)	17	7,079.40	990.57	0.00
GlobFix + (Int | Year) + (YrFreezTmin | Year)	17	7,094.49	1,005.66	0.00
GlobFix + (Int | Year)	15	7,095.90	1,007.06	0.00
GlobFix + (Int | Year) + (MajSpringPrecip | Year)	17	7,122.23	1,033.40	0.00

[a–d] *K = Number of parameters in model; AICc = Akaike's Information Criterion adjusted for small sample size; ΔAICc = Relative AICc; wi = AICc weight.*

[e–n] *GlobFix = Global fixed effects structure, containing all uncorrelated variables; Int = Model intercept ; Wet = Individual wetlands; Year = Individual years; MeanWintVpd = Mean winter vapor pressure deficit; TotSumPrecip = Total summer precipitation; MeanFallTmax = Mean autumn maximum temperature; YrFreezTmin = First date of the year when minimum temperatures fell below zero degrees Celsius; MajSpringPrecip = Number of spring days with more than 50.8 mm in precipitation; TotFallPrecip = Total autumn precipitation.*

Table 2: Results of information theoretic model selection identifying the optimum fixed effects structure for predicting annual springtime variation in Rainwater Basin, Nebraska, wetland occurrence. A set of 29 competing models compared different fixed effects structures while holding the optimum effects structure constant. Winter weather parameters were included in the model set. The confidence set consisted of the best supported model and all models with an AICc weight at least 10% that of the best supported model. AICc weights of models included in the confidence set are listed in bold.

Model	**K**[a]	**AICc**[b]	**ΔAICc**[c]	***wi***[d]
Int[e] + NonWeath[f] + TotSumPrecip[g] + TotFallPrecip[h] + MeanFallTmax[i] + MeanWintVpd[j] + TopRand[k]	16	6,084.82	0.00	**0.43**
Int + NonWeath + TotSumPrecip + TotFallPrecip + MeanFallTmax + MeanWintVpd + YrFreezTmin[l] + TopRand	17	6,086.82	2.00	**0.16**
Int + NonWeath + MajSpringPrecip[m] + TotSumPrecip + TotFallPrecip + MeanFallTmax + MeanWintVpd + TopRand	17	6,086.83	2.01	**0.16**
Int + NonWeath + TotSumPrecip + TotFallPrecip + MeanWintVpd + TopRand	15	6,087.54	2.72	**0.11**
GlobFix + TopRand	18	6,088.83	4.01	**0.06**
Int + NonWeath + TotSumPrecip + TotFallPrecip + MeanWintVpd + YrFreezTmin + TopRand	16	6,089.52	4.70	0.04
Int + NonWeath + TotSumPrecip + TotFallPrecip + MeanWintVpd + YrFreezTmin + TopRand	16	6,089.55	4.73	0.04
Int + NonWeath + MajSpringPrecip + TotSumPrecip + TotFallPrecip + MeanWintVpd + TopRand	17	6,091.53	6.71	0.01
Int + NonWeath + MajSpringPrecip + TotSumPrecip + TotFallPrecip + MeanWintVpd + YrFreezTmin + TopRand	16	6,100.12	15.30	0.00
Int + NonWeath + TotSumPrecip + MeanFallTmax + MeanWintVpd + YrFreezTmin + TopRand	14	6,101.37	16.55	0.00

Table 2: Continued.

Model	**K**[a]	**AICc**[b]	**ΔAICc**[c]	***wi***[d]
Int + NonWeath + TotFallPrecip + MeanWintVpd + TopRand	15	6,101.61	16.79	0.00
Int + NonWeath + MajSpringPrecip + TotFallPrecip + MeanWintVpd + TopRand	17	6,102.11	17.29	0.00
Int + NonWeath + MajSpringPrecip + TotSumPrecip + MeanFallTmax + MeanWintVpd + TopRand	15	6,102.46	17.64	0.00
Int + NonWeath + MajSpringPrecip + TotFallPrecip + MeanWintVpd + YrFreezTmin + TopRand	15	6,102.56	17.74	0.00
Int + NonWeath + MajSpringPrecip + TotSumPrecip + MeanFallTmax + MeanWintVpd + YrFreezTmin + TopRand	16	6,102.61	17.79	0.00
Int + NonWeath + TotFallPrecip + MeanWintVpd + YrFreezTmin + TopRand	16	6,102.99	18.17	0.00
Int + NonWeath + TotFallPrecip + MeanFallTmax + MeanWintVpd + TopRand	16	6,103.70	18.88	0.00
Int + NonWeath + MajSpringPrecip + TotFallPrecip + MeanFallTmax + MeanWintVpd + TopRand	15	6,103.95	19.13	0.00
Int + NonWeath + TotSumPrecip + MeanFallTmax + MeanWintVpd + TopRand	17	6,104.03	19.21	0.00
Int + NonWeath + MajSpringPrecip + TotFallPrecip + MeanFallTmax + MeanWintVpd + YrFreezTmin + TopRand	16	6,105.49	20.67	0.00
Int + NonWeath + TotSumPrecip + MeanWintVpd + TopRand	14	6,118.82	34.00	0.00
Int + NonWeath + MajSpringPrecip + TotSumPrecip + MeanWintVpd + YrFreezTmin + TopRand	16	6,121.77	36.96	0.00
Int + NonWeath + MeanFallTmax + MeanWintVpd + TopRand	14	6,127.00	42.19	0.00
Int + NonWeath + MajSpringPrecip + MeanFallTmax + MeanWintVpd + TopRand	15	6,128.44	43.62	0.00
Int + NonWeath + MeanFallTmax + MeanWintVpd + YrFreezTmin + TopRand	15	6,128.90	44.09	0.00
Int + NonWeath[p] + MeanWintVpd + TopRand	13	6,131.64	46.82	0.00
Int + NonWeath + MajSpringPrecip + MeanWintVpd + TopRand	14	6,133.27	48.45	0.00

Int + NonWeath + MeanWintVpd + YrFreezTmin + TopRand	14	6,133.64	48.83	0.00
Int + TopRand[o]	5	6,748.05	663.23	0.00

[a–d] *K = Number of parameters in model; AICc = Akaike's Information Criterion adjusted for small sample size; ΔAICc = Relative AICc; wi = AICc weight.*

[e–m] *Int = model intercept; NonWeath = non–weather parameters (Semi–permanent wetland type, temporary wetland type, Euclidian distance to nearest irrigation reuse pit, hydric footprint perimeter to area ratio; center–pivot irrigation, dryland and gravity irrigation); TotSumPrecip = Total summer precipitation; TotFallPrecip = Total autumn precipitation; MeanFallTmax = Mean autumn maximum temperature; MeanWintVpd = mean winter vapor pressure deficit; TopRand = Random effects structure allowing the model intercept to vary between wetlands and the model intercept and mean winter vapor pressure deficit to vary between years; YrFreezTmin = First autumn/winter date with minimum temperature < 0 degrees Celsius; MajSpringPrecip = Number of spring days with > 50.8 mm of precipitation.*

Table 3: Estimates of all parameters in the confidence set for predicting springtime occurrence of Rainwater Basin wetlands. Five models supported with an Akaike's Information Criterion corrected for small sample size (AICc) weight of at least 10% that of the best supported model were included in the confidence set. Competing models included winter weather parameters.

			95% Confidence Interval	
Parameter	**Estimate**	**Standard Error**	**Lower**	**Upper**
Intercept	2.1431	0.8516	0.4740	3.8122
Semi–perm[a]	0.9471	0.3477	0.2657	1.6286
Temp[b]	-1.1867	0.1783	-1.5362	-0.8372
Center–pivot[c]	-0.7062	0.1767	-1.0525	-0.3599
Gravity[d]	-0.4110	0.2343	-0.8704	0.0483
Dryland[e]	-0.8045	0.2308	-1.2569	-0.3521
Pit distance[f]	-0.1806	0.0717	-0.3212	-0.0400
Shape complexity[g]	-1.4723	0.0856	-1.6401	-1.3045
MajSpringPrecip[h]	-0.0014	0.0486	-0.0966	0.0938
TotSumPrecip[i]	0.3685	0.0821	0.2075	0.5295
TotFallPrecip[j]	1.1269	0.1896	0.7554	1.4985
MeanFallTmax[k]	-0.9951	0.3519	-1.6848	-0.3054
MeanWintVpd[l]	-0.3194	0.4814	-1.2629	0.6242
YrFreezTmin[m]	-0.0104	0.1169	-0.2395	0.2186

[a–m] Semi–perm = Semi–permanent wetland type; Temp = Temporary wetland type; Center–pivot = Center–pivot irrigation; Gravity = Gravity irrigation; Dryland = Dryland agriculture; Pit distance = Euclidian distance to nearest irrigation reuse pit; Shape complexity = Hydric footprint perimeter to area ratio; MajSpringPrecip = Number of spring days with > 50.8 mm of precipitation; TotSumPrecip = Total summer precipitation; TotFallPrecip = Total autumn precipitation; MeanFallTmax = Mean autumn maximum temperature; MeanWintVpd = Mean winter vapor pressure deficit; YrFreezTmin = First autumn/winter date with minimum temperature < 0 degrees Celsius.

Table 4: Results of information theoretic model selection identifying the optimum random effects structure for predicting annual springtime variation in Rainwater Basin, Nebraska, wetland occurrence. A set of 19 competing models compared different random effects structures while holding the global fixed effects structure constant. Random effects structures allow the model intercept and/or fixed effects parameters to vary between individual wetlands and/or years. Winter weather parameters were not included in the model set. The best supported model was determined to have the optimum random effects structure, and its AICc weight is listed in bold.

Model	**K**[a]	**AICc**[b]	**ΔAICc**[c]	***wi***[d]
GlobFix[e] + (Int[f] | Wet[g]) + (Int | Year[h]) + (TotFallPrecip[i] | Year)	16	6,062.01	0.00	**1.00**
GlobFix + (Int | Wet) + (Int | Year) + (TotSumPrecip[j] | Year)	16	6,096.98	34.96	0.00
GlobFix + (Int | Wet) + (Int | Year) + (MeanFallTmax[k] | Year)	16	6,134.97	72.96	0.00
GlobFix + (Int | Wet) + (TotSumPrecip | Wet) + (Int | Year)	16	6,163.27	101.26	0.00
GlobFix + (Int | Wet) + (Int | Year)	14	6,163.53	101.51	0.00
GlobFix + (Int | Wet) + (Int | Year) + (MajSpringPrecip[l] | Year)	16	6,165.16	103.14	0.00
GlobFix + (Int | Wet) + (MeanFallTmax | Wet) + (Int | Year)	16	6,167.28	105.27	0.00
GlobFix + (Int | Wet) + (MajSpringPrecip | Wet) + (Int | Year)	16	6,167.33	105.32	0.00
GlobFix + (Int | Wet) + (TotFallPrecip | Wet) + (Int | Wet)	16	6,167.53	105.52	0.00
GlobFix + (Int | Wet) + (TotSumPrecip | Wet)	15	6,643.59	581.58	0.00
GlobFix + (Int | Wet) + (MeanFallTmax | Wet)	15	6,679.48	617.47	0.00
GlobFix + (Int | Wet) + (MajSpringPrecip | Wet)	15	6,683.46	621.45	0.00
GlobFix + (Int | Wet)	13	6,692.55	630.54	0.00
GlobFix + (Int | Wet) + (TotFallPrecip | Wet)	15	6,696.51	634.50	0.00
GlobFix + (Int | Year) + (TotFallPrecip | Year)	15	7,031.93	969.92	0.00
GlobFix + (Int | Year) + (TotSumPrecip | Year)	15	7,060.73	998.72	0.00
GlobFix + (Int | Year) + (MeanFallTmax | Year)	15	7,078.21	1,016.20	0.00

Table 4: Continued.

Model	**K**[a]	**AICc**[b]	**ΔAICc**[c]	***wi***[d]
GlobFix + (Int | Year)	13	7,092.63	1,030.62	0.00
GlobFix + (Int | Year) + (MajSpringPrecip | Year)	15	7,094.53	1,032.51	0.00

[a–d] *K = Number of parameters in model; AICc = Akaike's Information Criterion adjusted for small sample size; ΔAICc = Relative AICc; wi = AICc weight.*

[e–l] *GlobFix = Global fixed effects structure, containing all uncorrelated variables; Int = Model intercept ; Wet = Individual wetlands; Year = Individual years; TotFallPrecip = Total autumn precipitation; TotSumPrecip = Total summer precipitation; MeanFallTmax = Mean autumn maximum temperature; MajSpringPrecip = Number of spring days with more than 50.8 mm of precipitation.*

Table 5: Results of information theoretic model selection identifying the optimum fixed effects structure for predicting annual springtime variation in Rainwater Basin, Nebraska, wetland occurrence. A set of 9 competing models compared different fixed effects structures while holding the optimum effects structure constant. Winter weather parameters were not included in the model set. The confidence set consisted of the best supported model and all models with an AICc weight at least 10% that of the best supported model. AICc weights of models included in the confidence set are listed in bold.

Model	**K**[a]	**AICc**[b]	**ΔAICc**[c]	***wi***[d]
GlobFix[e] + TopRand[f]	16	6,062.01	0.00	**0.55**
Int[g] + NonWeath[h] + TotSumPrecip[i] + TotFallPrecip[j] + MeanFallTmax[k] + TopRand	15	6,062.44	0.42	**0.45**
Int + NonWeath + MajSpringPrecip[l] + TotSumPrecip + TotFallPrecip + TopRand	15	6,076.20	14.18	0.00
Int + NonWeath + TotSumPrecip + TotFallPrecip + TopRand	14	6,076.31	14.30	0.00
Int + NonWeath + MajSpringPrecip + TotFallPrecip + MeanFallTmax + TopRand	15	6,077.16	15.15	0.00
Int + NonWeath + TotFallPrecip + MeanFallTmax + TopRand	14	6,086.36	24.35	0.00
Int + NonWeath + MajSpringPrecip + TotFallPrecip + TopRand	14	6,090.66	28.64	0.00
Int + NonWeath + TotFallPrecip + TopRand	13	6,099.57	37.56	0.00
Int + TopRand	5	6,703.54	641.52	0.00

[a–d] *K = Number of parameters in model; AICc = Akaike's Information Criterion adjusted for small sample size; ΔAICc = Relative AICc; wi = AICc weight.*

[e–l] *Global fixed effects structure, containing all uncorrelated variables; TopRand = Random effects structure allowing the model intercept to vary between wetlands and the model intercept and total autumn precipitation to vary between years; Int = model intercept; NonWeath = non–weather parameters (Semi–permanent wetland type, temporary wetland type, Euclidian distance to nearest irrigation reuse pit, hydric footprint perimeter to area ratio; center–pivot irrigation, dryland and gravity irrigation); TotSumPrecip = Total summer precipitation; TotFallPrecip = Total autumn precipitation; MeanFallTmax = Mean autumn maximum temperature; MajSpringPrecip = Number of spring days with > 50.8 mm of precipitation.*

Table 6: Estimates of all parameters in the confidence set for predicting springtime occurrence of Rainwater Basin wetlands. Two models supported with an Akaike's Information Criterion corrected for small sample size (AICc) weight of at least 10% that of the best supported model were included in the confidence set. Competing models did not include winter weather parameters.

			95% Confidence Interval	
Parameter	**Estimate**	**Standard Error**	**Lower**	**Upper**
Intercept	2.8669	0.7476	1.4017	4.3321
Semi–perm[a]	0.9626	0.3479	0.2807	1.6444
Temp[b]	0.9840	0.1752	-1.3274	-0.6405
Center–pivot[c]	-0.6963	0.1767	-1.0427	-0.3499
Gravit[d]	-0.4237	0.2344	-0.8330	0.0357
Dryland[e]	-0.7664	0.2309	-1.2189	-0.3139
Pit distance[f]	-0.1508	0.0716	-0.2912	-0.0104
Shape complexity[g]	-1.5439	0.0862	-1.7128	-1.3750
MajSpringPrecip[h]	-0.0795	0.0484	-0.1744	0.0153
TotSumPrecip[i]	0.3868	0.0849	0.2204	0.5533
TotFallPrecip[j]	0.7566	0.7468	-0.7071	2.2203
MeanFallTmax[k]	-1.1955	0.2635	-1.7120	-0.6790

[a–k] *Semi–perm = Semi–permanent wetland type; Temp = Temporary wetland type; Center–pivot = Center–pivot irrigation; Gravity = Gravity irrigation; Dryland = Dryland agriculture; Pit distance = Euclidian distance to nearest irrigation reuse pit; Shape complexity = Hydric footprint perimeter to area ratio; MajSpringPrecip = Number of spring days with > 50.8 mm of precipitation; TotSumPrecip = Total summer precipitation; TotFallPrecip = Total autumn precipitation; MeanFallTmax = Mean autumn maximum temperature.*

Table 7: Results of information theoretic model selection identifying the optimum random effects structure for predicting annual springtime wetland flooded area in the Rainwater Basin, Nebraska. A set of 26 competing models compared different random effects structures while holding the global fixed effects structure constant. Random effects structures allow the model intercept and/or fixed effects parameters to vary between individual wetlands and/or years. Winter weather parameters were included in the model set. The best supported model was determined to have the optimum random effects structure, and its AICc weight is listed in bold.

Model	**K**[a]	**AICc**[b]	**ΔAICc**[c]	***wi***[d]
GlobFix[e] + (Int[f] | Wet[g]) + (Int | Year[h]) + (TotSumPrecip[i] | Year)	19	9,898.23	0.00	**1.00**
GlobFix + (Int | Wet) + (Int | Year) + (TotWintPrecip[j] | Year)	19	9,922.26	24.03	0.00
GlobFix + (Int | Wet) + (Int | Year) + (MeanFallVpd[k] | Year)	19	9,935.51	37.27	0.00
GlobFix + (Int | Wet) + (TotSumPrecip | Wet) + (Int | Year)	19	9,941.26	43.02	0.00
GlobFix + (Int | Wet) + (Int | Year) + (MajSpringPrecip[l] | Year)	19	9,952.80	54.56	0.00
GlobFix + (Int | Wet) + (TotSumPrecip | Wet)	18	9,955.34	57.11	0.00
GlobFix + (Int | Wet) + (TotWintPrecip | Wet) + (Int | Year)	19	9,956.55	58.32	0.00
GlobFix + (Int | Wet) + (YrThawTmin[m] | Wet) + (Int | Year)	19	9,957.72	59.49	0.00
GlobFix + (Int | Wet) + (Int | Year) + (WintFreez[n] | Year)	19	9,959.76	61.53	0.00
GlobFix + (Int | Wet) + (Int | Year)	17	9,962.86	64.63	0.00
GlobFix + (WintFreez | Wet) + (Int | Year)	19	9,965.96	67.73	0.00
GlobFix + (Int | Wet) + (MajSpringPrecip | Wet) + (Int | Year)	19	9,966.01	67.78	0.00
GlobFix + (Int | Wet) + (MeanFallVpd | Wet) + (Int | Year)	19	9,966.25	68.02	0.00
GlobFix + (Int | Wet) + (Int | Year) + (YrThawTmin | Year)	19	9,966.72	68.49	0.00

Table 7: Continued.

Model	**K**[a]	**AICc**[b]	**ΔAICc**[c]	***wi***[d]
GlobFix + (Int | Wet) + (TotWintPrecip | Wet)	16	9,969.24	71.00	0.00
GlobFix + (Int | Wet) + (YrThawTmin | Wet)	18	9,971.17	72.94	0.00
GlobFix + (Int | Wet)	18	9,975.06	76.83	0.00
GlobFix + (Int | Wet) + (WintFreez | Wet)	18	9,977.89	79.66	0.00
GlobFix + (Int | Wet) + (MeanFallVpd | Wet)	18	9,978.22	79.99	0.00
GlobFix + (Int | Wet) + (MajSpringPrecip | Wet)	18	9,978.66	80.43	0.00
GlobFix + (Int | Year) + (TotSumPrecip | Year)	18	10,809.79	911.56	0.00
GlobFix + (Int | Year) + (TotWintPrecip | Year)	18	10,824.50	926.27	0.00
GlobFix + (Int | Year) + (MeanFallVpd | Year)	18	10,837.35	939.12	0.00
GlobFix + (Int | Year) + (MajSpringPrecip | Year)	18	10,844.16	945.93	0.00
GlobFix + (Int | Year) + (WintFreez | Year)	18	10,848.96	950.72	0.00
GlobFix + (Int | Year) + (YrThawTmin | Year)	18	10,854.97	956.73	0.00

[a–d] *K = Number of parameters in model; AICc = Akaike's Information Criterion adjusted for small sample size; ΔAICc = Relative AICc; wi = AICc weight.*

[e–n] *GlobFix = Global fixed effects structure, containing all uncorrelated variables; Int = Model intercept ; Wet = Individual wetlands; Year = Individual years; TotSumPrecip = Total summer precipitation; TotWintPrecip = Total winter precipitation; MeanFallVpd = Mean autumn vapor pressure deficit; MajSpringPrecip = Number of spring days with more than 50.8 mm in precipitation; YrThawTmin = First date of year when minimum temperatures rose above zero degrees Celsius; WintFreez = Number of winter days when the maximum temperature never rose above zero degrees Celsius.*

Table 8: Results of information theoretic model selection identifying the optimum fixed effects structure for predicting annual springtime variation in wetland flooded area in the Rainwater Basin, Nebraska. A set of 25 competing models compared different fixed effects structures while holding the optimum effects structure constant. Winter weather parameters were included in the model set. The confidence set consisted of the best supported model and all models with an AICc weight at least 10% that of the best supported model. AICc weights of models included in the confidence set are listed in bold.

Model	**K**[a]	**AICc**[b]	**ΔAICc**[c]	***wi***[d]
Int[e] + NonWeath[f] + TotSumPrecip[g] + TotWintPrecip[h] + WintFreez[i] + YrThawTmin[j] + TopRand[k]	17	9,831.74	0.00	**0.43**
Int + NonWeath + MajSpringPrecip[l] + TotSumPrecip + TotWintPrecip + WintFreez + YrThawTmin + TopRand	18	9,832.84	1.09	**0.25**
Int + NonWeath + TotSumPrecip + MeanFallVpd[m] + TotWintPrecip + WintFreez + YrThawTmin + TopRand	18	9,833.38	1.63	**0.19**
GlobFix + TopRand	19	9,834.41	2.67	**0.11**
Int + NonWeath + TotSumPrecip + MeanFallVpd + TotWintPrecip + WintFreez + TopRand	17	9,840.33	8.59	0.01
Int + NonWeath + MajSpringPrecip + TotSumPrecip + TotWintPrecip + WintFreez + TopRand	17	9,840.68	8.93	0.00
Int + NonWeath + MajSpringPrecip + TotSumPrecip + MeanFallVpd + TotWintPrecip + WintFreez + TopRand	18	9,841.98	10.24	0.00
Int + NonWeath + TotSumPrecip + MeanFallVpd + WintFreez + YrThawTmin + TopRand	17	9,845.36	13.62	0.00
Int + NonWeath + MajSpringPrecip + TotSumPrecip + MeanFallVpd + WintFreez + YrThawTmin + TopRand	18	9,845.57	13.82	0.00
Int + NonWeath + TotSumPrecip + WintFreez + TopRand	15	9,847.10	15.36	0.00

Table 8: Continued.

Model	**K**[a]	**AICc**[b]	**ΔAICc**[c]	***wi***[d]
Int + NonWeath + MajSpringPrecip + TotSumPrecip + WintFreez + TopRand	16	9,848.18	16.43	0.00
Int + NonWeath + TotSumPrecip + MeanFallVpd + WintFreez + TopRand	16	9,849.07	17.33	0.00
Int + NonWeath + MajSpringPrecip + TotSumPrecip + MeanFallVpd + WintFreez + TopRand	17	9,850.11	18.36	0.00
Int + NonWeath + TotSumPrecip + MeanFallVpd + TotWintPrecip + YrThawTmin + TopRand	17	9,872.33	40.59	0.00
Int + NonWeath + MajSpringPrecip + TotSumPrecip + MeanFallVpd + TotWintPrecip + YrThawTmin + TopRand	18	9,874.34	42.60	0.00
Int + NonWeath + TotSumPrecip + MeanFallVpd + YrThawTmin + TopRand	16	9,894.00	62.26	0.00
Int + NonWeath + TotSumPrecip + YrThawTmin + TopRand	15	9,896.10	64.36	0.00
Int + NonWeath + TotSumPrecip + TotWintPrecip + TopRand	15	9,911.37	79.62	0.00
Int + NonWeath + TotSumPrecip + MeanFallVpd + TotWintPrecip + TopRand	16	9,911.89	80.15	0.00
Int + NonWeath + MajSpringPrecip + TotSumPrecip + TotWintPrecip + TopRand	16	9,911.95	80.21	0.00
Int + NonWeath + MajSpringPrecip + TotSumPrecip + MeanFallVpd + TotWintPrecip + TopRand	17	9,912.44	80.69	0.00
Int + NonWeath + TotSumPrecip + MeanFallVpd + TopRand	15	9,927.27	95.52	0.00
Int + NonWeath + TotSumPrecip + TopRand	14	9,930.29	98.55	0.00
Int + NonWeath + MajSpringPrecip + TotSumPrecip + TopRand	15	9,931.75	100.01	0.00
Int + TopRand	6	10,361.61	529.87	0.00

[a–d] *K = Number of parameters in model; AICc = Akaike's Information Criterion adjusted for small sample size; ΔAICc = Relative AICc; wi = AICc weight.*
[e–m] *Int = model intercept; NonWeath = non–weather parameters (Semi–permanent wetland type, temporary wetland type, Euclidian distance to nearest irrigation reuse pit, hydric footprint perimeter to area ratio; center–pivot irrigation, dryland and gravity irrigation); TotSumPrecip = Total summer precipitation; TotWintPrecip = Total winter precipitation; WintFreez = Number of winter days with maximum temperature < 0 degrees Celsius; YrThawTmin = First winter/spring date with minimum temperature > 0 degrees Celsius; TopRand = Random effects structure allowing the model intercept to*

vary between wetlands and the model intercept and total summer precipitation to vary between years; MajSpringPrecip = Number of spring days with > 50.8 mm of precipitation; MeanFallVpd = Mean autumn vapor pressure deficit.

Table 9: Estimates of all parameters in the confidence set for predicting springtime flooded area in Rainwater Basin wetlands. Four models supported with an Akaike's Information Criterion corrected for small sample size (AICc) weight of at least 10% that of the best supported model were included in the confidence set. Competing models included winter weather parameters.

Parameter	Estimate	Standard Error	95% Confidence Interval Lower	Upper
Intercept	0.0713	0.1681	-0.2581	0.4008
Semi–perm[a]	0.6971	0.0735	0.5530	0.8411
Temp[b]	-0.3560	0.0490	-0.4520	-0.2600
Center–pivot[c]	-0.1734	0.0506	-0.2725	-0.0743
Gravity[d]	-0.3765	0.0673	-0.5084	-0.2446
Dryland[e]	0.1048	0.0674	-0.0274	0.2370
Pit distance[f]	0.0682	0.0202	0.0286	0.1079
Shape complexity[g]	-0.1425	0.0226	-0.1868	-0.0983
MajSpringPrecip[h]	0.0124	0.0133	-0.0137	0.0385
TotSumPrecip[i]	0.0752	0.0930	-0.1072	0.2575
MeanFallVpd[j]	0.0372	0.0490	-0.0589	0.1332
TotWintPrecip[k]	0.1132	0.0308	0.0528	0.1735
WintFreez[l]	0.2317	0.0355	0.1620	0.3013
YrThawTmin[m]	0.0550	0.0181	0.0195	0.0905

[a–m] Semi–perm = Semi–permanent wetland type; Temp = Temporary wetland type; Center–pivot = Center–pivot irrigation; Gravity = Gravity irrigation; Dryland = Dryland agriculture; Pit distance = Euclidian distance to nearest irrigation reuse pit; Shape complexity = Hydric footprint perimeter to area ratio; MajSpringPrecip = Number of spring days with > 50.8 mm of precipitation; MeanFallVpd = Mean autumn vapor pressure deficit; TotWintPrecip = Total winter precipitation; WintFreez = Number of winter days with maximum temperature < 0 degrees Celsius; YrThawTmin = First winter/spring date with minimum temperature > 0 degrees Celsius.

Table 10: Results of information theoretic model selection identifying the optimum random effects structure for predicting annual springtime wetland flooded area in the Rainwater Basin, Nebraska. A set of 23 competing models compared different random effects structures while holding the global fixed effects structure constant. Random effects structures allow the model intercept and/or fixed effects parameters to vary between individual wetlands and/or years. Winter weather parameters were not included in the model set. The best supported model was determined to have the optimum random effects structure, and its AICc weight is listed in bold.

Model	**K**[a]	**AICc**[b]	**ΔAICc**[c]	***wi***[d]
GlobFix[e] + (Int[f] | Wet[g]) + (Int | Year[h]) + (TotSumPrecip[i] | Year)	18	9,979.16	0.00	**1.00**
GlobFix + (Int | Wet) + (Int | Year) + (MeanFallVpd[j] | Year)	18	10,013.42	34.26	0.00
GlobFix + (Int | Wet) + (Int | Year) + (MajSpringPrecip[k] | Year)	18	10,128.04	148.88	0.00
GlobFix + (Int | Wet) + (TotSumPrecip | Wet) + (Int | Year)	18	10,139.90	160.74	0.00
GlobFix + (Int | Year) + (Int | Wet) + (MeanFallTmin[l] | Year)	18	10,148.52	169.36	0.00
GlobFix + (Int | Wet) + (MajFallPrecip[m] | Wet) + (Int | Year)	18	10,149.73	170.57	0.00
GlobFix + (Int | Wet) + (Int | Year) + (MajFallPrecip | Year)	18	10,150.32	171.17	0.00
GlobFix + (Int | Wet) + (Int | Year)	16	10,154.78	175.62	0.00
GlobFix + (Int | Wet) + (MeanFallTmin | Wet) + (Int | Year)	18	10,155.44	176.28	0.00
GlobFix + (Int | Wet) + (MajSpringPrecip | Wet) + (Int | Year)	18	10,158.21	179.05	0.00
GlobFix + (Int | Wet) + (MeanFallVpd | Wet) + (Int | Year)	18	10,158.66	179.50	0.00
GlobFix + (Int | Wet) + (TotSumPrecip | Wet)	17	10,444.82	465.66	0.00
GlobFix + (Int | Wet) + (MajFallPrecip | Wet)	17	10,465.23	486.07	0.00
GlobFix + (Int | Wet) + (MeanFallTmin | Wet)	17	10,466.89	487.73	0.00
GlobFix + (Int | Wet)	15	10,468.01	488.85	0.00

Table 10: Continued.

Model	**K**[a]	**AICc**[b]	**ΔAICc**[c]	***wi***[d]
GlobFix + (Int | Wet) + (MajSpringPrecip | Wet)	17	10,468.98	489.82	0.00
GlobFix + (Int | Wet) + (MeanFallVpd | Wet)	17	10,471.37	492.21	0.00
GlobFix + (Int | Year) + (TotSumPrecip | Year)	17	10,875.01	895.85	0.00
GlobFix + (Int | Year) + (MeanFallVpd | Year)	17	10,893.97	914.81	0.00
GlobFix + (Int | Year) + (MajSpringPrecip | Year)	17	10,964.59	985.43	0.00
GlobFix + (Int | Year) + (MeanFallTmin | Year)	17	10,980.42	1,001.26	0.00
GlobFix + (Int | Year)	15	10,985.41	1,006.25	0.00
GlobFix + (Int | Year) + (MajFallPrecip | Year)	17	10,986.82	1,007.66	0.00

[a–d] *K = Number of parameters in model; AICc = Akaike's Information Criterion adjusted for small sample size; ΔAICc = Relative AICc; wi = AICc weight.*

[e–m] *GlobFix = Global fixed effects structure, containing all uncorrelated variables; Int = Model intercept ; Wet = Individual wetlands; Year = Individual years; TotSumPrecip = Total summer precipitation; MeanFallVpd = Mean autumn vapor pressure deficit; MajSpringPrecip = Number of spring days with more than 50.8 mm of precipitation; MeanFallTmin = Mean autumn minimum temperature; MajFallPrecip = Number of autumn days with more than 25.4 mm of precipitation.*

Table 11: Results of information theoretic model selection identifying the optimum fixed effects structure for predicting annual springtime variation in wetland flooded area in the Rainwater Basin, Nebraska. A set of 25 competing models compared different fixed effects structures while holding the optimum effects structure constant. Winter weather parameters were not included in the model set. The confidence set consisted of the best supported model and all models with an AICc weight at least 10% that of the best supported model. AICc weights of models included in the confidence set are listed in bold.

Model	**K**[a]	**AICc**[b]	**ΔAICc**[c]	***wi***[d]
Int[e] + NonWeath[f] + TotSumPrecip[g] + MajFallPrecip[h] + MeanFallVpd[i] + TopRand[j]	16	9,919.69	0.00	**0.46**
Int + NonWeath + MajSpringPrecip[k] + TotSumPrecip + MajFallPrecip + MeanFallVpd + TopRand	17	9,921.13	1.44	**0.23**
Int + NonWeath + TotSumPrecip + MajFallPrecip + MeanFallTmin[l] + MeanFallVpd + TopRand	17	9,921.60	1.91	**0.18**
GlobFix + TopRand	18	9,923.05	3.36	**0.09**
Int + NonWeath + TotSumPrecip + MeanFallVpd + TopRand	15	9,927.27	7.57	0.01
Int + NonWeath + TotSumPrecip + MajFallPrecip + TopRand	15	9,927.76	8.06	0.01
Int + NonWeath + MajSpringPrecip + TotSumPrecip + MeanFallVpd + TopRand	16	9,928.60	8.90	0.01
Int + NonWeath + TotSumPrecip + MeanFallTmin + MeanFallVpd + TopRand	16	9,929.25	9.55	0.00
Int + NonWeath + MajSpringPrecip + TotSumPrecip + MajFallPrecip + TopRand	16	9,929.31	9.61	0.00
Int + NonWeath + TotSumPrecip + MajFallPrecip + MeanFallTmin + TopRand	16	9,929.76	10.07	0.00
Int + NonWeath + TotSumPrecip + TopRand	14	9,930.29	10.60	0.00
Int + NonWeath + MajSpringPrecip + TotSumPrecip + MeanFallTmin + MeanFallVpd + TopRand	17	9,930.58	10.89	0.00

Table 11: Continued.

Model	**K**[a]	**AICc**[b]	**ΔAICc**[c]	***wi***[d]
Int + NonWeath + MajSpringPrecip + TotSumPrecip + MajFallPrecip + MeanFallTmin + TopRand	17	9,931.32	11.62	0.00
Int + NonWeath + MajSpringPrecip + TotSumPrecip + TopRand	15	9,931.75	12.06	0.00
Int + NonWeath + TotSumPrecip + MeanFallTmin + TopRand	15	9,932.30	12.61	0.00
Int + NonWeath + MajSpringPrecip + TotSumPrecip + MeanFallTmin + TopRand	16	9,933.76	14.07	0.00
Int + TopRand	6	10,361.61	441.92	0.00

[a–d] *K = Number of parameters in model; AICc = Akaike's Information Criterion adjusted for small sample size; ΔAICc = Relative AICc; wi = AICc weight.*

[e–l] *Int = model intercept; NonWeath = non–weather parameters (Semi–permanent wetland type, temporary wetland type, Euclidian distance to nearest irrigation reuse pit, hydric footprint perimeter to area ratio; center–pivot irrigation, dryland and gravity irrigation); TotSumPrecip = Total summer precipitation; MajFallPrecip = Number of autumn days with > 25.4 mm of precipitation; MeanFallVpd = Mean autumn vapor pressure deficit; TopRand = Random effects structure allowing the model intercept to vary between wetlands and the model intercept and total summer precipitation to vary between years.*

Table 12: Estimates of all parameters in the confidence set for predicting springtime flooded area in Rainwater Basin wetlands. Four models supported with an Akaike's Information Criterion corrected for small sample size (AICc) weight of at least 10% that of the best supported model were included in the confidence set. Competing models did not include winter weather parameters.

			95% Confidence Interval	
Parameter	**Estimate**	**Standard Error**	**Lower**	**Upper**
Intercept	0.0763	0.2312	-0.3768	0.5293
Semi–perm[a]	0.6961	0.0741	0.5509	0.8413
Temp[b]	-0.3722	0.0506	-0.4713	-0.2731
Center–pivot[c]	-0.1720	0.0513	-0.2726	-0.0714
Gravity[d]	-0.3618	0.0677	-0.4946	-0.2291
Dryland[e]	0.1001	0.0684	-0.0339	0.2341
Pit distance[f]	0.0703	0.0205	0.0301	0.1104
Shape complexity[g]	-0.1303	0.0229	-0.1751	-0.0854
MajSpringPrecip[h]	-0.0103	0.0130	-0.0358	0.0153
TotSumPrecip[i]	0.0706	0.1326	-0.1893	0.3305
MajFallPrecip[j]	-0.0869	0.0278	-0.1414	-0.0323
MeanFallTmin[k]	0.0158	0.0395	-0.0616	0.0931
MeanFallVpd[l]	-0.1574	0.0494	-0.2543	-0.0605

[a–m] *Semi–perm = Semi–permanent wetland type; Temp = Temporary wetland type; Center–pivot = Center–pivot irrigation; Gravity = Gravity irrigation; Dryland = Dryland agriculture; Pit distance = Euclidian distance to nearest irrigation reuse pit; Shape complexity = Hydric footprint perimeter to area ratio; MajSpringPrecip = Number of spring days with > 0 degrees Celsius; TotSumPrecip = Total summer precipitation; MajFallPrecip = Total autumn precipitation; MeanFallTmin = Mean autumn mean temperature; MeanFallVpd = Mean autumn vapor pressure deficit.*

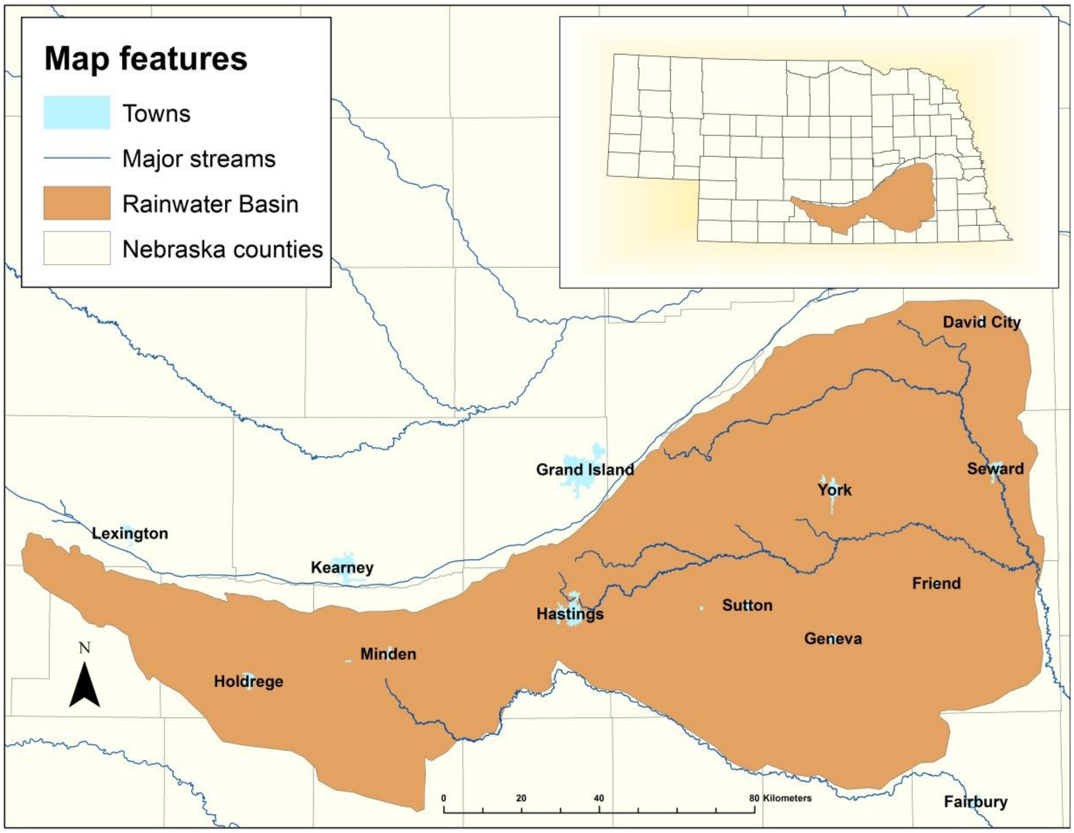

Figure 1: Location of the Rainwater Basin in south-central Nebraska, U.S.A, displaying Nebraska counties, major towns and rivers.

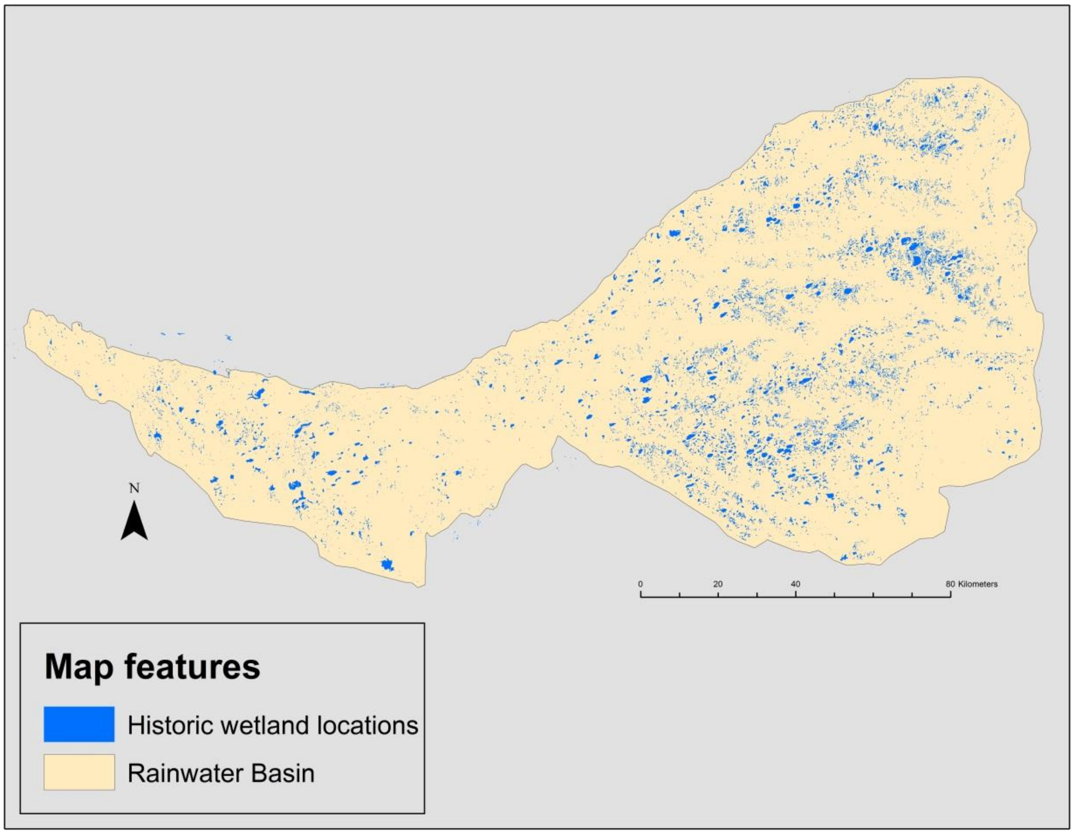

Figure 2: Historical Rainwater Basin wetlands, derived from soil survey maps, National Wetlands Inventory (NWI) surveys and Annual Habitat Surveys. As many as 1,000 major wetlands and 10,000 minor wetlands existed at the time of 19th Century European settlement.

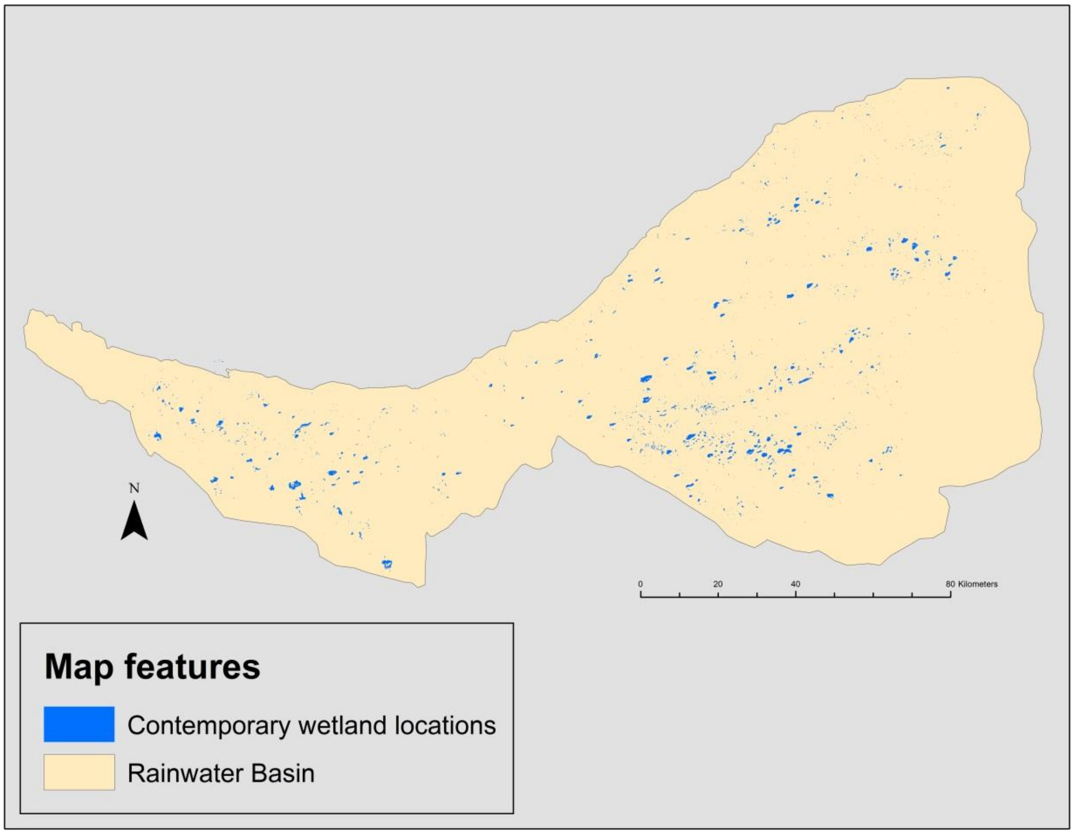

Figure 3: Contemporary Rainwater Basin wetlands determined by Rainwater Basin Joint Venture staff to be currently functioning, based on 2004 – 2007 Annual Habitat Survey data.

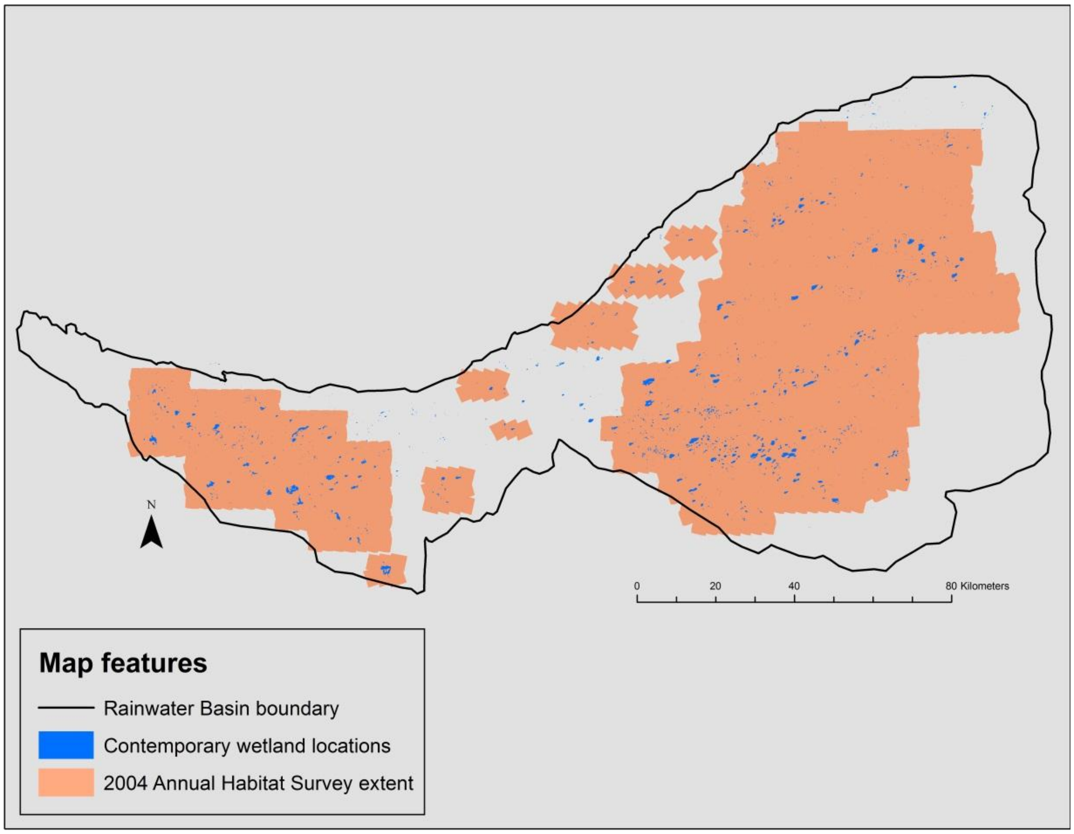

Figure 4: 2004 Annual Habitat Survey extent and locations of contemporary Rainwater Basin wetlands. Contemporary Rainwater Basin wetlands were determined by Rainwater Basin Joint Venture staff to be currently functioning, based on 2004 – 2007 Annual Habitat Survey data.

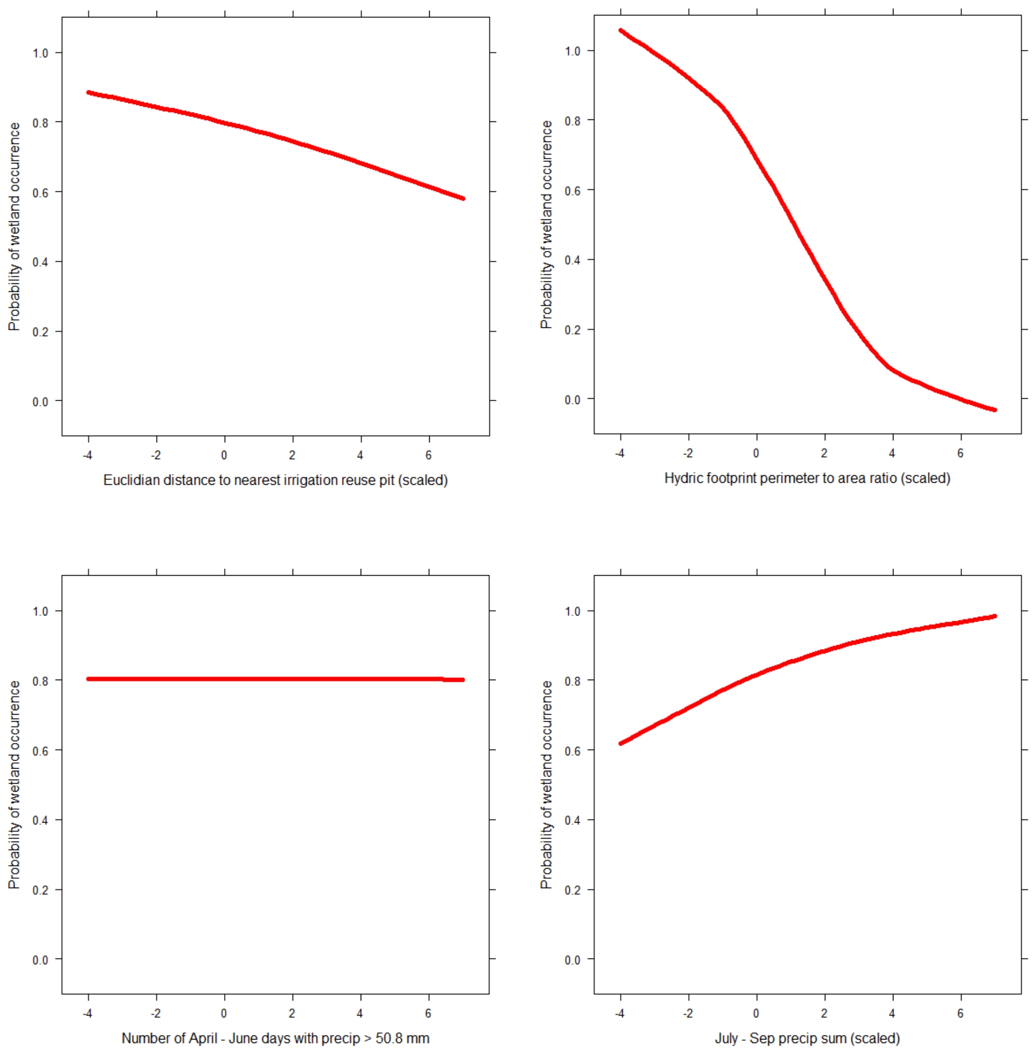

Figure 5: Predicted influences of Euclidian distance to the nearest irrigation reuse pit (upper left), hydric footprint perimeter to area ratio (upper right), number of spring days receiving more than 50.8 mm of precipitation (lower left), and of summer precipitation (lower right) on springtime Rainwater Basin wetland occurrence.

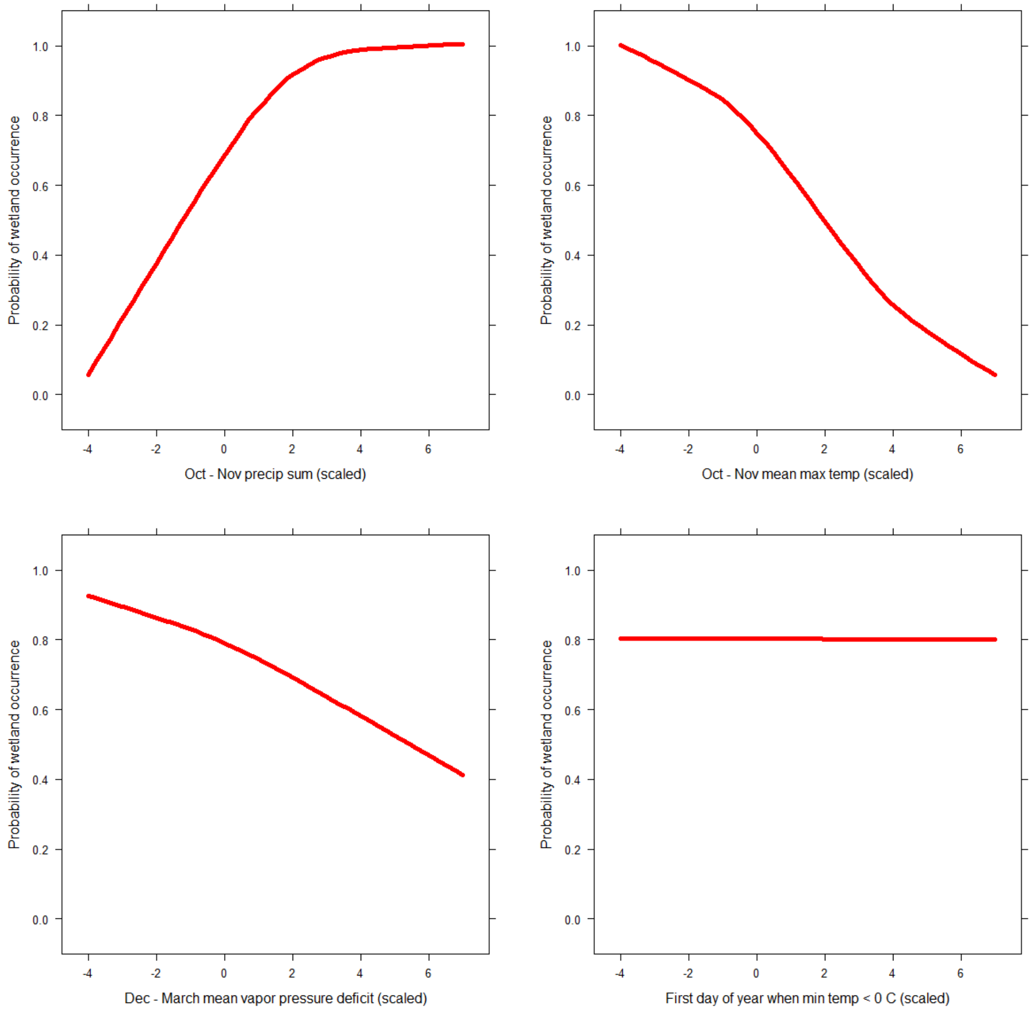

Figure 6: Predicted influences of total autumn precipitation (upper left), mean autumn maximum temperature (upper right), mean winter vapor pressure deficit (lower left), and the first autumn/winter date with minimum temperature < 0 degrees Celsius (lower right) on springtime Rainwater Basin wetland occurrence.

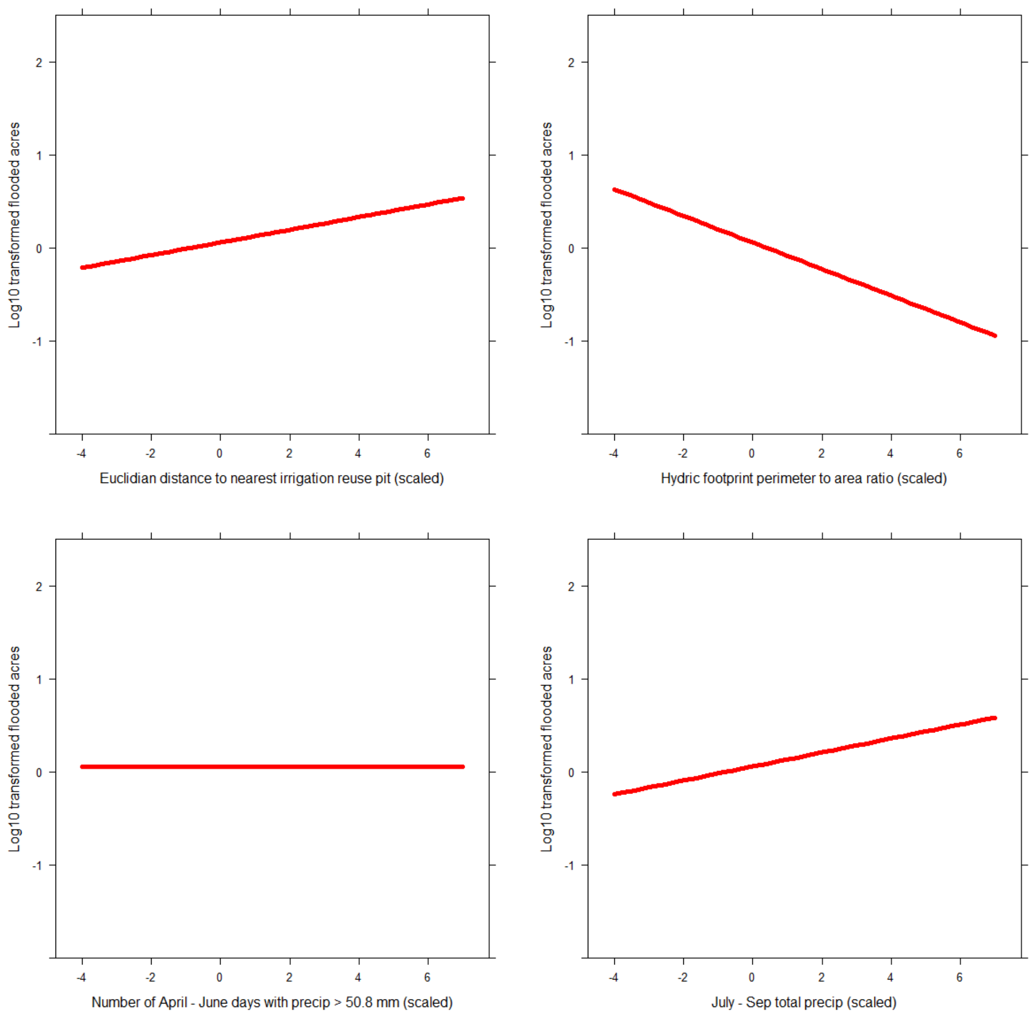

Figure 7: Predicted influences of Euclidian distance to nearest irrigation reuse pit (upper left), of hydric footprint perimeter to area ratio (upper right), the number of spring days with > 50.8 mm of precipitation (lower left), and total summer precipitation (lower right) on springtime Rainwater Basin flooded area.

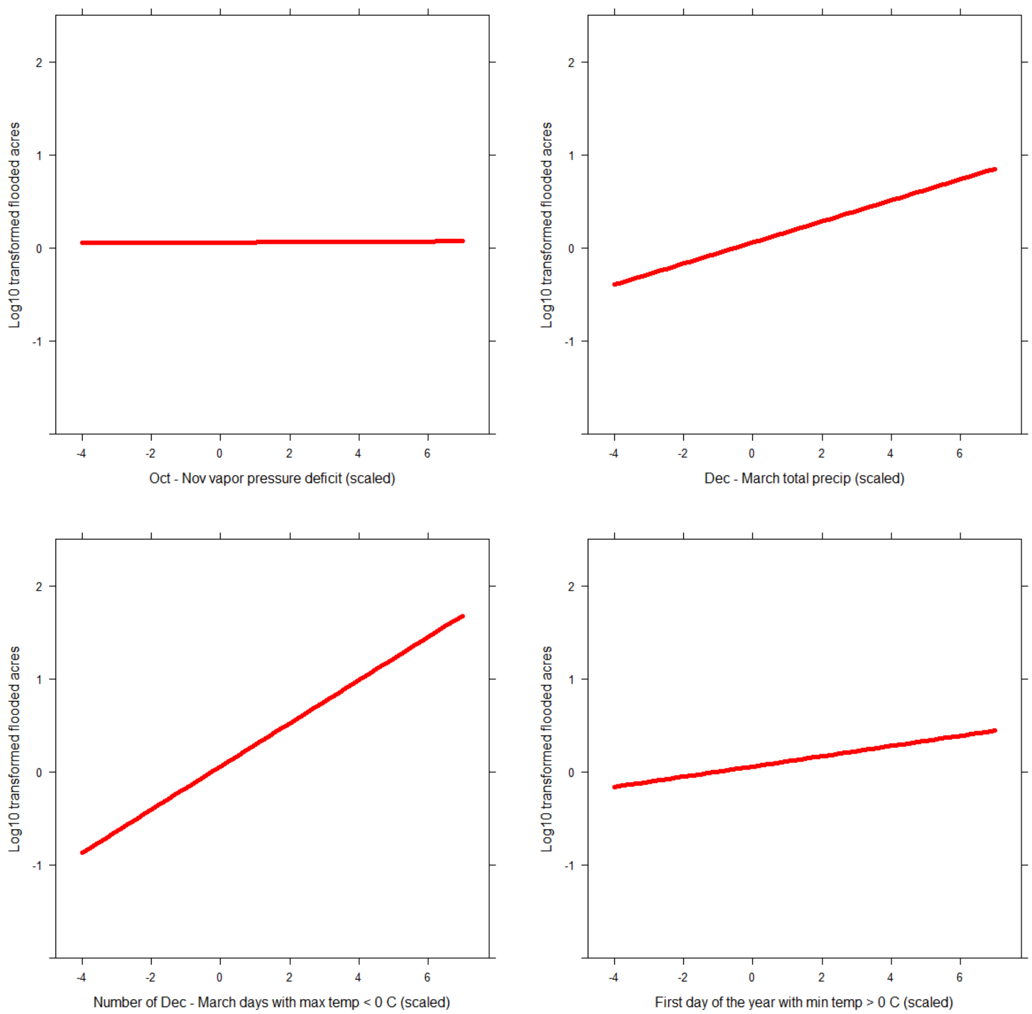

Figure 8 : Predicted influences of mean autumn vapor pressure deficit (upper left), of total winter precipitation (upper right), the number of winter days with maximum temperature < 0 degrees Celsius (lower left), and the first winter/spring date with minimum temperature > 0 degrees Celsius on springtime Rainwater Basin flooded area.

Chapter 7: SUMMARY AND SYNTHESIS

The replacement of Great Plains grasslands and wetlands by agriculture has increased food and bioenergy production, but has reduced grassland bird and migratory waterfowl and shorebird habitat (Samson & Knopf 1994; Higgins et al. 2002; Brennan & Kuvlesky 2005) and depleted groundwater resources in some areas (Rosenberg et al. 1999; Peterson et al. 2003b). Future agricultural landuse changes may be driven by bioenergy demands and regional climatic changes, and could affect bioenergy production, avifauna and agricultural water use. Given the uncertainties associated with future agricultural landuse and climate change, scenario planning is an appropriate tool for considering a variety of potential futures and informing future management actions (Peterson et al. 2003a; Williams et al. 2009).

This study incorporated uncertainties over future climatic conditions, ethanol demand, farmer decisions and agricultural policy adjustments into landuse change scenarios, in an effort to understand how future changes in agricultural landuse might impact ethanol production, grassland bird populations and groundwater irrigation withdrawals in the Rainwater Basin region. This study also used multi–model inference to develop predictive models explaining the influence of wetland characteristics, surrounding landuse and local weather events on springtime Rainwater Basin wetland occurrence and flooded area. Study results are useful for envisioning how future landuse and climatic changes may reshape the Rainwater Basin and surrounding agricultural landscapes in the future.

In Chapter 2, I addressed the feasibility of supplying adequate biomass for year–round cellulosic ethanol production from residual maize (*Zea mays*) stover and switchgrass (*Panicum virgatum*) within the 40 km road network service area of the Abengoa Bioenergy ethanol plant near York, Nebraska. I identified marginally productive rowcrop fields within the service area suitable for conversion from annual rowcrops to switchgrass and remaining areas of maize enrolled rowcrop fields from which maize stover could be collected. Together, potential annual switchgrass and maize stover supplies account for 77% – 135% of the biomass necessary to produce the same volume of ethanol currently produced from maize grain at the Abengoa Bioenergy plant. Results suggest that the eastern Rainwater Basin agricultural landscape is capable of generating the quantities of biomass necessary for year–round cellulosic production from switchgrass and maize stover. This conclusion could increase the relevance of studies assessing the economic, environmental or ecological impacts of cellulosic ethanol production from switchgrass and maize stover.

In Chapters 3 and 4, I explored the potential impacts of conversions between rowcrops, switchgrass and Conservation Reserve Program (CRP) grassland on Rainwater Basin grassland bird populations. Chapter 3 incorporated climatic change into landuse change scenarios and dealt solely with the conversion of marginally productive rowcrop fields to switchgrass stands, whereas Chapter 4 focused on the conversion of CRP grassland to switchgrass and rowcrops, and the conversion of rowcrops to CRP grassland. In general, the replacement of rowcrops with switchgrass benefitted grassland birds. The greatest increases in grassland bird abundance were observed under landuse change scenarios that assumed extreme climatic changes, irrigation limitations and the conversion of a great number of rowcrop hectares to switchgrass. Abundances of most grassland bird species also increased following the conversion of rowcrops to CRP grassland. Alternatively, converting CRP grassland to rowcrops or switchgrass stands negatively influenced most grassland bird species, with the conversion to rowcrops being more detrimental. These results highlight the importance of CRP grassland restorations to grassland bird populations in rowcrop dominated landscapes, and suggest that the impacts of switchgrass on grassland bird populations is likely to depend on which forms of landuse switchgrass replaces.

In Chapter 5, landuse change scenarios driven by potential climatic changes and irrigation limitations were used to consider how the conversion of marginally productive irrigated rowcrop fields in Natural Resource Districts (NRDs) with histories of implementing irrigation limitations to switchgrass could impact total annual groundwater irrigation withdrawals. Converting marginally productive irrigated rowcrop fields to switchgrass could reduce annual groundwater withdrawals by 2.6% – 5.6% in Rainwater Basin areas currently serviced by starch–based ethanol plants, or by 9.6% – 19.1% in areas with NRDs that have implemented irrigation limitations in the past. If future cellulosic ethanol production is initiated in NRDs with fully or over–appropriated water resources, converting some irrigated rowcrop fields to switchgrass could contribute to water conservation goals.

In Chapter 6, I used multi–model inference to develop predictive models that used wetland characteristics, surrounding landuse and weather events to explain annual variation in springtime Rainwater Basin wetland occurrence and flooded area. Several weather related and non–weather related model parameters were strong predictors of both wetland occurrence and flooded area. Increased total autumn precipitation, winter precipitation and number of winter days with maximum temperatures < 0 degrees Celsius increased wetland occurrence and flooded area. Increased hydric wetland footprint shape complexity, mean autumn temperature and mean winter vapor pressure deficit decreased wetland occurrence and flooded area. The models developed in this analysis could assist managers in predicting the availability of spring wetland stopover habitat and taking appropriate management actions to supply it through groundwater pumping in years when it is predicted to be limited.

Food production, bioenergy production and wildlife habitat conservation are likely to continue competing for landuse in Great Plains agricultural landscapes. Economics is typically a driving factor in land management decisions, and most landowners are expected to enroll in landuses that secure the greatest profit. However, if climate change spurs the implementation of additional irrigation limitations, it could promote the enrollment of marginally productive croplands in alternative forms of landuse like bioenergy switchgrass or CRP grassland. This diversification could benefit avian populations and conserve limited groundwater resources.

The continued promotion of grassland and wetland conservation programs, in coordination with the conversion of marginally productive rowcrop fields to switchgrass could be economically profitable for farmers, conserve groundwater resources and benefit avian populations. However, replacing conservation lands with rowcrops or switchgrass stands might offset the benefits associated with converting rowcrops to switchgrass. Agricultural landscapes interspersed with conservation lands, perennial bioenergy crops and rowcrops may represent a realistic compromise between agricultural producers and wildlife managers in times when high commodity prices encourage the enrollment of marginally productive lands in rowcrop production.

FUTURE RESEARCH

Cellulosic ethanol production feasibility

This study determined that adequate biomass supplies could be generated with the 40 km road network service area of the Abengoa Bioenergy ethanol plant to support year–round ethanol production at the plant. The considered study area is a highly cultivated region where rowcrop production dominates landuse. Future studies could assess the feasibility of supplying adequate biomass to ethanol plants in less intensively cultivated landscapes where greater proportions of the landscape is occupied by some form of grassland. In less cultivated landscapes, the potential for conversion to switchgrass may be greater than in landscapes with limited grassland, and cellulosic ethanol plants may be more reliant on switchgrass biomass than residual maize stover.

Bioenergy switchgrass and grassland birds

Future ecological bioenergy switchgrass research endeavors could focus on establishing and managing switchgrass stands as bioenergy crops in the Great Plains and conducting avian and insect surveys in them. Switchgrass grassland bird densities utilized in Chapters 3 and 4 were obtained from Murray and Best (2003), who conducted avian surveys in the Chariton Valley region of southern Iowa, U.S.A. Grassland bird density estimates in switchgrass stands established in Great Plains agricultural landscapes could be used to more realistically predict avian responses to the large scale production of bioenergy switchgrass in Nebraska.

Switchgrass and groundwater withdrawals

In Chapter 5, I concluded that the adoption of switchgrass as a bioenergy crop under changing climatic conditions could reduce groundwater withdrawals for rowcrop irrigation in water stressed regions. However, increased crop water use requirements were not taken into account in this study. Studies that incorporate the effects of increased evapotranspiration (ET) on maize and switchgrass water use and irrigation requirements could provide more reliable estimates of how switchgrass could affect regional agricultural groundwater use. In addition, restricting the study area to specific basins, subbasins or reaches within NRDs that have histories of implementing irrigation limitations could be used to directly infer how raising bioenergy switchgrass could be incorporated into existing and future integrated management plans developed for fully and overappropriated areas basins, subbasins and reaches.

Wetland occurrence and flooding

My assessment of the factors driving annual springtime Rainwater Basin occurrence and flooded area was largely exploratory in nature and focused on detecting general patterns influencing wetland inundation. Significant potential exists for the testing, refinement and fine tuning of the dataset and predictive models. Refinements could improve model performance and provide managers with more accurate predictions of springtime wetland stopover habitat availability.

During the development of predictive models, I did not break the data into training and testing sets, and therefore, was not able to validate models with data not involved in model development. As additional Annual Habitat Survey (AHS) data and weather data becomes available, models could be validated to better assess their predictive abilities. If additional weather data is not made available through the Yellowstone Ecological Research Center (YERC), data from Rainwater Basin weather stations could be used to produce weather data rasters through Geographic Information System (GIS) kriging. Creation of weather raster data layers through kriging would allow for the customization of weather data variables believed to be the most important drivers of springtime wetland inundation and could be useful to other analyses.

Restricting the contemporary wetlands dataset to even fewer wetlands known to be fully functional could improve model fit by removing some residual variation due to agricultural landscape alterations. This refinement would likely restrict model inference to highly functioning wetlands not located in agricultural fields and on public property that are managed with the intent of providing maximum benefit to wildlife populations. However, if management objectives include estimating the flooded wetland area present within rowcrop fields, the models presented in Chapter 6 may be most useful.

The predictive models presented in this study did not incorporate interactions between explanatory variables. While the main effects of multiple parameters were shown to be important drivers of wetland occurrence and flooded area, interactions between variables could have similar or stronger effects. This may especially be true of interactions between autumn or winter precipitation and winter freezing temperatures. If wetland soils are saturated by precipitation events, and then experience freezing temperatures throughout the winter, frost layers may develop in wetland soils and better retain water in wetlands during spring migration. The fact that autumn and winter precipitation and maximum temperatures were shown to be strong drivers of wetland occurrence and flooded area individually lends support to this hypothesis. Predictive models could also be customized to timeframes managers prefer for implementing management actions aimed at providing additional stopover wetland habitat. This would allow for predictions of wetland stopover habitat availability to be made far enough in advance for management actions aimed at increased habitat area to be taken.

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