Opportunities and Trade-offs among BECCS and the Food, Water, Energy, Biodiversity, and Social Systems Nexus at Regional Scales

PAUL C. STOY, SELENA AHMED, MEGHANN JARCHOW, BENJAMIN RASHFORD, DAVID SWANSON, SHANNON ALBEKE, GABRIEL BROMLEY, E. N. J. BROOKSHIRE, MARK D. DIXON, JULIA HAGGERTY, PERRY MILLER, BRENT PEYTON, ALISA ROYEM, LEE SPANGLER, CRISTA STRAUB, AND BENJAMIN POULTER

Carbon dioxide must be removed from the atmosphere to limit climate change to 2°C or less. The integrated assessment models used to develop climate policy acknowledge the need to implement net negative carbon emission strategies, including bioenergy with carbon capture and storage (BECCS), to meet global climate imperatives. The implications of BECCS for the food, water, energy, biodiversity, and social systems (FWEBS) nexus at regional scales, however, remain unclear. Here, we present an interdisciplinary research framework to examine the trade-offs as well as the opportunities among BECCS scenarios and FWEBS on regional scales using the Upper Missouri River Basin (UMRB) as a case study. We describe the physical, biological, and social attributes of the UMRB, and we use grassland bird populations as an example of how biodiversity is influenced by energy transitions, including BECCS. We then outline a "conservation" BECCS strategy that incorporates societal values and emphasizes biodiversity conservation.

Keywords: agroecosystems, biofuels, land-use management, natural resources, sustainability

Atmospheric concentrations of carbon dioxide (CO_2) and other greenhouse gases (GHGs) continue to increase as a result of land-use change, fossil energy production, and other anthropogenic activities (Le Quéré et al. 2013). To ameliorate the impact of GHGs on climate, international negotiations led by the United Nations Framework Convention on Climate Change (UNFCCC) target a 2°C maximum increase in global average temperature (Meinshausen et al. 2009), assumed to be a "safe" threshold for climate change. The Paris Agreement, signed on 22 April 2016 by 195 countries, takes this effort a step further by pursuing efforts to limit warming to 1.5°C (Hulme 2016, Rogelj et al. 2016). Such targets guide policy scenarios for fossil-fuel management via integrated assessment models (IAMs) to achieve climate stabilization (Moss et al. 2010).

Integrated assessment models emphasize interactions among global economic, energy, land-use, and technology systems (Jones et al. 2013, Collins et al. 2015) and play a major role in climate-change-mitigation policy, with large implications for Earth-system management (Schellnhuber 1999, Barros 2014, Stocker 2014). Since the *Fifth Assessment Report of the Intergovernmental Panel on Climate Change* (IPCC AR5; IPCC 2014), the development of global GHG reduction scenarios via IAMs has shifted to emphasize net *negative* CO_2 emission—that is, net carbon sequestration. This is because GHG emissions will now peak later than previously hoped and atmospheric GHG concentrations will decline less steeply than necessary to avoid climate warming of 2°C or less (Rockström et al. 2017).

Negative CO_2 emission pathways rely on emerging technologies, including *bioenergy with carbon capture and storage* (BECCS; Kriegler et al. 2013, van Vuuren et al. 2013), in which biomass is used to generate energy and CO_2 is removed from the atmosphere through geologic sequestration or by enhancing natural carbon (C) storage (Fuss et al. 2013, Smith et al. 2015). The proposed BECCS economy is important to modeling efforts in the latest IPCC AR5 (Tavoni et al. 2014) and continues to play a large role in the shared socioeconomic pathways (SSPs) of the forthcoming *Sixth IPCC Assessment Report* (Lotze-Campen et al. 2013, Riahi et al. 2017). To meet the goals of the Paris Agreement, global anthropogenic CO_2 emissions need to be reduced

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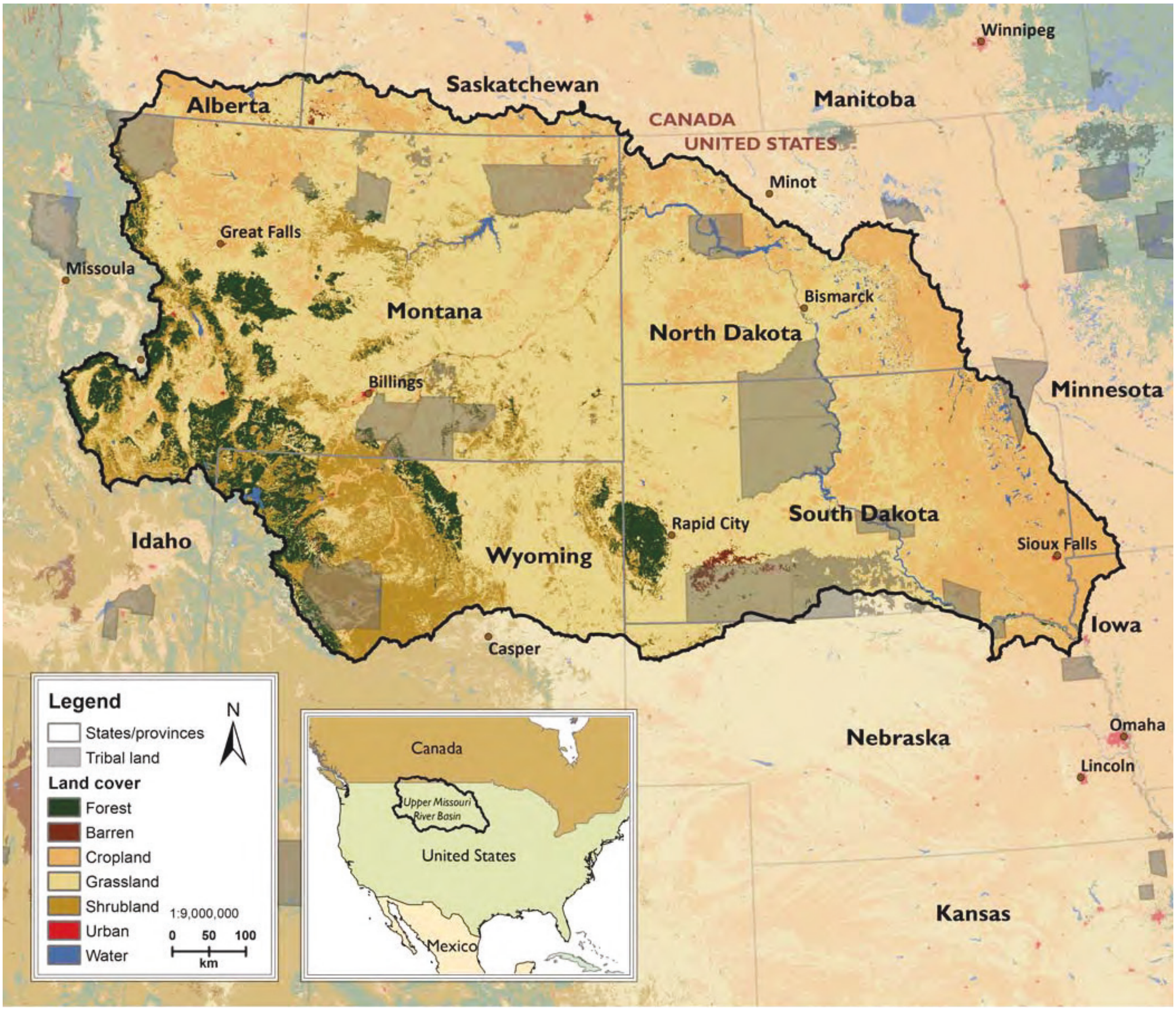

Figure 1. The Upper Missouri River Basin (UMRB) is defined as the region upriver from the confluence of the Big Sioux and Missouri Rivers in Sioux City, Iowa (excluding the Niobrara watershed), with major land-use classifications and administrative (state and reservation) boundaries.

by approximately half every decade, and atmospheric CO_2 removal needs to approach 5 metric gigatons per year with no net land-use emissions—including those due to land-use change—by 2050 (Rockström et al. 2017), underscoring the importance of adopting CO_2 removal techniques such as BECCS globally.

Although BECCS may make sense in global climate scenarios, the implications of BECCS for food security, clean energy, water resources, biodiversity, social systems, and other attributes of value to society at regional scales are less clear (Rhodes and Keith 2008, Bonsch et al. 2014, Tian et al. 2016). Despite the importance of BECCS in the UNFCCC process, environmental and socioeconomic trade-offs for large-scale deployment of BECCS are poorly considered in regional studies and are of growing concern, calling into question the overall validity of IAMs as they guide policy (Fuss et al. 2014, Smith et al. 2015, Zilberman 2015).

Here, we describe an interdisciplinary framework for analyzing the trade-offs and opportunities among emerging BECCS strategies and the regional food, water, energy, biodiversity, and social systems (FWEBS) that they affect across a diverse and changing region of North America, the Upper Missouri River Basin (UMRB; figure 1). We first describe the FWEBS research framework (figure 2) and characterize the UMRB as a case study for regional BECCS implementation; we then discuss how scenario development can help us understand its interaction with the FWEBS nexus (figure 3). The discussion is guided by our goal to understand whether negative CO_2 emissions can be reached in the UMRB, under what land-use configurations, and at what cost or benefit to

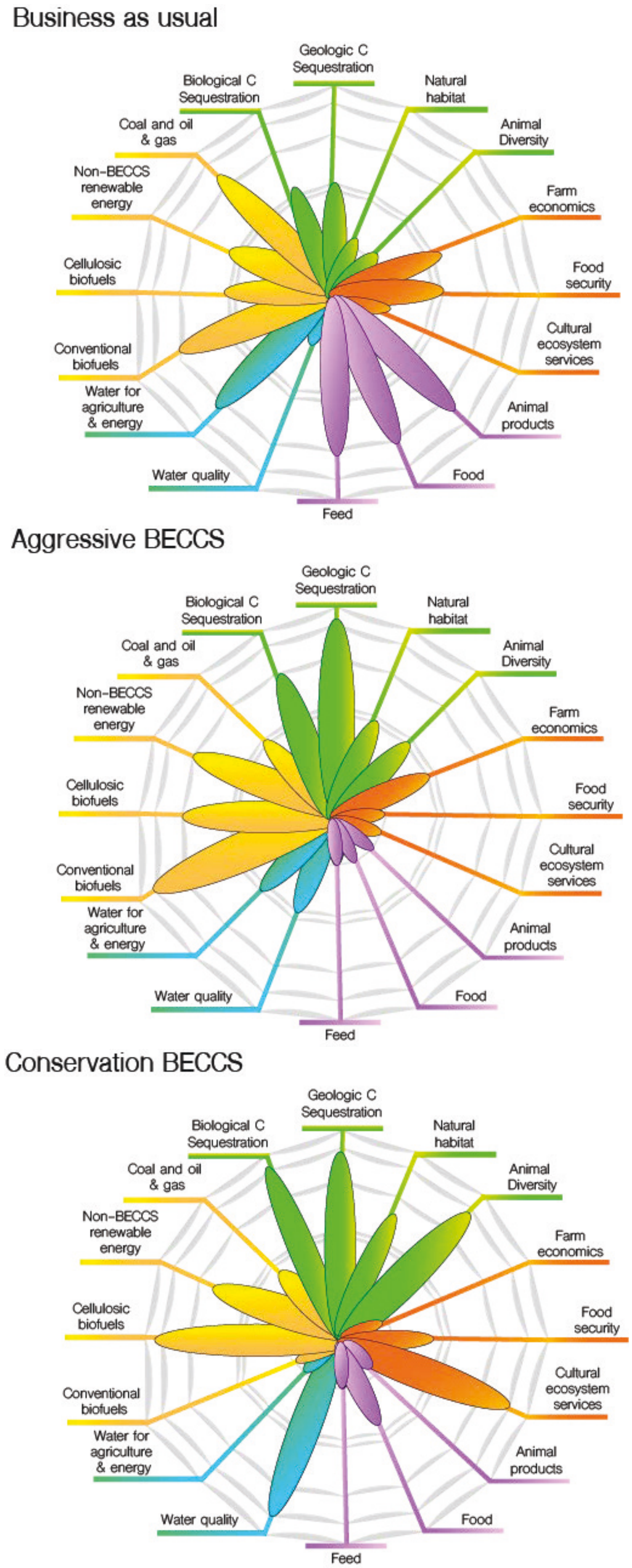

Figure 2. Conceptual diagrams following Foley and colleagues (2005) for business-as-usual scenarios, "aggressive" bioenergy with carbon capture and storage (BECCS) scenarios, and "conservation" BECCS scenarios that integrate sustainable management of the food, water, energy, biodiversity, and social systems (FWEBS) nexus.

local communities and ecosystem (as well as Earth-system) services.

The food, water, energy, biodiversity, and social-systems research framework

The implementation of a BECCS-based economy will affect multiple ecosystem and societal services, including water quality and supply (Popp et al. 2014, Albanito et al. 2015), human nutrition (Tilman and Clark 2014), technology (Baum 2014), regional economics (Muratori et al. 2016), biodiversity (Powell and Lenton 2013), and cultural ecosystem services (Galaz 2012, Scholes 2016). The processes influenced by regional BECCS strategies must be studied in concert; we need to take into account how to provide for society's growing demand for food, water, and energy while maintaining biodiversity, ecosystem services, and economic and social systems, including cultural values and identity, social networks, and livelihoods. The interconnectedness of these systems that support human well-being and lifestyles is increasingly evident and has led researchers to approach these systems as a nexus—the water–energy–food (WEF) nexus—for identifying cross-sector efficiencies (Scanlon et al. 2017) and to develop solutions to pressing resource challenges without unintended consequences (Scott et al. 2015). Each system within the WEF nexus can be viewed as a socioecological system comprising biophysical components and human components that are characterized by dynamic feedback loops. BECCS approaches that emphasize terrestrial C storage may prove technically feasible, but in the context of the WEF nexus, their implications for regional economies may make such approaches socially impractical. Scholars, practitioners, and policymakers have promoted the WEF nexus as a conceptual tool for approaching sustainability, including the United Nation's sustainable development goals (SDGs), and protecting against potential risks of future water, energy, and food insecurity (Biggs et al. 2015). However, research frameworks for nexus thinking often fail to incorporate biodiversity and other ecosystem services, as well as social dimensions such as livelihoods (Biggs et al. 2015).

In order to address this shortcoming regarding the WEF nexus, we propose a research framework that explicity considers biodiversity and social systems as part of the WEF nexus in what we present here as the FWEBS nexus (figures 2 and 3). It is expected that a FWEBS research framework that explicitly accounts for biodiversity and social systems will allow us to more comprehensively examine trade-offs and opportunities with various climate change and climate mitigation scenarios including BECCS. We anticipate that others can adapt the FWEBS framework for application and testing in other regions, including low-, middle-, and high-income countries. In addition, it is expected that the FWEBS framework can be widely applied by practitioners, scientists, and policymakers to develop and monitor policy and management plans in regional- and global-climate and sustainable-development agendas.

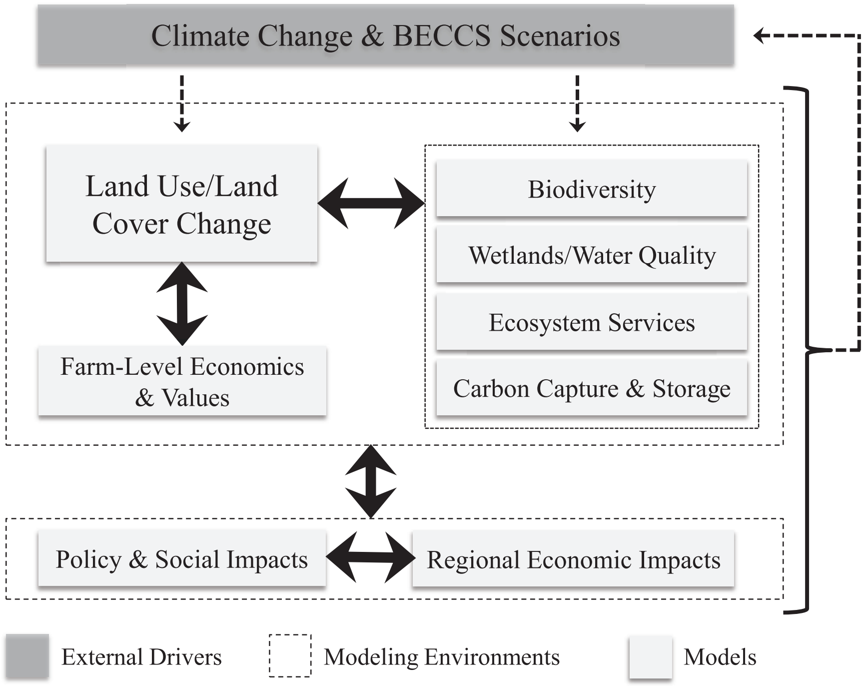

Figure 3. The interaction among climate change and bioenergy with carbon capture and storage (BECCS) scenarios, with key attributes of the food, water, energy, biodiversity, and social systems (FWEBS) nexus, including the domain in which coupled interactions in the Upper Missouri River Basin will be modeled.

The Upper Missouri River Basin

For the purposes of this study, we consider the Upper Missouri River Basin to be upriver of confluence of the Missouri and Big Sioux Rivers in Sioux City, Iowa, excluding the Niobrara watershed. By any definition, the UMRB extends from the Crown-of-the-Continent headwaters in Montana and the Front Range of Wyoming to the Prairie Pothole region of North and South Dakota (figure 1). The UMRB as we define it is dominated by the states of Montana, North Dakota, South Dakota, and Wyoming (and small parts of Canada, Iowa, Minnesota, and Nebraska). It represents some 30% of wheat production in the United States, 13% of soybean production, 11% of cattle production, and 9% of corn production, the last concentrated in the eastern Dakotas. Most of the region is rural, and only Alaska has a lower population density among US states than Wyoming, Montana, North Dakota, and South Dakota. The largest city in the UMRB, Sioux Falls in South Dakota, has a population of approximately 175,000. The UMRB encompasses diverse land uses and land-use trajectories, climate attributes, and social and cultural geographies, as well as carbon capture and storage (CCS) potential, all of which must be considered when understanding the consequences and opportunities of BECCS.

Land management. Over the past decade, land-use practices in the agricultural and industrial sectors of the UMRB have responded to policy drivers, markets (especially the amenities market), commodity price cycles, climate variability, and energy production, among other factors. Regional elasticity to market pressures appears to be high, as has been illustrated by recent conversion rates between grassland and cropland (figures 4 and 5; Wright and Wimberly 2013). Agricultural land in the region has been exiting the Conservation Reserve Program (CRP) at increasing rates (figure 5), with over 50% (17,000 square kilometers) of enrolled land exiting the program since 2007 because of declining federal enrollment caps, expiring CRP acreage, and economic incentives to plant, largely to corn and soybean (Morefield et al. 2016). Such conversions from extensive to intensive land uses are associated with negative consequences for soil C sequestration and biodiversity (Claassen 2011). Expansion of oil and gas production since the mid-2000s has also created new hybrid landscapes in which agricultural- and energy-production demands for water and land intersect in complex ways.

Land management across the UMRB changes distinctly from west to east, and more than 20 Native American tribes manage tens of thousands of square kilometers within the UMRB (figure 1). The capacity of tribes to influence regional land- and water-use patterns is gaining momentum, as has been demonstrated, for example, by the active restoration of native species on tribal lands and worldwide sympathy for the Water Protectors movement (e.g., Elbein 2017). Together, these trends add complexity to the social dimensions of land management (Hendrickson et al. 2016) and their influence on the FWEBS nexus in a rapidly changing region with ongoing fossil-fuel extraction (Jackson et al. 2014) and associated CCS potential.

Climate. High decadal climate variability and warming temperature trends, especially during winter (figure 6), are superimposed on this matrix of changing land cover (Mehta et al. 2013), raising concerns about the resiliency of existing socioeconomic systems and food security faced with unprecedented climate change (Seifert and Lobell 2015, Cook et al. 2015). Interestingly, climatological summer (June, July, and August) temperatures may have cooled across parts of the UMRB from the 1970s until 2015 (figure 6), similar to the adjacent Canadian Prairie Provinces, for reasons thought to be due in part to changes in land management, including the reduction of summer fallow and the widespread adoption of no-till agriculture (Gameda et al. 2007, Vick et al. 2016), although 2017 brought an acute summer drought to much of

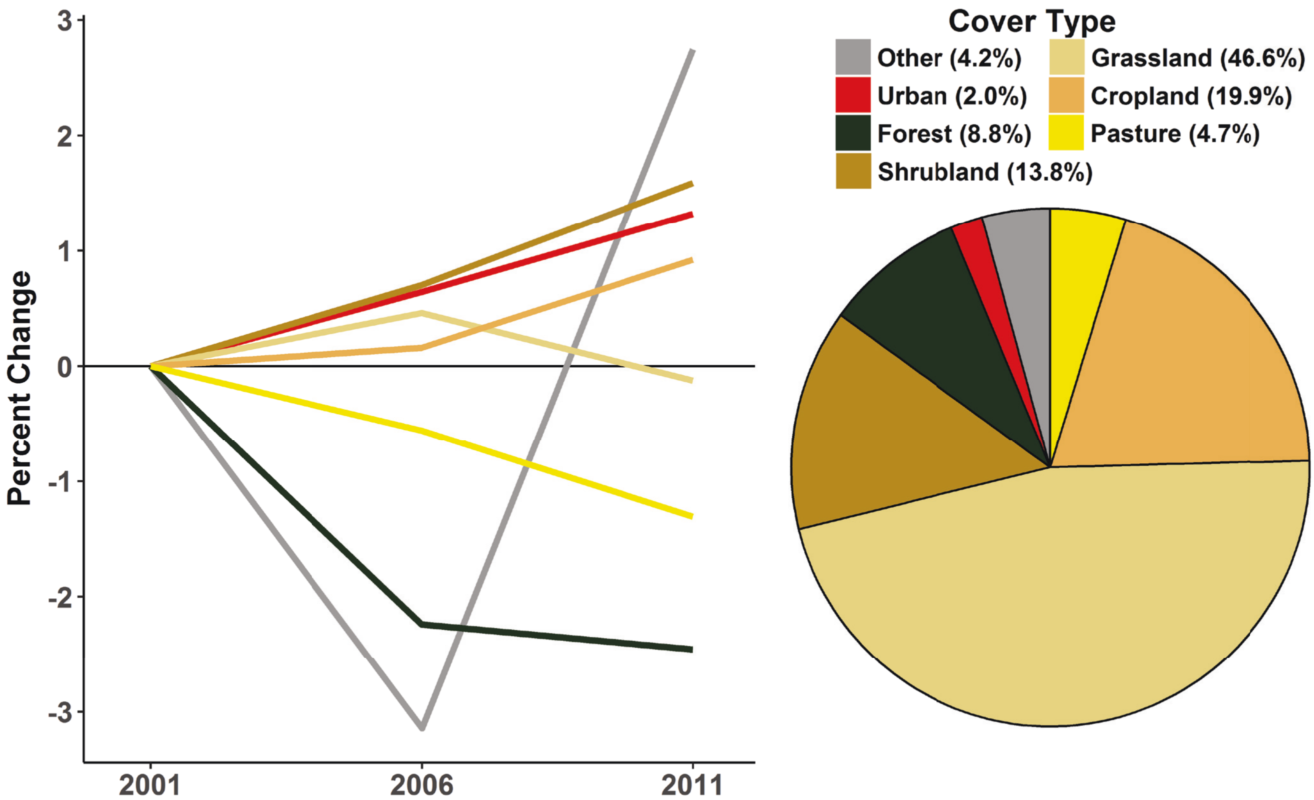

Figure 4. Recent trends in land cover (2001–2011) and the percentage of total land-cover area (2011) in the Upper Missouri River Basin. The cover classes of similar type were aggregated to a common class (e.g., four urban classes were collapsed into a single class). The "other" cover class includes water, wetlands, and barren and are subject to the interannual variability of the exposed shoreline of reservoirs, as well as misclassification errors given the ephemerality of wetlands and/or irrigation practices. The data were obtained from the National Land Cover Database (Homer et al. 2007, Fry et al. 2012, Homer et al. 2015).

the UMRB. General circulation models (GCMs) agree that annual average temperatures in the UMRB will continue to increase, using the bias-corrected ensemble Representative Concentration Pathway (RCP) 8.5 predictions as an upper limit to expected future temperature changes in figure 7, but it remains unclear how future changes in land management, including BECCS strategies, will affect water, energy, and GHG balances and thereby global and regional climate (Hallgren et al. 2013, DeLucia 2015).

Carbon capture and storage. Carbon capture and storage efforts can be internal or external to any region for global BECCS to take place (e.g., Muratori et al. 2016). The UMRB and surrounding regions have extensive carbon storage potential in geologic formations (Litynski et al. 2009), and a number of CCS test sites have been established by the Big Sky Carbon Sequestration Partnership in carbonate formations (e.g., Kevin Dome, Montana), in deep basalts in Washington State, in depleted oil reservoirs or for enhanced oil recovery, and with respect to enhanced coal-bed methane in the Powder River Basin of Montana and Wyoming within the UMRB, where it was found that additional incentives were required to make CCS economical. Initial storage resource estimations indicate large storage potential, but implementation of the Environmental Protection Agency's Underground Injection Control (UIC) Class VI regulations for CO_2 injection defines underground drinking water sources by salinity only, not allowing exemptions available under other UIC well classes. This rule will reduce the geologic carbon storage potential in the UMRB owing to fresh water recharge of formations at basin edges. The UMRB also has the potential to store C in agricultural soils given the widespread adoption of no-till agriculture (West and Post 2002, Watts et al. 2011) and the ongoing decline of the practice of summer fallow, which represents a source of CO_2 to the atmosphere (Merrill et al. 1999, Vick et al. 2016). In other words, select CCS efforts are possible within the UMRB and interact with the FWEBS nexus.

Food, water, energy, biodiversity, and social systems in the Upper Missouri River Basin

We discuss the FWEBS nexus as it applies to the UMRB sequentially, noting of course the interactions among food, water, energy, biodiversity, and social systems that we highlight in part in supplemental appendix S1.

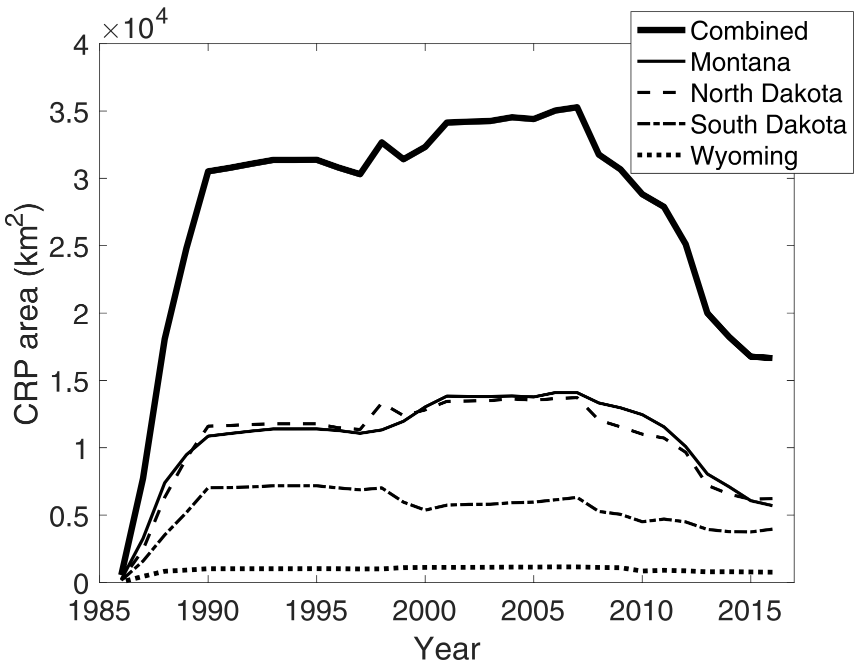

Figure 5. Trends in conservation reserve program (CRP) areal extent in the four states that constitute the greatest area of the Upper Missouri River Basin, as we defined in figure 1.

Food. BECCS presents unique opportunities and trade-offs with the FWEBS nexus in the UMRB (figure 2). Agriculture in the western UMRB is concentrated on the production of feed crops and animal products, with limited inroads by bioenergy production at the present, mainly due to the high value placed on food and, to some degree, climatic conditions. Bioenergy production is currently more prominent in the eastern UMRB and is largely derived from standard agricultural row crops, such as corn-grain ethanol. Common crops in the western UMRB include winter and spring wheat, with a growing influence of "pulse" legumes, such as lentils and peas (Burgess et al. 2012). Corn and soybeans dominate the eastern UMRB and continue to increase in area (figure 4). Large swaths of the UMRB remain in native grasslands used for range-cattle production (Gascoigne et al. 2013).

More diverse cropping systems, including pulse crops, are improving regional soil quality in the western UMRB (Miller et al. 2015), especially versus alternative management practices such as summer fallow, which is still common in parts of Montana but detrimental to soil C (Merrill et al. 1999, Vick et al. 2016). If managed appropriately, fallow replacement with pulses can grant economic benefits to producers, resulting in a win–win from both economic and climate perspectives (Bagley et al. 2015, Miller et al. 2015). Increases in the areal extent of pulse crops and oilseed bioenergy production have followed incentives from the US Farm Bill, but it remains to be seen whether enhanced bioenergy and pulse cropping is economically viable in a variable climate (Cutforth et al. 2007) and whether biofertilizers, such as N-fixing cyanobacteria, could improve nutrient management (Bhat et al. 2015). The consequences of BECCS strategies for regional biogeochemical cycles, particularly those of carbon and nitrogen, have not been studied to date.

Water. Water resource management faces multiple challenges across the UMRB, including intersectoral competition between energy production, agriculture, biodiversity, and utilities as well as interjurisdictional competition among states and between states and sovereign Native American nations. The consequences of water competition are exacerbated by institutional failures, such as overallocation of ground- and surface-water resources and major difficulties in adjudicating interjurisdictional and Tribal water rights. The response of water-use issues to a BECCS economy given current conflicts and with a changing climate requires additional research (Smith et al. 2015).

Trends in water quality emphasize the scalar mismatch between land-use dynamics and existing governance frameworks (Allred et al. 2015). For example, the onset of new land and water uses associated with the rapid expansion of hydraulic fracturing activities in the region revealed the limits of existing regulatory frameworks and the limited capacity of state and local governments for oversight, monitoring, and enforcement. Environmental monitoring provides insight about aggregate land-use effects such as the management of resource extraction and energy production waste (Bauder et al. 1993, Stackpoole et al. 2014) and would need to be expanded to account for additional impacts of BECCS strategies on agricultural and industrial practices, as well as biodiversity and other FWEBS attributes.

Energy. The energy industry of the UMRB is dominated by conventional systems, namely fossil fuels and large-scale hydropower, despite substantial solar and wind resources (Elliot et al. 1992, Lopez et al. 2012). For example, the Colstrip power plant in eastern Montana is the second-largest coal-fired generating facility west of the Mississippi River and produces approximately 45% of Montana's total CO2 emissions. The energy industry is changing rapidly (e.g., two units of the Colstrip plant are slated for decommissioning), providing new opportunities such as retrofitting power generators to use alternative fuels or spare transmission capacity for development of new generation facilities (Cao and Caldeira 2010).

The dramatic expansion of oil and gas extraction in the UMRB includes the mid-2000s coal-bed methane boom in the Powder River Basin and the 2004–2014 Bakken shale-oil

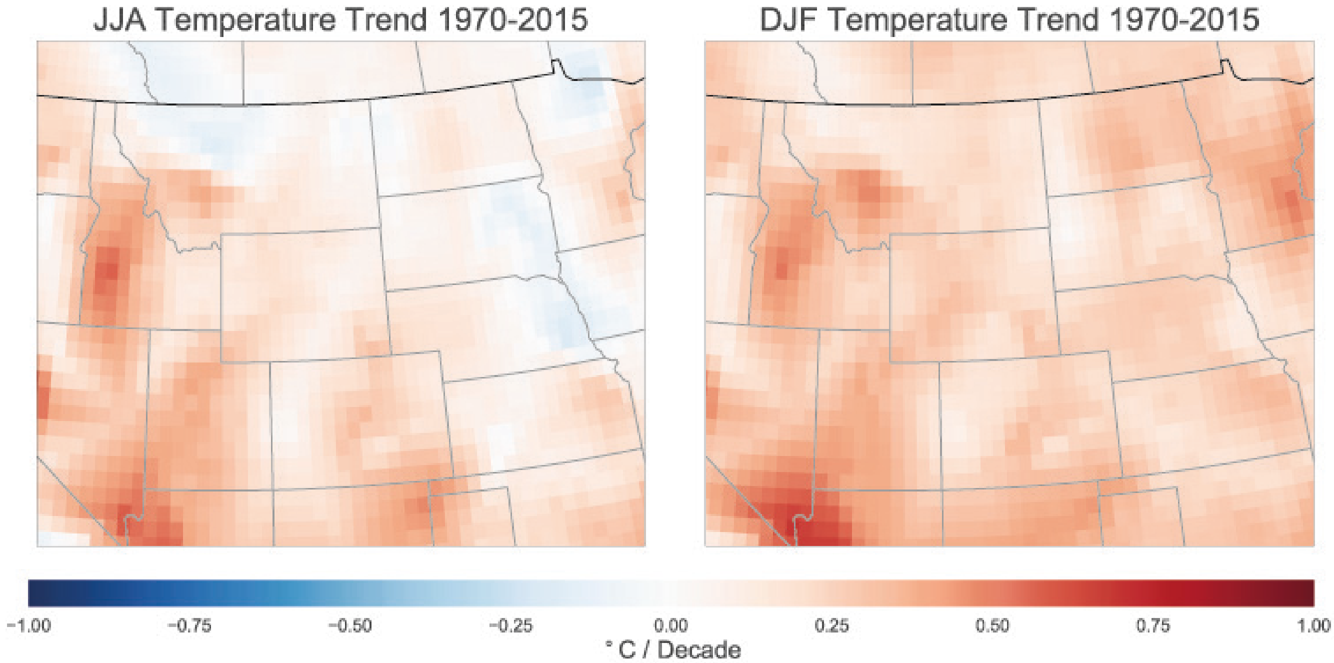

Figure 6. Decadal trends in summer (JJA) and winter (DJF) temperature from 1970 until 2015 in the region, including and surrounding the Upper Missouri River Basin (figure 1) from the Climatic Research Unit (CRU) database (Harris et al. 2013).

boom. These activities have resulted in an approximately 700% increase in regional crude-oil production between 2000 and 2017 and nearly a 400% increase in natural-gas production, along with new pressures on already limited water resources (Jackson et al. 2014). Energy production could potentially be coupled with geological CCS (Eccles et al. 2012) or the removal of atmospheric CO_2 by ecosystems (Zhu et al. 2014), with both approaches demonstrating high potential in the UMRB (West and Post 2002, Litynski et al. 2009).

The feasibility of CCS, via public and political acceptance of such technology and its risks, is not clearly quantified. Using natural ecosystems to store carbon may also be problematic because of climatic constraints within the UMRB that limit net primary production. Potential reductions in carbon storage in carbon-rich grasslands converted to crops or woody vegetation must be taken into consideration when accounting for net atmospheric CO_2 removal (Jackson et al. 2002, Gelfand et al. 2011). The existing matrix of coal- and natural-gas-based energy production and carbon sequestration from geologic and natural ecosystems in the UMRB provides a rich opportunity for interdisciplinary research (Humpenöder et al. 2014).

Bioenergy expansion in the western UMRB would require substantial economic incentives because of strong and sustained markets for high-quality food production, particularly cereals and beef. Bioenergy production may also become more financially competitive under projected climate change or with advancements in new bioenergy (including biofuel) crop cultivars (Berdahl et al. 2005, Gesch et al. 2015). The expanded adoption of bioenergy ultimately rests on economic viability but also intersects with cultural values, including biodiversity protection, that likewise influence decision-making.

Biodiversity. It is estimated that 70% of the grasslands in the Great Plains have been converted to other land uses. Those that remain are crucial reservoirs of biodiversity (Samson et al. 2004). The UMRB has attracted public and private ecological restoration efforts at local to landscape scales, but recent reductions of Conservation Reserve Program (CRP) lands (figure 5), native grasslands, and wetlands (Johnston 2013, Wright and Wimberly 2013) are key examples of how quickly land management can respond to economic drivers and associated changes in policy. Intensively managed agricultural landscapes can provide habitat, but conversion of CRP, native grasslands, and wetlands to agriculture—especially row-crop production (Brown et al. 2005)—can have strong negative impacts on biodiversity (Best et al. 1995, Lehtinen et al. 1999). These impacts extend beyond direct habitat loss (see supplemental appendix S1); for example, water quality and contaminant exposure pose a range of serious risks to amphibians, from direct mortality (Relyea 2005) to endocrine disruption (Hayes et al. 2002), emphasizing the need to study connections within the FWEBS nexus.

Social systems. It is expected that BECCS expansion in the UMRB will influence social systems via impacts on farm

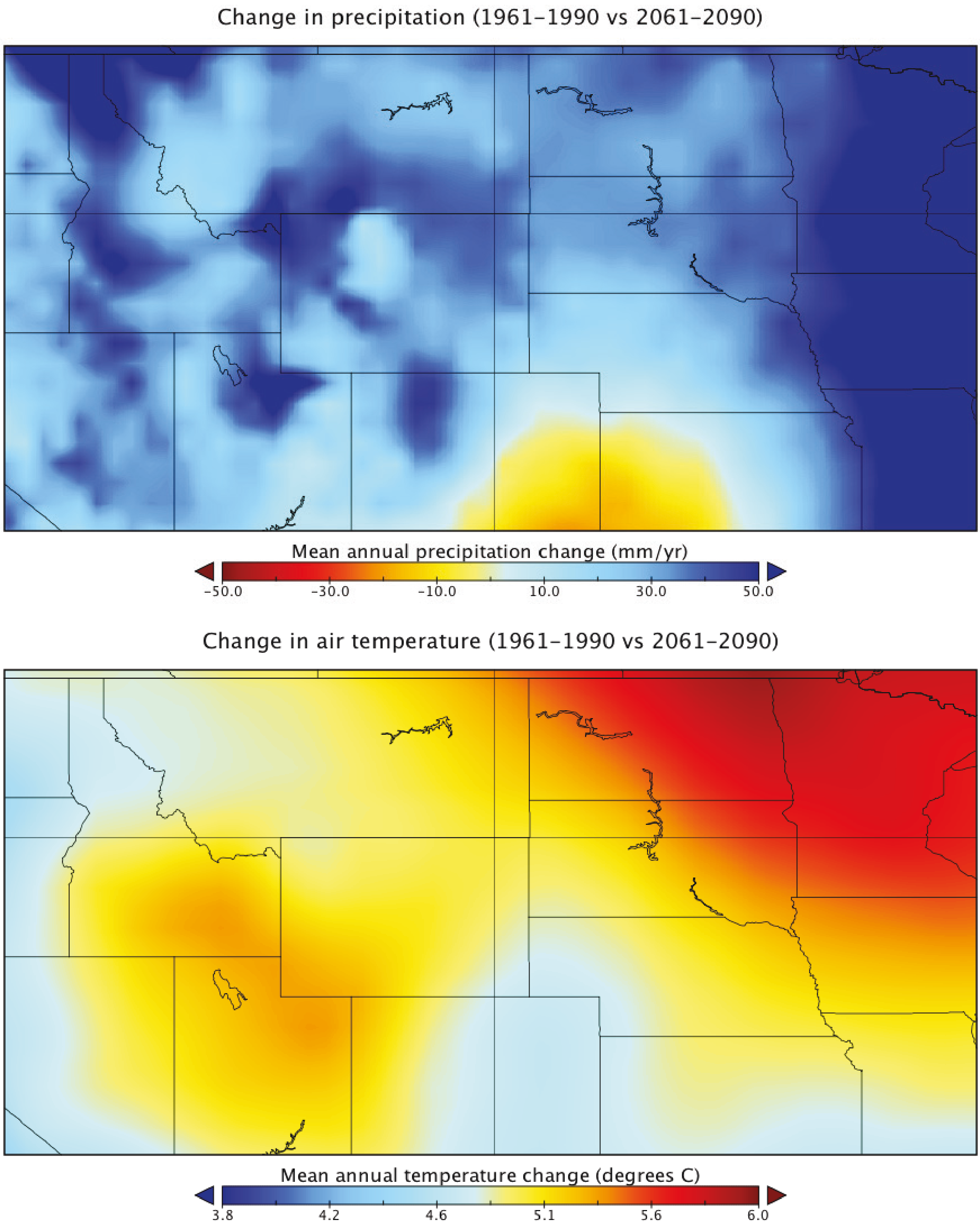

Figure 7. Future climate under full Intergovernmental Panel on Climate Change Representative Concentration Pathway (RCP) 8.5 ensemble bias corrected using CRU and downscaled to 0.5 degrees resolution, following Poulter and colleagues (2010).

economics and overall livelihoods, competition for land and labor, working conditions and renumeration for workers, governmental policies, cultural ecosystem services, and food security. Some social systems, such as regional economics, are readily quantifiable and can be directly compared. Other social systems, such as values and traditions, are often less meaningful when expressed in monetary terms (Daily et al. 2009), but they have important social value (Bagstad et al. 2015) and play an important role in decision-making (Wainger et al. 2010). For example, Native American and rural communities in Montana rely on hunting and harvesting of wild edible plants for cultural identity, food sovereignty, family ties to previous generation, and health benefits (Byker Shanks et al. 2015). Considering diverse stakeholder perspectives, attitudes, and decisions in response to the potential expansion of BECCS in the UMRB will allow us to elucidate barriers and opportunities for BECCS implementation. For example, meat production, including rangeland and cropland for growing animal feed, is the largest land use in the eastern UMRB, and much of this land could be used for bioenergy production (Langholtz et al. 2016), but there are strongly held values toward animal agriculture and meat consumption that make such land-use changes more difficult (Foley et al. 2011, Turner et al. 2014, Langholtz et al. 2016). Previous research suggests that bioenergy expansion can compete for land and labor resources and result in increased food prices that ultimately lead to higher food insecurity, particularly for low-income and landless populations as affordable food becomes less accessible (Müller et al. 2008, Ewing and Msangi 2009). On the other hand, higher food prices can stimulate the agricultural sector and create new opportunities for rural communities (Müller et al. 2008), including increased purchasing power and enhanced resilience to market instability (Ewing and Msangi 2009).

In summary, all elements of the FWEBS nexus interact with BECCS strategies in the UMRB and elsewhere, and understanding the complex trade-offs, as well as opportunities, of multiple BECCS approaches across different spatial and temporal scales requires careful attention to each attribute as well as their interactions.

Developing regional bioenergy with carbon capture and storage scenarios for assessing ecological and socioeconomic interactions

To examine the critical trade-offs and opportunities of alternative BECCS strategies within the FWEBS nexus at regional scales such as the UMRB, researchers must define a set of plausible scenarios for achieving negative CO_2 emissions. The definition of scenarios has itself become a complex area of study, with varying definitions of what constitutes a scenario across different disciplines and applications (van Vuuren et al. 2012). The general strategy for developing scenarios for global-change assessment typically involves using qualitative descriptions, such as narratives or storylines, that characterize a broad array of possible futures and then developing increasingly quantitative assumptions consistent with the broad narratives to inform specific modeling exercises (Moss et al. 2010, Rounsevell and Metzger 2010). Increasingly, interdisciplinary processes are being used to develop scenarios with more robust qualitative and quantitative assumptions and better recognition of feedback processes in human and ecological systems, such as the latest SSPs for assessing climate mitigation and adaptation (O'Neill et al. 2017). Despite substantial efforts in scenario development, "downscaling" broad narratives to regional scales remains a challenge, because broad narratives do not easily align with local contexts (Kriegler et al. 2012).

Rather than propose specific quantitative scenarios here, we discuss general narratives for developing scenarios that can inform a regional analysis of BECCS impacts on FWEBS in the UMRB. Achieving net negative CO_2 emissions in the UMRB could conceivably be achieved by implementing a wide range of mitigation and adaptation measures, although as we have noted, these may conflict with other management goals (figure 2). We propose, as a starting point, two general narratives that capture the extremes of a continuum of BECCS-related strategies. At one extreme, an *aggressive BECCS approach* would emphasize technological and land-intensive approaches, including geological CCS, producing bioenergy crops for electricity and fuel (to displace fossil sources) and increasing electricity production from renewable sources as part of a broader energy transition (figure 2). At the other extreme, a *conservation BECCS approach* would emphasize more land-extensive approaches, including biological and geological carbon sequestration through soil-management practices and CCS (Chabbi et al. 2017), afforestation and avoided land conversion, and the production of perennial cellulosic bioenergy crops. Whereas the conservation BECCS approach may miss some opportunities to sequester C, such a strategy may align BECCS with other ecosystem services and cultural values, including biodiversity conservation. These general narratives provide a framework for assessing FWEBS trade-offs and opportunities along a continuum of quantitative scenarios between aggressive and conservation, all of which can be compared to business-as-usual or status-quo alternatives. The general narratives also fit within, and must ultimately be consistent with, existing broader global-change storylines, such as the latest RCP and SSP storylines (O'Neill et al. 2017).

Crucial to refining quantitative BECCS scenarios for analyzing potential future conditions in the UMRB is an appreciation for local context—local socioeconomic conditions, technologies, and institutions—which ultimately determines the feasibility and impacts of alternative BECCS strategies. Incorporating such local context will ultimately require an iterative process, including interdisciplinary scientists and local stakeholder experts, whereby scenario assumptions are tested and refined both through modeling exercises and stakeholder feedback (Sleeter et al. 2012). The interactions between local attributes of the FWEBS nexus and human response will determine the extent to which aggressive, conservation, or other BECCS strategies are technically feasible, socially acceptable, and economically sustainable. By working with local experts and stakeholders in an iterative process, researchers can define a limited set of alternative quantitative scenarios that can achieve net negative CO_2 emissions (if technically possible) and, given those scenarios, determine the key FWEBS trade-offs needed to guide regional-scale policymaking. Such an effort must also point to synergistic interactions that may provide opportunities to improve multiple factors in the FWEBS nexus (figure 2).

How will different elements of the FWEBS nexus change as BECCS development becomes more prominent, and, as has been demonstrated by the case study of biodiversity (appendix S1), could "conservation" BECCS scenarios be developed that satisfy multiple societal objectives (figure 2)? Alternatively, are aggressive BECCS strategies necessary to mitigate climate warming such that hard compromises will have to be made regarding FWEBS and other ecosystem and Earth-system services (Boysen et al. 2017, Rockström et al. 2017)? We hypothesize that business-as-usual strategies provide insufficient atmospheric C removal and aggressive BECCS strategies may present too many conflicts with the FWEBS nexus to become adopted. Thus, a conservation BECCS strategy that relies on a balanced array of BECCS activities (from geological and biological CSS to cellulosic ethanol and non-BECCS renewable energy) designed to minimize socioeconomic trade-offs while simultaneously benefitting biodiversity conservation may be the only realistic approach to serve multiple societal objectives in the UMRB and likely other global regions. Testing such a hypothesis requires a highly multidisciplinary approach that combines surveys and interviews of perceptions to BECCS and data-informed models of economic, biogeochemical, hydrological, biodiversity, and climate systems that capture the feedback loops and interrelationships between system drivers and outcomes (figure 3). New regulatory and incentivization approaches to guide multiple actors, including industry, governments, and individuals, toward behaviors that help us become positive actors in the climate system are ultimately needed. To do so, we must design BECCS strategies and contrast them against alternate strategies to find the correct balance among atmospheric C removal, likelihood of adoption, and ecological and socioeconomic sustainability.

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Supplemental material

Supplementary data are available at *BIOSCI* online.

References cited

Albanito F, Beringer T, Corstanje R, Poulter B, Stephenson A, Zawadzka J, Smith P. 2015. Carbon implications of converting cropland to bioenergy crops or forest for climate mitigation: A global assessment. GCB Bioenergy 8: 81–95.

Allred BW, Kolby Smith W, Twidwell D, Haggerty JH, Running SW, Naugle DE, Fuhlendorf SD. 2015. Ecosystem services lost to oil and gas in North America. Science 348: 401–402.

Bagley JE, Miller J, Bernacchi CJ. 2015. Biophysical impacts of climate-smart agriculture in the Midwest United States. Plant, Cell, and Environment 38: 1913–1930.

Bagstad KJ, Reed JM, Semmens DJ, Sherrouse BC, Troy A. 2015. Linking biophysical models and public preferences for ecosystem service assessments: A case study for the southern Rocky Mountains. Regional Environmental Change 16: 2005–2018.

Barros VR, et al, eds. 2014. Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part B: Regional Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press.

Bauder JW, Sinclair KN, Lund RE. 1993. Physiographic and land use characteristics associated with nitrate–nitrogen in Montana groundwater. Journal of Environmental Quality 22: 255–262.

Baum SD. 2014. The great downside dilemma for risky emerging technologies. Physica Scripta 89 (art. 128004).

Berdahl JD, Frank AB, Krupinsky JM, Carr PM, Hanson JD, Johnson HA. 2005. Biomass yield, phenology, and survival of diverse switchgrass cultivars and experimental strains in western North Dakota. Agronomy Journal 97: 549–555.

Best LB, Freemark KE, Dinsmore JJ, Camp M. 1995. A review and synthesis of habitat use by breeding birds in agricultural landscapes of Iowa. American Midland Naturalist 134: 1–29.

Bhat TA, Ahmad D, Ganai A, Khan OA. 2015. Nitrogen fixing biofertilizers; mechanism and growth promotion: A review. Journal of Pure Applied Microbiology 9: 1675–1690.

Biggs EM, et al. 2015. Sustainable development and the water–energy–food nexus: A perspective on livelihoods. Environmental Science and Policy 54: 389–397.

Bonsch M, et al. 2014. Trade-offs between land and water requirements for large-scale bioenergy production. Global Change Biology: Bioenergy 8: 11–24.

Boysen LR, Lucht W, Gerten D. 2017. The limits to global-warming mitigation by terrestrial carbon removal. Earth's Future 5: 463–474.

Brown DG, Johnson KM, Loveland TR, Theobald DM. 2005. Rural land-use trends in the conterminous United States, 1950–2000. Ecological Applications 15: 1851–1863.

Burgess MH, Miller PR, Jones CA. 2012. Pulse crops improve energy intensity and productivity of cereal production in Montana, USA. Journal of Sustainable Agriculture 36: 699–718.

Byker Shanks C, Smith T, Ahmed S, Hunts H. 2015. Assessing foods offered in the Food Distribution Program on Indian Reservations (FDPIR) using the Healthy Eating Index 2010. Public Health Nutrition 19: 1315–1326.

Cao L, Caldeira K. 2010. Atmospheric carbon dioxide removal: Long-term consequences and commitment. Environmental Research Letters 5 (art. 024011).

Chabbi A, et al. 2017. Aligning agriculture and climate policy. Nature Climate Change 7: 307–309.

Claassen RL, Carriazo F, Cooper JC, Hellerstein D, Ueda K. 2011. Grassland to Cropland Conversion in the Northern Plains: The Role of Crop Insurance, Commodity, and Disaster Programs. US Department of Agriculture Economic Research Service.

Collins WD, et al. 2015. The integrated Earth System Model (iESM): Formulation and functionality. Geoscientific Model Development Discussions 8: 381–427.

Cook BI, Ault TR, Smerdon JE. 2015. Unprecedented 21st century drought risk in the American Southwest and Central Plains. Science Advances 1 (art. e1400082).

Cutforth HW, McGinn SM, McPhee KE, Miller PR. 2007. Adaptation of pulse crops to the changing climate of the Northern Great Plains. Agronomy Journal 99: 1684–1699.

Daily GC, Polasky S, Goldstein J, Kareiva PM, Mooney HA, Pejchar L, Ricketts TH, Salzman J, Shallenberger R. 2009. Ecosystem services in decision making: Time to deliver. Frontiers in Ecology and the Environment 7: 21–28.

DeLucia EH. 2015. How biofuels can cool our climate and strengthen our ecosystems. Eos 96: 14–19.

Eccles JK, Pratson L, Newell RG, Jackson RB. 2012. The impact of geologic variability on capacity and cost estimates for storing CO_2 in deep-saline aquifers. Energy Economics 34: 1569–1579.

Elbein S. 2017. These are the defiant "Water Protectors" of Standing Rock. National Geographic. (28 November 2017; *http://news.nationalgeographic.com/2017/01/tribes-standing-rock-dakota-access-pipeline-advancement*)

Elliot DL, Holladay CG, Barchet WR, Foote HP, Sandusky WF. 1992. Wind Energy Resource Atlas of the United States. Solar Technical Information Program, Solar Energy Research Institute. National Technical Information Service Publication no. NTIS-PR-360.

Ewing M, Msangi S. 2009. Biofuels production in developing countries: Assessing trade-offs in welfare and food security. Environmental Science Policy 12: 520–528.

Foley JA, et al. 2005. Global consequences of land use. Science 309: 570–574.

Foley JA, et al. 2011. Solutions for a cultivated planet. Nature 478: 337–342.

Fry J, Xian GZ, Jin S, Dewitz J, Homer CG, Yang L, Barnes CA, Herold ND, Wickham JD. 2012. Completion of the 2006 National Land Cover Database for the conterminous United States. Photogrammetric Engineering and Remote Sensing 77: 858–864.

Fuss S, et al. 2014. Betting on negative emissions. Nature Climate Change 4: 850–853.

Fuss S, Reuter WH, Szolgayová J, Obersteiner M. 2013. Optimal mitigation strategies with negative emission technologies and carbon sinks under uncertainty. Climate Change 118: 73–87.

Galaz V. 2012. Geo-engineering, governance, and social–ecological systems: Critical issues and joint research needs. Ecology and Society 17 (art. 24). doi:10.5751/es-04677-170124

Gameda S, Qian B, Campbell CA, Desjardins RL. 2007. Climatic trends associated with summerfallow in the Canadian Prairies. Agricultural and Forest Meteorology 142: 170–185.

Gascoigne WR, Hoag DLK, Johnson RR, Koontz LM, Thomas CC. 2013. Land-Use Change, Economics, and Rural Well-Being in the Prairie Pothole Region of the United States. US Geological Survey. Fact Sheet no. 2013-3046.

Gelfand I, Zenone T, Jasrotia P, Chen J, Hamilton SK, Robertson GP. 2011. Carbon debt of Conservation Reserve Program (CRP) grasslands converted to bioenergy production. Proceedings of the National Academy of Sciences 108: 13864–13869.

Gesch RW, et al. 2015. Comparison of several *Brassica* species in the north central US for potential jet fuel feedstock. Industrial Crops and Products 75: 2–7.

Hallgren W, Schlosser CA, Monier E, Kicklighter D, Sokolov A, Melillo J. 2013. Climate impacts of a large-scale biofuels expansion. Geophysical Research Letters 40: 1624–1630.

Harris I, Jones PD, Osborn TJ, Lister DH. 2013. Updated high-resolution grids of monthly climatic observations: The CRU TS3.10 Dataset. International Journal of Climatology 34: 623–642.

Hayes T, Haston K, Tsui M, Hoang A, Haeffele C, Vonk A. 2002. Herbicides: Feminization of male frogs in the wild. Nature 419: 895–896.

Hendrickson JR, Elk LB, Faller T. 2016. Development of the renewal on the Standing Rock Sioux Reservation Project. Rangelands 38: 1–2.

Homer CG, Dewitz JA, Fry J. 2007. Completion of the 2001 National Land Cover Database for the conterminous United States. Photogrammetric Engineering and Remote Sensing 73: 337–341.

Homer CG, Dewitz JA, Yang L, Jin S, Danielson P, Xian G, Coulston J, Herold N, Wickham J, Megown K. 2015. Completion of the 2011 National Land Cover Database for the conterminous United States—Representing a decade of land cover change information. Photogrammetric Engingeering and Remote Sensing 81: 345–354.

Hulme M. 2016. 1.5°C and climate research after the Paris Agreement. Nature Climate Change 6: 222–224.

Humpenöder F, Popp A, Dietrich JP, Klein D, Lotze-Campen H, Bonsch M, Bodirsky BL, Weindl I, Stevanovic M, Müller C. 2014. Investigating afforestation and bioenergy CCS as climate change mitigation strategies. Environmental Research Letters 9 (art. 064029).

[IPCC] Intergovernmental Panel on Climate Change. 2014. Synthesis report. Contribution of Working Groups I, II, and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. IPCC.

Jackson RB, Banner JL, Jobbágy EG, Pockman WT, Wall DH. 2002. Ecosystem carbon loss with woody plant invasion of grasslands. Nature 418: 623–626.

Jackson RB, Vengosh A, Carey JW, Davies RJ, Darrah TH, O'Sullivan F, Pétron G. 2014. The environmental costs and benefits of fracking. Annual Review of Environment and Resources 39: 327–362.

Johnston CA. 2013. Wetland losses due to row crop expansion in the Dakota Prairie Pothole Region. Wetlands 33: 175–182.

Jones AD, et al. 2013. Greenhouse gas policy influences climate via direct effects of land-use change. Journal of Climate 26: 3657–3670.

Kriegler E, O'Neill BC, Hallegatte S, Kram T, Lempert RJ, Moss RH, Wilbanks T. 2012. The need for and use of socio-economic scenarios for climate change analysis: A new approach based on shared socio-economic pathways. Global Environmental Change 22: 807–822.

Kriegler E, Edenhofer O, Reuster L, Luderer G, Klein D. 2013. Is atmospheric carbon dioxide removal a game changer for climate change mitigation? Climate Change 118: 45–57.

Langholtz MH, Stokes BJ, Eaton LM. 2016. Billion-Ton Report: Advancing Domestic Resources for a Thriving Bioeconomy. US Department of Energy Office of Energy Efficiency and Renewable Energy.

Lehtinen RM, Galatowitsch SM, Tester JR. 1999. Consequences of habitat loss and fragmentation for wetland amphibian assemblages. Wetlands 19: 1–12.

Le Quéré C, et al. 2013. The global carbon budget 1959–2011. Earth System Science Data 5: 165–185.

Litynski JT, Plasynski S, Spangler L, Finley R, Steadman E, Ball D, Nemeth KJ, McPherson B, Myer L. 2009. The United States Department of Energy's Regional Carbon Sequestration Partnerships program: Overview. Energy Procedia 1: 3959–3967.

Lopez A, Roberts B, Heimiller D, Blair N, Porro G. 2012. US Renewable Energy Technical Potentials: A GIS-Based Analysis. US Department of Energy Office of Energy Efficiency and Renewable Energy, National Renewable Energy Laboratory. Technical Report no. NREL/TP-6A20-51946.

Lotze-Campen H, et al. 2013. Impacts of increased bioenergy demand on global food markets: An AgMIP economic model intercomparison. Agricultural Economics 45: 103–116.

Mehta VM, Knutson CL, Rosenberg NJ, Olsen JR, Wall NA, Bernadt TK, Hayes MJ. 2013. Decadal climate information needs of stakeholders for decision support in water and agriculture production sectors: A case study in the Missouri River Basin. Weather, Climate, and Society 5: 27–42.

Meinshausen M, Meinshausen N, Hare W, Raper SCB, Frieler K, Knutti R, Frame DJ, Allen MR. 2009. Greenhouse-gas emission targets for limiting global warming to 2°C. Nature 458: 1158–1162.

Merrill SD, Black AL, Fryrear DW, Saleh A, Zobeck TM, Halvorson AD, Tanaka DL. 1999. Soil wind erosion hazard of spring wheat–fallow as affected by long-term climate and tillage. Soil Science Society of America Journal 63: 1768–1777.

Miller PR, Bekkerman A, Jones CA, Burgess MH, Holmes JA, Engel RE. 2015. Pea in rotation with wheat reduced uncertainty of cconomic returns in Southwest Montana. Agronomy Journal 107: 541–550.

Morefield PE, LeDuc SD, Clark CM, Iovanna R. 2016. Grasslands, wetlands, and agriculture: The fate of land expiring from the Conservation

Reserve Program in the Midwestern United States. Environmental Research Letters 11 (art. 094005).
Moss RH, et al. 2010. The next generation of scenarios for climate change research and assessment. Nature 463: 747–756.
Müller A, Schmidhuber J, Hoogeveen J, Steduto P. 2008. Some insights in the effect of growing bio-energy demand on global food security and natural resources. Water Policy 10: 83–94.
Muratori M, Calvin K, Wise M, Kyle P, Edmonds J. 2016. Global economic consequences of deploying bioenergy with carbon capture and storage (BECCS). Environmental Research Letters 11 (art. 095004).
O'Neill BC, et al. 2017. The roads ahead: Narratives for shared socioeconomic pathways describing world futures in the 21st century. Global Environmental Change 42: 169–180.
Popp A, et al. 2014. Land-use protection for climate change mitigation. Nature Climate Change 4: 1095–1098.
Poulter B, Aragão L, Heyder U, Gumpenberger M, Heinke J, Langerwisch F, Rammig A, Thonicke K, Cramer W. 2010. Net biome production of the Amazon Basin in the 21st century. Global Change Biology 16: 2062–2075.
Powell TWR, Lenton TM. 2013. Scenarios for future biodiversity loss due to multiple drivers reveal conflict between mitigating climate change and preserving biodiversity. Environmental Research Letters 8 (art. 025024).
Relyea RA. 2005. The lethal impacts of Roundup and predatory stress on six species of North American tadpoles. Archives of Environmental Contamination and Toxicology 48: 351–357.
Rhodes JS, Keith DW. 2008. Biomass with capture: Negative emissions within social and environmental constraints: An editorial comment. Climate Change 87: 321–328.
Riahi K, et al. 2017. The Shared Socioeconomic Pathways and their energy, land use, and greenhouse gas emissions implications: An overview. Global Environmental Change 42: 153–168.
Rockström J, Gaffney O, Rogelj J, Meinshausen M, Nakicenovic N, Schellnhuber HJ. 2017. A roadmap for rapid decarbonization. Science 355: 1269–1271.
Rogelj J, den Elzen M, Höhne N, Fransen T, Fekete H, Winkler H, Schaeffer R, Sha F, Riahi K, Meinshausen M. 2016. Paris Agreement climate proposals need a boost to keep warming well below 2°C. Nature 534: 631–639.
Rounsevell MDA, Metzger MJ. 2010. Developing qualitative scenario storylines for environmental change assessment. Wiley Interdisciplinary Reviews: Climate Change 1: 606–619.
Samson FB, Knopf FL, Ostlie WR. 2004. Great Plains ecosystems: Past, present, and future. Wildlife Society Bulletin 32: 6–15.
Scanlon BR, Ruddell BL, Reed PM, Hook RI, Zheng C, Tidwell VC, Siebert S. 2017. The food–energy–water nexus: Transforming science for society. Water Resources Research 53: 3550–3556. doi:10.1002/2017WR020889
Schellnhuber HJ. 1999. "Earth system" analysis and the second Copernican revolution. Nature 402: C19–C23.
Scholes RJ. 2016. Climate change and ecosystem services. Wiley Interdisciplinary Reviews: Climate Change 7: 537–550.
Seifert CA, Lobell DB. 2015. Response of double cropping suitability to climate change in the United States. Environmental Research Letters 10 (art. 024002).
Sleeter BM, et al. 2012. Scenarios of land use and land cover change in the conterminous United States: Utilizing the special report on emission scenarios at ecoregional scales. Global Environmental Change 22: 896–914.
Smith P, et al. 2015. Biophysical and economic limits to negative CO_2 emissions. Nature Climate Change 6: 42–50.
Stackpoole SM, Stets EG, Striegl RG. 2014. The impact of climate and reservoirs on longitudinal riverine carbon fluxes from two major watersheds in the Central and Intermontane West. Journal of Geophysical Research: Biogeosciences 119: 848–863.
Tavoni M, et al. 2014. Post-2020 climate agreements in the major economies assessed in the light of global models. Nature Climate Change 5: 119–126.
Tian H, et al. 2016. The terrestrial biosphere as a net source of greenhouse gases to the atmosphere. Nature 531: 225–228.
Tilman D, Clark M. 2014. Global diets link environmental sustainability and human health. Nature 515: 518–522.
Turner BL, Wuellner M, Nichols T, Gates R. 2014. Dueling land ethics: Uncovering agricultural stakeholder mental models to better understand recent land use conversion. Journal of Agricultural and Environmental Ethics 27: 831–856.
Usgaard RE, Naugle DE, Osborn RG, Higgins KF. 1997. Effects of wind turbines on nesting raptors at Buffalo Ridge in southwestern Minnesota. Proceedings of the South Dakota Academy of Science 76: 113–117.
Van Vuuren DP, Deetman S, van Vliet J, van den Berg M, van Ruijven BJ, Koelbl B. 2013. The role of negative CO_2 emissions for reaching 2°C: Insights from integrated assessment modelling. Climate Change 118: 15–27.
Van Vuuren DP, Kok MTJ, Girod B, Lucas PL, de Vries B. 2012. Scenarios in global environmental assessments: Key characteristics and lessons for future use. Global Environmental Change 22: 884–895.
Vick ESK, Stoy PC, Tang ACI, Gerken T. 2016. The surface–atmosphere exchange of carbon dioxide, water, and sensible heat across a dryland wheat–fallow rotation. Agriculture, Ecosystems, and Environment 232: 129–140.
Wainger LA, King DM, Mack RN, Price EW, Maslin T. 2010. Can the concept of ecosystem services be practically applied to improve natural resource management decisions? Ecological Economics 69: 978–987.
Watts J, Lawrence R, Miller PR. 2011. An analysis of cropland carbon sequestration estimates for north central Montana. Climate Change 108: 301–331.
West TO, Post WM. 2002. Soil organic carbon sequestration rates by tillage and crop rotation. Soil Science Society of America Journal 66: 1930–1946.
Wright CK, Wimberly MC. 2013. Recent land use change in the Western Corn Belt threatens grasslands and wetlands. Proceedings of the National Academy of Sciences 110: 4134–4139.
Zhu Z, Reed BC, eds. 2014. Baseline and Projected Future Carbon Storage and Greenhouse-Gas Fluxes in Ecosystems of the Eastern United States. US Geological Survey. Professional Paper no. 1804.
Zilberman D. 2015. IPCC AR5 overlooked the potential of unleashing agricultural biotechnology to combat climate change and poverty. Global Change Biology 21: 501–503.

Paul C. Stoy (paul.stoy@gmail.com), Selena Ahmed, Gabriel Bromley, E. N. J. Brookshire, Julia Haggerty, Perry Miller, Brent Peyton, Alisa Royem, and Lee Spangler are affiliated with Montana State University. Meghann Jarchow, David Swanson, and Mark D. Dixon are affiliated with the University of South Dakota. Benjamin Rashford, Shannon Albeke, and Crista Straub are with the University of Wyoming. Benjamin Poulter is affiliated with NASA Goddard.

Appendix A: Bird populations as a case study of BECCS interactions

Bird populations within the UMRB present an ideal case study for understanding how changes in coupled food, energy, and water systems impact biodiversity and interact with societal values. Grasslands comprise a significant portion of the land cover in the UMRB (Figures 1 & 4) and grassland birds have undergone the greatest recent population declines of any avian habitat guild in North America (Sauer et al. 2013; Schipper et al. 2016). These declines are largely attributed to habitat loss and degradation (Samson and Knopf 1994; Hill et al. 2014), although other factors such as insecticide toxicity (Mineau and Whiteside 2013) and climate change (Gorzo et al. 2016) play important roles. Conversion of grasslands to cropland in the western Corn Belt of North America, including the eastern portions of the UMRB study area, has accelerated recently with high prices for corn and soybeans associated with the expanding biofuels industry (Wright and Wimberly 2013). Land-use change and river flow regulation in the UMRB associated with energy development (wind, biofuels, and hydropower) has affected regional biodiversity, including impacts on bird populations associated with grassland, wetland and riparian systems (Dixon et al. 2012; Fargione et al. 2012; Hill et al. 2014; Sohl 2014; Munes et al. 2015; Rashford et al. 2015). The impacts of geologic CCS on bird populations is less clear. Continued land use change in response to food, energy, and water pressures is likely to further affect bird populations and productivity, but these impacts are poorly known. Expansion of BECCS in the UMRB has the potential to greatly impact abundance and diversity for birds of grassland and other habitat types within the region. Particularly important to grassland birds are bioenergy crops and wind energy under BECCS scenarios, which would likely put more grasslands (including restored prairie, CRP grasslands, and dedicated bioenergy crops) on the landscape (Figure 2).

Other impacts of land cover change on bird biodiversity are indirectly related to human pressures. For example, native prairie provides high quality nesting habitat for grassland birds, but the extensive grasslands of the UMRB have been greatly fragmented and degraded (e.g., by invasive non-native plant species and encroachment of woody vegetation), with subsequent impacts on bird populations (Samson and Knopf 1994). Encroachment of woody vegetation into grasslands has negative effects on occurrence, abundance and nesting success of grassland birds in the UMRB (Samson and Knopf 1994; Grant et al. 2004; Greer et al. 2016), and often has a negative impact on soil C stocks (Jackson et al. 2002). Similarly, exotic grasses and other invasive plants in grasslands also tend to negatively impact bird populations across the Northern Prairie region of North America (Bakker and Higgins 2009; Greer et al. 2016). In addition, a number of grassland bird species are area-sensitive, showing negative population responses as grassland patch size decreases (Davis 2004). This area sensitivity is not always consistent among species or studies (Walk et al. 2010; Greer et al. 2016), and such factors as edge-to-interior ratio, vegetation characteristics, and landscape-scale habitat characteristics may modify area sensitivity for grassland birds in the UMRB (Bakker et al. 2002; Davis 2004; Ribic et al. 2009). At the local patch scale, bare ground, vegetation height, and litter depth are consistent predictors of habitat occupancy by grassland birds (and are also relevant for the regional C cycle and hydrology), although relationships with these variables and occupancy, abundance or nesting success may differ among different grassland bird species (Fisher and Davis 2010).

CRP grasslands generally provide favorable habitat for grassland birds, although vegetation structure (e.g., high grass coverage vs. low grass coverage vs. bare patches) and plant species composition, year-to-year variation in precipitation, and landowner management (e.g., haying), in addition to landscape-level characteristics, influence suitability for various grassland bird species in the UMRB (Johnson and Schwartz 1993). It should also be noted that CRP grasslands do not replace native prairie with regard to either the vegetative or the bird communities; this is especially relevant to species of conservation concern, such as Sprague's pipit (*Anthus spragueii*) and Baird's sparrow (*Ammodramus bairdii*; Johnson and Schwartz 1993).

Switchgrass (*Panicum virgatum*) or other bioenergy grasslands as cellulosic biofuel crops could also serve as potential suitable breeding habitat for grassland birds (Murray et al. 2003; Robertson et al. 2012b; Blank et al. 2014, 2015), although appropriate timing of harvest (i.e., after the breeding season is complete) is critical to grassland bird productivity in these habitats. Abundances of many grassland birds are higher in switchgrass fields than in row crops, but bird species showing positive relationships with taller grassland vegetation are those for which switchgrass is likely to be suitable habitat (Murray and Best 2003; Roth et al. 2005). Late-summer harvest, however, can make switchgrass fields more suitable for species favoring short-grass habitats, such as grasshopper sparrow (*Ammodramus savannarum*) and horned lark (*Eremophila alpestris*) (Murray and Best 2003; Roth et al. 2005). Nevertheless, breeding bird biodiversity in switchgrass is also not likely to reach levels supported by native prairies, which have more varied vegetation and structural diversity, so conversion of native prairie to switchgrass or other bioenergy grasslands is likely to negatively impact grassland birds as a whole (Robertson et al. 2012b; Blank et al. 2014). In addition to breeding-season benefits to grassland birds, switchgrass fields are also used as *en route* migration stopover habitat for migrating grassland birds (Robertson et al. 2012a), and abundance and species richness for migrant grassland birds in switchgrass fields did not differ significantly from those in grasslands with a composition of mixed grasses and forbs.

BECCS scenarios are likely to be coupled to development of renewable energy sources such as wind, solar radiation, and hydropower (Figure 2). Wind energy development is likely to increase in the future in the UMRB due to high and consistent winds (Fargione et al. 2012). Such development of wind energy potential within the UMRB is likely to influence regional bird populations (Kuvlesky et al. 2007; Smith and Dwyer 2016), and several studies have examined effects of wind farms on the regional avifauna. Direct mortality of birds in the Northern Prairie region from collisions with turbines appears to be relatively low. For example, (Osborn et al. 2000; Johnson et al. 2003) estimated bird mortalities at the Buffalo Ridge Wind Resource Area (BRWRA) in southwestern Minnesota to range from 0.5-4.5 mortalities per turbine per year, with the majority of birds killed belonging to the Passeriformes. Graff et al. (2016) studied wind farms in southern North Dakota and northern South Dakota and estimated mortalities during the spring and early summer to range from 0.8-2.6 mortalities per MW of energy produced, with waterfowl deaths constituting a majority of mortalities and a higher diversity of birds being killed at turbines located in grasslands than at agricultural sites. Perhaps more problematic to bird populations than direct mortalities are reduced abundances in habitats surrounding wind turbines (often up to 800 m), resulting in lower occupancy or lower bird abundances in wind farm areas (Drewitt and

Langston 2006; Stewart et al. 2007; Pearce-Higgins et al. 2009). Such reduced abundances in wind farm habitats, however, do not necessarily occur for all species (Douglas et al. 2011). Within the Northern Prairie region, Usgaard et al. (1997) found that raptor abundances within the BRWRA were similar to other habitats within the region, but that raptor nest sites avoided areas where turbines were present. Densities of grassland birds within CRP grasslands in the BRWRA were about 3-fold lower at 80 m than at 180 m from turbines. Niemuth et al. (2013) found that occupancy of wetland sites by water birds and shorebirds did not differ markedly between wind farm and non-wind farm sites in southern North Dakota and northern South Dakota, although occupancy was slightly but consistently lower for a few species at sites near turbines where agriculture was the dominant habitat on the landscape. Collectively, these data suggest that site location of wind farms within the UMRB is likely to influence their impact on birds. Placement of wind farms in agricultural or other disturbed habitats while avoiding undisturbed grassland areas is likely to provide maximum benefits to grassland bird biodiversity (Kiesecker et al. 2011; Graff et al. 2016). In this regard, Fargione et al. (2012) modeled bird habitat and bird abundances within the Northern Great Plains to identify sites within the UMRB with high wind potential but relatively low potential for impacting bird populations.

Wetlands in the UMRB, particularly within the Prairie Pothole Region (PPR) of the Dakotas, are critical habitats for wetland-associated birds (Lehtinen et al. 1999; Naugle et al. 2001; Johnson et al. 2005; Mushet et al. 2014; Steen et al. 2016). Land use change has markedly impacted wetland habitats and future climate and land use changes are projected to continue to negatively impact wetlands within the region and their functionality, including impacts on such ecosystem services as water quality, carbon sequestration and biodiversity (Whited et al. 2000; Johnson et al. 2010; Fennessy and Craft 2011; Rashford et al. 2015). Current pressures to alter wetlands for row-crop production within the PPR have resulted in recent average wetland loss rates of 0.28 - 0.35% per year (as well as across the UMRB, Figure 4), with greater losses in central and eastern regions and lesser losses in western and northern edges of the Dakotas (Johnston 2013). Coupled with loss of wetlands due to agricultural expansion in the PPR, agricultural acres with tile drainage have also recently expanded recently, and this trend is likely to continue into the future. Expansion of tile drainage in agricultural areas alters wetland hydrology, reduces surface water storage, increases nutrient turnover rates, increases effective drainage areas and increases flows of surface water into stream and wetland systems (Blann et al. 2009). Thus, increasing tile drainage is likely to compound wetland losses due to agricultural practices, shifting available wetland area away from ephemeral and seasonal wetlands to semi-permanent and permanent wetlands and increasing agricultural contaminant levels (Blann et al. 2009). Moreover, fluctuation of water levels in wetlands within tilled agricultural lands may be 3-fold greater than in those in grasslands within the PPR, with lesser fluctuation in more permanent wetlands (Euliss and Mushet 1996), and the increased surface water flows in areas with tile drainage is likely to compound these fluctuations. Thus, land use changes within the PPR are likely to markedly impact the suitability of wetlands for wetland-associated birds.

Habitat suitability models for wetland-associated birds suggest that unfragmented prairie-wetland complexes provide more and better habitat than isolated wetlands within row-crop agricultural habitats in the PPR (Naugle et al. 2001). Johnson et al. (2005) developed climate-change models for semi-permanent wetlands in the PPR, projecting regional reductions in the amount of productive wetland habitat for waterfowl and a shift of the most productive habitat to available wetlands in the eastern and northern regions of the PPR. Expanding climate-change models to include surface water, groundwater, and wetland vegetation dynamics suggested a substantial shrinkage and eastward shift of productive wetland habitat for waterfowl (Johnson et al. 2010). More recent bioclimatic models also project loss of suitable wetland habitat for wetland birds within the PPR (Steen et al. 2016). Rashford et al. (2015) modeled climate and land use change within the PPR and their models suggested that the combined pressures of current land use and climate change trends would reduce wetland productivity and suitable habitat for wetland-associated species.

To project trends in biodiversity under future region-wide land use predictions, future studies using spatially explicit predictive models to link abundances and distributions of grassland and wetland bird species to changes in land cover and landscape configuration across the region are needed. Such studies should focus on spatially-explicit land cover change scenarios (Sohl et al. 2014) using recent remotely-sensed land cover data, derived from sources such as classified Landsat imagery (e.g., USGS National Land Cover Database or LANDFIRE). These studies will provide much better region-wide projections for biodiversity responses to landscape change, including landscape change associated with alternative BECCS scenarios within the UMRB. Models developed for the UMRB may be suitable for application or extrapolation to other regions with similar agriculturally dominated landscapes and social systems.

References

Bakker KK, Higgins KF (2009) Planted grasslands and native sod prairie: Equivalent habitat for grassland birds? West N Am Nat 69:235–242.

Bakker KK, Naugle DE, Higgins KF (2002) Incorporating landscape attributes into models for migratory grassland bird conservation. Conserv Biol 16:1638–1646.

Blank PJ, Sample DW, Williams CL, Turner MG (2014) Bird communities and biomass yields in potential bioenergy grasslands. PLoS One 9:e109989.

Blank PJ, Williams CL, Sample DW, et al (2015) Alternative scenarios of bioenergy crop production in an agricultural landscape and implications for bird communities. Ecol Appl 150511140501007.

Blann KL, Anderson JL, Sands GR, Vondracek B (2009) Effects of agricultural drainage on aquatic ecosystems: A review. Crit Rev Environ Sci Technol 39:909–1001.

Davis SK (2004) Area sensitivity in grassland passerines: Effects of patch size, patch shape, and vegetation structure on bird abundance and occurrence in southern Saskatchewan. Auk 121:1130.

Dixon MD, Johnson WC, Scott ML, et al (2012) Dynamics of plains cottonwood (*Populus deltoides*) forests and historical landscape change along unchannelized segments of the Missouri River, USA. Environ Manage 49:990–1008.

Douglas DJT, Bellamy PE, Pearce-Higgins JW (2011) Changes in the abundance and distribution of upland breeding birds at an operational wind farm. Bird Study 58:37–43.

Drewitt AL, Langston RHW (2006) Assessing the impacts of wind farms on birds. Ibis 148:29–42.

Euliss NH, Mushet DM (1996) Water-level fluctuation in wetlands as a function of landscape condition in the prairie pothole region. Wetlands 16:587–593.

Fargione J, Kiesecker J, Slaats MJ, Olimb S (2012) Wind and wildlife in the Northern Great Plains: identifying low-impact areas for wind development. PLoS One 7:e41468.

Fennessy S, Craft C (2011) Agricultural conservation practices increase wetland ecosystem services in the Glaciated Interior Plains. Ecol Appl 21:S49–S64.

Fisher RJ, Davis SK (2010) From Wiens to Robel: A review of grassland-bird habitat selection. J Wildl Manage 74:265–273.

Gorzo JM, Pidgeon AM, Thogmartin WE, et al (2016) Using the North American Breeding Bird Survey to assess broad-scale response of the continent's most imperiled avian community, grassland birds, to weather variability. Condor 118:502–512.

Graff BJ, Jenks JA, Stafford JD, et al (2016) Assessing spring direct mortality to avifauna from wind energy facilities in the Dakotas. J Wildl Manage 80:736–745.

Grant TA, Madden E, Berkey GB (2004) Tree and shrub invasion in northern mixed-grass prairie: implications for breeding grassland birds. Wildl Soc Bull 32:807–818.

Greer MJ, Bakker KK, Dieter CD (2016) Grassland bird response to recent loss and degradation of native prairie in central and western South Dakota. Wilson J Ornithol 128:278–289.

Hill JM, Franklin Egan J, Stauffer GE, Diefenbach DR (2014) Habitat availability is a more plausible explanation than insecticide acute toxicity for U.S. grassland bird species declines. PLoS One 9:e98064.

Jackson RB, Banner JL, Jobbágy EG, et al (2002) Ecosystem carbon loss with woody plant invasion of grasslands. Nature 418:623–626.

Johnson DH, Schwartz MD (1993) The conservation reserve program and grassland birds. Conserv Biol 7:934–937.

Johnson GD, Erickson WP, Dale Strickland M, et al (2003) Mortality of bats at a large-scale wind power development at Buffalo Ridge, Minnesota. Am Midl Nat 150:332–342.

Johnson WC, Carter Johnson W, Millett BV, et al (2005) Vulnerability of northern prairie wetlands to climate change. Bioscience 55:863.

Johnson WC, Carter Johnson W, Werner B, et al (2010) Prairie wetland complexes as landscape functional units in a changing climate. Bioscience 60:128–140.

Kiesecker JM, Evans JS, Fargione J, et al (2011) Win-win for wind and wildlife: a vision to facilitate sustainable development. PLoS One 6:e17566.

Kuvlesky WP, Brennan LA, Morrison ML, et al (2007) Wind energy development and wildlife conservation: Challenges and opportunities. J Wildl Manage 71:2487–2498.

Lehtinen RM, Galatowitsch SM, Tester JR (1999) Consequences of habitat loss and fragmentation for wetland amphibian assemblages. Wetlands 19:1–12.

Mineau P, Whiteside M (2013) Pesticide acute toxicity is a better correlate of U.S. grassland bird declines than agricultural ontensification. PLoS One 8:e57457.

Munes EC, Dixon MD, Swanson DL, et al (2015) Large, infrequent disturbance on a regulated river: response of floodplain forest birds to the 2011 Missouri River flood. Ecosphere 6:art212.

Murray LD, Best LB (2003) Short-term bird response to harvesting switchgrass for biomass in Iowa. J Wildl Manage 67:611.

Murray LD, Best LB, Jacobsen TJ, Braster ML (2003) Potential effects on grassland birds of converting marginal cropland to switchgrass biomass production. Biomass Bioenergy 25:167–175.

Mushet DM, Neau JL, Euliss NH (2014) Modeling effects of conservation grassland losses on amphibian habitat. Biol Conserv 174:93–100.

Naugle DE, Johnson RR, Estey ME, Higgins KF (2001) A landscape approach to conserving wetland bird habitat in the prairie pothole region of eastern South Dakota. Wetlands 21:1–17.

Niemuth ND, Walker JA, Gleason JS, et al (2013) Influence of wind turbines on presence of Willet, Marbled Godwit, Wilson's Phalarope and Black Tern on wetlands in the Prairie Pothole Region of North Dakota and South Dakota. Waterbirds 36:263–276.

Osborn RG, Higgins KF, Usgaard RE, et al (2000) Bird mortality associated with wind turbines at the Buffalo Ridge Wind Resource Area, Minnesota. Am Midl Nat 143:41–52.

Pearce-Higgins JW, Stephen L, Langston RHW, et al (2009) The distribution of breeding birds around upland wind farms. J Appl Ecol. doi: 10.1111/j.1365-2664.2009.01715.x

Rashford BS, Adams RM, Wu J, et al (2015) Impacts of climate change on land-use and wetland productivity in the Prairie Pothole Region of North America. Regional Environ Change 16:515–526.

Ribic CA, Koford RR, Herkert JR, et al (2009) Area sensitivity in North American grassland birds: Patterns and processes. Auk 126:233–244.

Robertson BA, Landis DA, Scott Sillett T, et al (2012a) Perennial agroenergy feedstocks as en route habitat for spring migratory birds. Bioenergy Res 6:311–320.

Robertson BA, Rice RA, Scott Sillett T, et al (2012b) Are agrofuels a conservation threat or opportunity for grassland birds in the United States? Condor 114:679–688.

Roth AM, Sample DW, Ribic CA, et al (2005) Grassland bird response to harvesting switchgrass as a biomass energy crop. Biomass Bioenergy 28:490–498.

Samson F, Knopf F (1994) Prairie Conservation in North America. Bioscience 44:418–421.

Sauer JR, Link WA, Fallon JE, et al (2013) The North American Breeding Bird Survey 1966–2011: Summary Analysis and Species Accounts. North American Fauna 79: 1-32.

Schipper AM, Belmaker J, de Miranda MD, et al (2016) Contrasting changes in the abundance and diversity of North American bird assemblages from 1971 to 2010. Glob Chang Biol 22:3948–3959.

Smith JA, Dwyer JF (2016) Avian interactions with renewable energy infrastructure: An update. Condor 118:411–423.

Sohl TL (2014) The relative impacts of climate and land-use change on conterminous United States bird species from 2001 to 2075. PLoS One 9:e112251.

Sohl TL, Sayler KL, Bouchard MA, et al (2014) Spatially explicit modeling of 1992-2100 land cover and forest stand age for the conterminous United States. Ecol Appl 24:1015–1036.

Steen VA, Skagen SK, Melcher CP (2016) Implications of climate change for wetland-dependent birds in the Prairie Pothole Region. Wetlands 36:445–459.

Stewart GB, Pullin AS, Coles CF (2007) Poor evidence-base for assessment of windfarm impacts on birds. Environ Conserv 34:1.

Walk JW, Kershner EL, Benson TJ, Warner RE (2010) Nesting success of grassland birds in small patches in an agricultural landscape. Auk 127:328–334.

Whited D, Galatowitsch S, Tester JR, et al (2000) The importance of local and regional factors in predicting effective conservation. Landsc Urban Plan 49:49–65.

Wright CK, Wimberly MC (2013) Recent land use change in the Western Corn Belt threatens grasslands and wetlands. Proc Natl Acad Sci U S A 110:4134–4139.