Land management and climate change determine second-generation bioenergy potential of the US Northern Great Plains

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Abstract
Bioenergy with carbon capture and storage (BECCS) has been proposed as a potential climate mitigation strategy raising concerns over trade-offs with existing ecosystem services. We evaluate the feasibility of BECCS in the Upper Missouri River Basin (UMRB), a landscape with diverse land use, ownership, and bioenergy potential. We develop land-use change scenarios and a switchgrass (Panicum virgatum L.) crop functional type to use in a land-surface model to simulate second-generation bioenergy production. By the end of this century, average annual switchgrass production over the UMRB ranges from 60 to 210 Tg dry mass/year and is dependent on the Representative Concentration Pathway for greenhouse gas emissions and on land-use change assumptions. Under our simple phase-in assumptions this results in a cumulative total production of 2,000–6,000 Tg C over the study period with the upper estimates only possible in the absence of climate change. Switchgrass yields decreased as average CO2 concentrations and temperatures increased, suggesting the effect of elevated atmospheric CO2 was small because of its C4 photosynthetic pathway. By the end of the 21st century, the potential energy stored annually in harvested switchgrass averaged between 1 and 4 EJ/year assuming perfect conversion efficiency, or an annual electrical generation capacity of 7,000–28,000 MW assuming current bioenergy efficiency rates. Trade-offs between bioenergy and ecosystem services were identified, including cumulative direct losses of 1,000–2,600 Tg C stored in natural ecosystems from land-use change by 2090. Total cumulative losses of ecosystem carbon stocks were higher than the potential ~300 Tg C in fossil fuel emissions from the single largest power plant in the region over the same time period, and equivalent to potential carbon removal from the atmosphere from using biofuels grown in the same region. Numerous trade-offs from BECCS expansion in the UMRB must be balanced against the potential benefits of a carbon-negative energy system.

KEYWORDS
bioenergy with carbon capture and storage (BECCS), climate change, land use and land-cover change, negative emission technologies, switchgrass
1 | INTRODUCTION

Negative emission technologies and other carbon dioxide removal (CDR) techniques, including bioenergy with carbon capture and storage (BECCS), have been proposed as approaches to achieve the goals set in the 2015 Paris Agreement to limit global warming to below 1.5 or 2.0°C (Fuss et al., 2018; Minx et al., 2018). Almost all greenhouse gas emission scenarios considered by the Representative Concentration Pathways (RCP) that aim to limit global warming to 1.5°C rely heavily on CDR approaches. BECCS is a widely promoted CDR technique designed to produce energy from bioenergy crops while capturing the resulting CO2 emissions for long-term storage. The RCP scenarios that rely on BECCS assume a combination of large atmospheric CO2 removal, 100–1,000 Gt C, over the 21st century (IPCC, 2019) while providing 150–400 EJ/year in bioenergy (Bauer et al., 2018; van Vuuren et al., 2011). At regional scales, these are significant changes in the energy sector and carbon cycle, for example in the United States, CO2 removal from BECCS to meet a 2°C target has been estimated to be as much as 1 Gt CO2/year in 2050, and between 1 and 3 Gt CO2/year in 2100 between different integrated assessment models (IAMs; Baik et al., 2018; Peters et al., 2017).

Climate mitigation, however, may lead to unintended consequences for non-carbon processes, that is, water and nutrient cycles, regional economics, and ecological diversity, because CDR techniques like BECCS displace other land use and ecosystem services (Searchinger et al., 2015; Stoy et al., 2018). Consequently, an integrated approach for carbon management is required to avoid disruption in other social or natural systems while achieving ambitious climate mitigation goals (Harper et al., 2018; Jackson et al., 2005; Luyssaert et al., 2018). Global IAMs build macro-socioeconomic scenarios consistent with climate targets using regional-to-global scale assumptions regarding land-use allocation and climate mitigation, while considering the impact of gross domestic product on caloric and energy demands of future populations (Popp et al., 2017). Yet the realities of deploying such scenarios at national to sub-national scales requires a detailed assessment of trade-offs and benefits of carbon removal with other ecological and socioeconomic services (Bengtsson et al., 2019; Foley, 2005; Robertson et al., 2017; Stoy et al., 2018).

The use of “marginal lands,” lands not currently used for food production due to low productivity or other environmental constraints, has been of particular interest for BECCS development to avoid food-biofuel conflict, and potentially for providing other ecological benefits like soil conservation that come with the planting of perennial cellulosic crops (Cai, Zhang, & Wang, 2011; Campbell, Lobell, Genova, & Field, 2008; Robertson et al., 2017). Current “second-generation” bioenergy crops grow on so-called marginal lands where food is not produced so as not to conflict with food production like earlier “first-generation” bioenergy crop scenarios. Third-generation bioenergy stocks include algae, for example, but are not included in this study. The utilization of marginal lands for the implementation of CDR techniques is among the most promising pathways for mitigating carbon emissions while minimizing trade-offs with food security (Boysen, Lucht, & Gerten, 2017).

Here, we assess second-generation bioenergy potential under various land management and climate scenarios using a dynamic global vegetation model. We use the Upper Missouri River Basin (UMRB) of the United States as a regional case study as part of a larger project assessing the opportunities and trade-offs among BECCS and food, water, energy, biodiversity, and social systems in the area (Stoy et al., 2018). In a geospatial analysis of near-term potential for carbon-negative bioenergy opportunities in the United States, Baik et al. (2018) identified large areas of the UMRB as a national hot spot for rapid and prioritized deployment of BECCS due to high storage capacity and good access to existing CO2 pipelines and other infrastructure. We explore the consequences for the wide-scale deployment of a native high-yielding bioenergy grass, switchgrass (P. virgatum L.), on the coupled carbon and water cycles in the UMRB using the LPJmL model (Bondeau et al., 2007; Sitch et al., 2003). We evaluate projected climate and atmospheric CO2 effects on switchgrass growth on marginal lands after considering ownership and conservation status and discuss the implications of regional-scale bioenergy deployment on carbon and water budgets.

2 | METHODS

2.1 | Study region

The UMRB as defined here includes all land areas flowing into the Missouri River above the confluence of the Big Sioux and Missouri Rivers in Sioux City, Iowa, and excluding the Niobrara watershed (Stoy et al., 2018). Major portions of Montana, South Dakota, northern Wyoming, and southern North Dakota are part of this watershed along with much small areas of Nebraska, Iowa and Minnesota and the southern borders of Alberta, Saskatchewan, Canada (Figure 1). For the purpose of this study we have excluded land in Canada in summary statistics, with a resulting study domain covering ~800,000 km2. Population density in the UMRB is relatively low with the majority of counties below four individuals per km2. The largest city, Sioux Falls, South Dakota, has a total population that is less than 200,000 individuals, equating to ~10% of the entire UMRB population. There are strong gradients in topography, climate, land cover and land use. The western portion of the watershed is dominated by federal
ownership of forest lands along the Rocky Mountains, transitioning into private dryland agriculture and rangelands as one moves down the watershed, with a mixture of lands managed by tribal, state and federal agencies. Soil type is highly variable and more fertile soils dominate the glaciated portions of the UMRB largely east of the Missouri River that are dominated by row-crop agriculture. The climate of non-mountainous portions of the UMRB transitions from cold semiarid (Köppen climate: BSk) in the west to humid continental (Dfa) in the east and precipitation roughly triples from west to east, ~300 mm/year to nearly ~1 m/year (Millett, Carter Johnson, & Guntenspergen, 2009). Crops and pasture in the UMRB represent some 30% of total US wheat production, 13% soybean production, 11% of cattle production, and 9% of maize production, which is highly concentrated in the eastern region of the watershed in North and South Dakota. More than 20 Tribal Nations comprise ~10% (86,000 km²) of the UMRB. Land registered in the Conservation Reserve Program, a Federal program to restore lands formerly in agriculture to their natural state, peaked in the early 2000s, with enrollment since declining in response to reduced Federal enrollment caps, expiring lands, and changes in economic incentives with a marked intensification of agriculture in areas previously enrolled (Morefield, LeDuc, Clark, & Iovanna, 2016; Schmer, Vogel, Mitchell, & Perrin, 2008; Stoy et al., 2018). Irrigation is uncommon in the study area, for example only 1% of the total cultivated land in North Dakota is currently irrigated and is largely restricted to areas around rivers and unique geological formations.

2.2 | Bioenergy feedstock

This study focuses on switchgrass, a warm-season native C4 perennial grass that is adapted to a wide range of environmental conditions. Switchgrass has been cultivated in the US Great Plains since the 1940s and has been studied as a potential biomass energy crop in the United States since the mid-1980s (Casler et al., 2007; Downing et al., 2011). Despite having lower yields than *Miscanthus × giganteus*, switchgrass is a widely dispersed and fast-growing native plant to most of North America (Schmer et al., 2008). Much work has been done to cultivate varieties of switchgrass that are tolerant to extreme conditions including cold, drought and/or extended inundation (McLaughlin & Kszos, 2005; Sanderson et al., 1996). Switchgrass has relatively few major insects or pathogens to limit its productivity (Vogel, Sarath, Saatho, & Mitchell, 2011) and can be managed while minimizing nitrogen fertilizer usage (Valdez, Hockaday, Masiello, Gallagher, & Philip Robertson, 2017; Vogel, Brejda, Walters, & Buxton, 2002). It can be converted into energy through direct combustion, liquid fuel production, that is, cellulosic ethanol.
and butanol, and even manufactured into synthetic gasses. Conventional haying equipment can be used for the harvest, transport, and storage of switchgrass, though continued development of infrastructure to increase efficiency of transportation and losses from long-term storage are needed. For the reasons described above and for simplifying purposes we chose to only look at one generalized second-generation crop across our region, but recognize other herbaceous species are being explored for bioenergy production, e.g., *Camelina sativa* (Sanderson & Adler, 2008). Further our study region does not include many forested lands outside of the Black Hills of South Dakota and the Rocky Mountains where climate constrains the rapid woody growth favored for bioenergy development. Many riparian areas where woody bioenergy may be possible, for example, willows (*Salix* spp.), have already been converted to agriculture or is protected. These areas likewise comprise a small fraction of our study region.

### 2.3 Model and switchgrass parametrization

Rather than model switchgrass yields by substituting generic C4 grass parameters (Harper et al., 2018; Krause et al., 2017), we specifically developed a model of switchgrass production embedded within LPJmL. LPJmL is a process-based Dynamic Global Vegetation Model that simulates global terrestrial biogeography and biogeochemistry via coupled water-carbon cycling and vegetation dynamics (Sitch et al., 2003). The model represents natural terrestrial vegetation as Plant Functional Types whose distributions are prognostically determined by competition for resources, physiological processes and bioclimatic limits (Box, 1996; Poulter et al., 2011). The model also represents a category for managed lands, which include more than 11 crop functional types (CFTs), as well as a separate managed grasslands category, with various management practices such as irrigation, treatment of harvest residue, and intercropping practices (Bondeau et al., 2007). Together, the net carbon fluxes simulated by LPJmL are used to estimate net biome productivity (NBP), a measure of the balance of inputs and outputs of carbon fluxes. Here we defined NBP as net primary productivity (NPP) plus carbon flux of new vegetation establishment and sowing minus heterotrophic respiration (rH) as well as the carbon flux from crop harvest, land-use change and fire.

To develop the switchgrass CFT, we began with the default parameters from the existing corn (maize) CFT, which also has a C4 photosynthetic pathway, from Bondeau et al. (2007), then updated the parameterization using LPJmL4, “Comprehensive Updates and Biofuels” (Schaphoff, Bloh, et al., 2018; Schaphoff, Forkel, et al., 2018) and ORCHIDEE-MICT-BIOENERGY (Li et al., 2018; see TableS1). Additionally, unlike previous representations of switchgrass that treat it as an annual crop, we modify switchgrass to have perennial attributes. These include biannual root turnover, where we specify that belowground carbon allocation investments are greater in the first year, within a dynamic range between 20% and 40% depending on soil moisture, and then fixed in the following years at 5% of gross primary production. Harvest residue was assumed to remain on site. We then evaluated simulated annual yields averaged over the period 1980–2003 to field measurements at 26 sites from Arundale et al. (2013) and the Biofuel Ecophysiological Traits and Yields Database, BETYdb (LeBauer et al., 2017; FigureS1).

#### 2.4 Driver data and simulation setup

Regional simulations of LPJmL were made at 0.5 degree spatial resolution using data on soil texture (fraction of sand, silt, clay) from the Harmonized World Soil Database (HWSD v1.2; Köchy, Hiederer, & Freibauer, 2014), and air temperature, precipitation, and cloud cover from the HadGEM2-AO general circulation model (GCM). The HadGEM2-AO is representative of a moderate CO2-climate sensitivity scenario for the region (Stoy et al., 2018). Climate projections and atmospheric CO2 concentrations were used for the RCP 2.6 and 8.5 to bound a lower and upper limit for uncertainty, with the GCM data downscaled and bias-corrected to match the spatial resolution (0.5°) and climatological mean for 1960–1990 of the Climatic Research Unit Time Series (CRU TS-Version 3.2.2; Harris, Jones, Osborn, & Lister, 2014; Riahi et al., 2011; Taylor, Stouffer, & Meehl, 2012; Zhang et al., 2017). The number of wet days required to distribute monthly precipitation inputs was derived using the 20th century climatology CRU dataset, as this meteorological variable was not available as part of the HadGEM2-AO outputs (Harris et al., 2014; New, Hulme, & Jones, 2000; Taylor et al., 2012).

Vegetation and soil carbon stocks were first simulated to be in dynamic equilibrium with climate (1901–1930) and a preindustrial atmospheric CO2 of 287 ppm by running the model for a 1,000 year “spin up” with potential natural vegetation, omitting land use and crops. Following the spinup, a historical simulation was conducted for the period 1860–2005 using time-varying meteorology, atmospheric CO2, and prescribed land-use change. Historic inputs of rainfed and irrigated agriculture lands data (1901–2003) were the same as used in Bondeau et al. (2007), where Ramankutty and Foley (1999) provided historical cropland fractions and with Leff, Ramankutty, and Foley (2004) providing the distribution of CFTs. The livestock grazing grid-cell fraction was informed by the class “grass and fodder” of the HYDE dataset (Klein Goldewijk & Battjes, 1997). Following the same procedures as Bondeau et al. (2007), rangeland/
pasture growth was modeled the same as natural grasslands with the exception that a portion of the grass (50%) is “removed” when leaf area index reaches a certain threshold, of which 45% is emitted to the atmosphere and 5% returns to the litter pool. Note that no distinction is made between whether the removed material is harvested and or grazed in Bondeau et al. (2007) and in this analysis. This is relevant as it could affect the soil carbon impacts of switchgrass in the “Aggressive Pasture” scenario (see below). Finally, the irrigated agricultural fraction was determined by the Global Map of Irrigated Area in 2000 (GMIA 4; Döll & Siebert, 1999). Managed lands were left constant at 2003 levels for the remainder of the historic run up to 2005.

A set of factorial simulations were made to evaluate the individual and combined effects of climate, CO₂ and land-use change and their uncertainties on switchgrass production and ecosystem services. These included evaluating the drivers from three land-use change scenarios (see next section), the two RCP scenarios, and their interactions. For the climate and CO₂ scenarios, we applied a time series of annual atmospheric CO₂ concentrations from the CMIP5 database, with CO₂ in RCP 2.6 increasing to a maximum of 443 ppm in 2050, whereas atmospheric CO₂ continuously increases to more than 900 ppm by the end of the century for RCP 8.5. After 2005, the number of wet days per month were randomly selected from a random sample of a 30 year period between 1970 and 1999 due to their unavailability in CMIP5. In addition to the time-varying climate and CO₂ scenarios, a scenario with constant, stationary climate trends was made by randomly recycling annual meteorological data from the years 1970–1999 and keeping atmospheric CO₂ levels fixed at 2005 concentrations of 379 ppm through the entire simulation ending in 2099.

### 2.5 Land management and switchgrass scenario development

We developed a series of land-use change scenarios for the UMRB to assess potential production of switchgrass and the ecological impacts of land conversion on carbon and water fluxes under different climate and CO₂ scenarios. A set of simple assumptions were made to develop annually evolving bioenergy expansion using a two-step process described below. We explored the use of pre-existing datasets for both marginal (Cai et al., 2011) and abandoned crop and pasture lands (Campbell et al., 2008; Zumkehr & Campbell, 2013). However, issues with the coarseness of the global datasets, such as large areas in the middle of our study domain being excluded from the marginal lands layer (Cai et al., 2011), turned us to the more recent and higher resolution Zumkehr and Campbell (2013) dataset. Despite higher resolution, this dataset only considered abandoned lands as of the year 2000 (~20,000 km² or 2.5% of the UMRB) and did not include marginal or abandoned pasture lands. After assessment of these existing datasets we decided to move forward with a new dataset for “marginal” lands to encompass our framework of moderate to “aggressive” switchgrass expansion scenarios that required knowing the marginal lands that were abandoned, but also in rangeland or pasture land-use categories (Table 1).

#### 2.6 Step 1: Determining potential lands for switchgrass conversion

To create simple and transparent rules for determining potential marginal lands for switchgrass conversion, we began

| TABLE 1 Description of lands available for switchgrass expansion in the Upper Missouri River Basin for four land-cover and land-use change scenarios |
|-------------------------------------------------|-----------------|-----------------|-----------------|
| Crop lands (c. 2003 Bondeau et al., 2007)        | No              | No              | No              |
| Federal State and Private lands (PADUS 1.4 2016)  | No              | No              | No              |
| Areas classified as forested, wetland, and or urban between 2000 and 2010 (Broxton et al., 2014) | No              | No              | No              |
| Pasture lands (c. 2003 Bondeau et al., 2007)      | No              | No              | Yes             |
| Areas designated as Priority for Ecological Conservation (TNC) | No              | No              | Yes             |
| Native American owned Lands not previously excluded | No              | Yes             | Yes             |
| Remaining lands not excluded by rules above       | No              | Yes             | Yes             |
with the global 0.5 km MODIS-based land-cover climatology (Broxton, Zeng, Sulla-Menashe, & Troch, 2014). The land-cover dataset represents the period 2001–2010 using the International Geosphere Biosphere Program (IGBP) land-cover classification scheme. We reclassified this dataset into two classes: land potentially available for switchgrass conversion adjusted for existing cropland in a second step), and land not available for switchgrass conversion. Lands available for switchgrass conversion, according to IGBP categories, were considered to be (9) savanna (10) grasslands (12) croplands (14) cropland/natural vegetation, and (16) barren. Lands not available for switchgrass conversion were: (1–8) evergreen needleleaf, evergreen broadleaf, deciduous needle-leaf, deciduous broadleaf, mixed forests, closed shrublands, open shrublands, woody savannas (11) permanent wetlands (13) urban and built up areas, and (15) snow/ice. We then summed the available lands for switchgrass conversion to potential fraction available at 1 km to match the resolution for the next processing step.

The available lands for switchgrass were further modified restricting by land ownership and removing existing cropland. We assumed that lands are unavailable for expansion if in Federal, State, or other public (i.e., county) ownership. Additionally, public lands with current conservation status were also considered unavailable to switchgrass conversion. We used a 1 km gridded version of the Protected Areas Database of the United States (U.S. Geological Survey, Gap Analysis Program [GAP], 2016) to subtract from the available lands dataset. Further, to ensure that switchgrass expansion did not compete with food crops, we assumed that switchgrass expansion can only occur on non-cropland areas. To achieve this assumption, we aggregated available lands for potential switchgrass conversion to 0.5 degree resolution and subtracted fractional area defined as cropland from the LPJmL-driver dataset of Bondeau et al. (2007) providing consistency between the land-cover driver dataset used for LPJmL and the switchgrass expansion scenario. This approach results in a 0.5 degree resolution maximum potential planting area dataset for second-generation bioenergy crops. This dataset was used for LPJmL simulations implementing switchgrass conversion for bioenergy (Table 1).

### 2.7 | Step 2: Bioenergy scenario development

We designed a range of time-varying switchgrass expansion scenarios beginning in the year 2020 assuming a switchgrass area of zero, and then using the total potential switchgrass area as a constraint to conversion (Table 1; Figure 1). Total cropland area and the respective CFT proportions were held constant after 2005 for all land management scenarios, whereas pasture/rangelands were held constant for all scenarios except in the Most Aggressive scenario (scenario AP). In our Aggressive Scenario (A) we subtracted pasture/rangelands c. 2003 defined in Bondeau et al. (2007) from the potential available lands dataset. An additional, Conservation Scenario (C), protected potential switchgrass areas by masking areas identified by The Nature Conservancy (TNC) as Priority Conservation Areas (CEC & TNC, 2005). A null case, Constant Baseline Scenario (B), was created which excluded conversion to switchgrass bioenergy crops and maintained constant current managed lands c. 2003 defined in Bondeau et al. (2007).

The temporal dynamics of the switchgrass expansion scenarios made the simple assumption that, from 2020 to 2100, the entire grid cell could be converted to its potential land area for switchgrass production. Thus, if the entire grid cell was 100% available for switchgrass, the conversion rate would be 100% divided by 81 years, or a conversion rate of 1.23% per year. This constant conversion rate was applied to all grid cells, and so if a grid cell had 50% available for switchgrass expansion, at a rate of 1.23% conversion per year the expansion would end in 40 years. The wide range of potential switchgrass land availability across the UMRB results in a nonlinear conversion at regional scales (Figure 1). In scenarios that allow pasture/rangeland conversions, pasture/ rangeland land is converted first followed by other available lands.

The four land-cover change scenarios (B, C, A, AP) in combination with the three unique climate scenarios (Recycled, RCP 2.6, RCP 8.5) combined to create 12 unique model simulations for which we could assess potential energy production as well as ecological impact of switchgrass production in the region. Specifically, we looked at changes in NBP, and ecosystem carbon stocks and water fluxes (evaporation, transpiration and runoff) over time. We summarized results over snapshots of time and also compared differences in mean responses of ecosystem indicators between simulations during defined time periods using the Tukey–Kramer HSD (honestly significant difference) test using an alpha value of 0.05 to determine significant differences.

### 3 | RESULTS

#### 3.1 | Land-use scenarios

Based on our input datasets, cropland area in our modeled domain increased from 94,000 km² (12%) of the UMRB in 1901 to 252,000 km² (32%) by the end of the historic period, 2003 (see Figure 1). In contrast, pastureland remained relatively steady over the historic time period with approximately 96,000 km² in pastureland in 1901 (nearly the same area as cropland) and 103,000 km² (13% of the domain) by 2000 (Figure 1). Out of the 11 LPJmL original CFTs, eight were
represented in our domain with temperate cereals making up 63% of all cropland area c. 2003, followed by maize, the only C4 CFT modeled in the region, at (16.5%), soybean (10%) and sunflower (4%). The CFTs tropical cereals, temperate roots, pulses, and rapeseed, each covered between 1% and 2% of the study domain. Approximately 6% of cropland was irrigated as of 2003 in the UMRB. While only 5% of the temperate cereals area was irrigated but accounted for half the irrigated cropland area. Fourteen percent of the 42,000 km² in maize production was irrigated while 18% of pulse crops and nearly 40% of temperate root crops were irrigated (Table S2). To evaluate model performance for crops, we compared yields of the four most common crop categories in the UMRB averaged for a 30 year period (1975–2005) to those reported for the region by the United States Department of Agriculture (USDA) National Agriculture Statistics Service “Quick Stats” (2019). As this was not a primary objective of this paper, we present results in the supplementary document.

In developing our scenarios of potential locations for bioenergy switchgrass planting we excluded just over 252,000 km² of lands identified as agricultural crops. We further excluded another 205,000 km² from all scenarios by using high-resolution data to exclude woody, built up, and snow-covered areas as well as public and private lands with permanent conservation status. We allowed for conversion of pasturaleands on non-public lands in the AP scenario as noted, which resulted in just over 333,000 km² available for switchgrass bioenergy conversion. Potential available land for switchgrass bioenergy production dropped to 236,000 km² if we maintained current pastureland (A). The conservation scenario, C, constrained land for potential switchgrass bioenergy to 150,000 km² (Figure 1). Approximately 20%–25% of all the land in each scenario identified as potential available for bioenergy crops fell within the boundaries of Tribal Nations (see Figure S2).

### 3.2 Climate

Increases in air temperature were predicted under the representative RCP 2.6 and 8.5 scenarios relative to the average temperatures in the second half of the 20th century over the UMRB (Figure 2a; Figure S3; Table S3). Air temperatures under both the RCP 2.6 and RCP 8.5 diverged from the recycled climate early within the 21st century, with average annual temperatures in 2040 ~2.5°C and 3.5°C higher than the average annual temperature of 6°C in 1970–1999. During the second half of the 21st century, temperature from the RCP 2.6 scenario remained relatively stable whereas the RCP 8.5 continued to rise to an annual average temperature of more than 14°C (Figure 2a). Total annual precipitation showed larger interannual variability; annual

![Figure 2](image-url)
**TABLE 2** Ten year average yields (tons DM/ha), harvest (Tg DM), net energy content of harvest (EJ), and potential electricity generation capacity of harvest. The average yield for the entire planting period (2020–2090), total cumulative harvest and energy potential over the 80 year time period (2020–2099)

| Year       | Scenario                          | Area planted (1,000 km²) | Average yield (tons DM/ha) | Harvest (Tg DM) | Energy content (EJ) \(a\) | Equivalent electricity generating capacity estimate (MWh) \(b\) |
|------------|-----------------------------------|--------------------------|---------------------------|-----------------|-----------------------------|-------------------------------------------------------------|
|            |                                   |                          | RCYL | RCP2.6 | RCP8.5 | RCYL | RCP2.6 | RCP8.5 | RCYL | RCP2.6 | RCP8.5 | RCYL | RCP2.6 | RCP8.5 |
| 2030       | Conservation (C)                  | 56.66                    | 7.45 | 5.42   | 7.31   | 41.47 | 31.15  | 40.55  | 0.7  | 0.6    | 0.7    | 5,115 | 3,842  | 5,001 |
| (2025–2034)| Aggressive (A)                   | 80.93                    | 7.35 | 5.36   | 7.18   | 58.54 | 44.03  | 56.92  | 1.1  | 0.8    | 1.0    | 7,220 | 5,431  | 7,020 |
|            | Aggressive with pasture (AP)      | 87.54                    | 6.98 | 5.14   | 6.81   | 60.39 | 46.08  | 58.62  | 1.1  | 0.8    | 1.1    | 7,448 | 5,684  | 7,230 |
| 2050       | Conservation (C)                  | 121.92                   | 6.53 | 5.72   | 4.58   | 78.95 | 69.72  | 55.74  | 1.4  | 1.3    | 1.0    | 9,738 | 8,599  | 6,874 |
| (2045–2054)| Aggressive (A)                   | 182.55                   | 6.36 | 5.59   | 4.52   | 115.12| 101.83 | 82.36  | 2.1  | 1.8    | 1.5    | 14,198| 12,559 | 10,158|
|            | Aggressive with pasture (AP)      | 220.34                   | 6.54 | 5.77   | 4.68   | 142.60| 126.83 | 102.94 | 2.6  | 2.3    | 1.9    | 17,588| 15,643 | 12,697|
| 2070       | Conservation (C)                  | 145.24                   | 5.57 | 6.52   | 4.20   | 80.81 | 94.90  | 60.91  | 1.5  | 1.7    | 1.1    | 9,967 | 11,705 | 7,513 |
| (2065–2074)| Aggressive (A)                   | 225.80                   | 5.35 | 6.30   | 4.05   | 120.68| 142.56 | 91.31  | 2.2  | 2.6    | 1.6    | 14,884| 17,582 | 11,261|
|            | Aggressive with pasture (AP)      | 301.56                   | 5.58 | 6.49   | 4.22   | 167.72| 196.63 | 126.60 | 3.0  | 3.5    | 2.3    | 20,686| 24,251 | 15,615|
| 2090       | Conservation (C)                  | 149.97                   | 6.29 | 4.55   | 3.91   | 94.30 | 68.17  | 58.70  | 1.7  | 1.2    | 1.1    | 11,630| 8,408  | 7,240 |
| (2085–2094)| Aggressive (A)                   | 235.64                   | 6.17 | 4.40   | 3.81   | 145.37| 103.65 | 89.70  | 2.6  | 1.9    | 1.6    | 17,929| 12,784 | 11,064|
|            | Aggressive with pasture (AP)      | 331.54                   | 6.27 | 4.52   | 3.92   | 207.85| 149.89 | 129.98 | 3.7  | 2.7    | 2.3    | 25,636| 18,487 | 16,031|
| 2020–2099  | Conservation (C)                  | —                        | 6.36 | 5.29   | 5.05   | 5,997.2| 4,818.4| 4,530.3| 108.0| 86.7   | 81.6   | 739,671| 594,277| 558,750|
|            | Aggressive (A)                   | —                        | 6.21 | 5.17   | 4.97   | 8,942.4| 7,183.6| 6,791.7| 161.0| 129.3  | 122.3  | 1,102,920| 886,001| 837,655|
|            | Aggressive with pasture (AP)      | —                        | 6.25 | 5.21   | 4.99   | 11,883.2| 9,503.8| 8,938.3| 213.9| 171.1  | 160.9  | 1,465,630| 1,172,161| 1,102,415|

The bold value indicates the mean across years.

\(a\)McLaughlin and Kszos (2005) 18 GJ per MG of DM.

\(b\)MWh were calculated assuming a 20% loss in biomass during storage (Searle and Malins) and a 27% plant efficiency for a power plant with CCS (Baik et al., 2018).
precipitation averaged 409 ± 4 (mean ± standard deviation) mm and ranged from 253 to 518 mm during 1960–2004. A small but significant decline in average annual precipitation was detected for the RCP 8.5 scenario with the predicted mean annual precipitation dropping to 369 ± 64 mm over the second half of the 21st century, with the largest declines occurring between June and August (Figure 2b; Figure S3; Table S3). No significant change in precipitation was observed between the RCP 2.6 scenarios and the latter part of the 20th century. There was a noticeable but non-significant decline in mean monthly precipitation in July and August during the second half of the 21st century (Figure 2b; Figure S3; Table S3).

3.3 Switchgrass harvest and bioenergy potential

Cumulative switchgrass harvest, integrated over 2020–2099, ranged from 4.530 Tg total dry matter (DM) under the Conservation RCP 8.5 scenario to 11.883 Tg DM under the Aggressive with Pasture with recycled climate scenario (Table 2). During the last two decades of the study period (2080–2100), when land conversion had largely stabilized, the Aggressive with Pasture under the RCP 8.5 climate produced lower average annual harvests (146 ± 13 (mean ± 95% CI) Tg DM/year) than the Aggressive scenario assuming a recycled climate (156 ± 14 Tg DM/year) despite having 40% more land in switchgrass production (Figure S4). Similarly the Aggressive scenario under the RCP 8.5 scenario produced approximately the same tonnage (100 ± 9 Tg DM/year) as the Conservation scenario assuming a recycled climate (102 ± 9 Tg DM/year) despite having 55% more land in switchgrass production (Figure 3; Figure S4). Large interannual variability was observed in the simulated annual switchgrass yield and total production for all climate and land-use change scenarios (Figure 3; Table 2; Figures S4–S6). Both the RCP 2.6 and 8.5 scenarios showed statistically significant negative trends in mean annual switchgrass yield over time from ~6 to 4 Mg DM ha⁻¹ year⁻¹ (Figure 3; Table 2; Figure S4). Generally speaking, yield decreased as differences in the estimated productivity of the lands between planting scenarios was minimal with no significant difference detected in the mean yields between switchgrass planting scenarios when climate was held constant (Figure 3; Figure S4).

The magnitude of climate impacts on food-crop yields and their total harvest varied by crop type. For example, rainfed winter wheat and rapeseed (largely canola in the UMRB) showed significant increases in yield under the most warmer climate and high CO₂ scenario, with total dry winter wheat biomass harvest and yield more than doubling from 15 Tg DM (~2 tons DM/ha) at the start of the 21st century to over 30 Tg DM (~4 tons DM/ha) at the

![FIGURE 3](image-url)  
Average annual switchgrass harvest (dry material, Tg) under various climate and land-use scenarios with the equivalent net energy conversion are shown on the left panel for the 10 year period centered around 2040 (top) and 2090 (bottom). Average annual yields (metric tons dry mass per hectare are shown in the right panel with corresponding standard deviation for the decadal time period)
end of the century and rapeseed biomass increasing nearly fivefold from 0.1 Tg to more than 0.5 Tg. Conversely, rainfed maize pulse crops, temperate roots, and soybean all showed declining yield trends given warmer air temperatures and as confirmed by empirical observations of temperature versus yield relationships (Lobell & Field, 2007). Irrigated temperate roots and pulse crops showed a significant increase in yield under the RCP 8.5 scenario (Table S2; Figure S7).

3.4 | Ecosystem carbon dynamics

Both the above- and below-ground stocks and fluxes of carbon from natural ecosystems showed large responses to climate change. As the combined effects of warming and atmospheric water demand increased, biomass turnover became faster as carbon losses via mortality and subsequent decomposition (Rh) outpaced gains in NPP (Table S4; Figure 4). These responses are important to consider when determining the impact of land-use change associated with switchgrass plantings on total ecosystems stocks and fluxes.

Large interannual variation was observed in NBP (Figure 4; Figure S8), with differences between climate and land-use scenarios becoming more evident after the mid-21st century, but simulated NBP started to diverge in 2005 (Figure 4; Table 3). When comparing mean annual fluxes of individual components of NBP between all 12 simulations between 2020 and 2099, we did not see any significant differences due to climate or land use (α = 0.05); however, NPP was highest in the RCP 8.5 baseline management scenario and declined with more aggressive switchgrass conversion, with the exception being a higher mean annual NPP in the Aggressive scenario versus the Aggressive with Pasture scenario. Mean annual rH showed significant increases in respiration with intensified climate change, outweighing increases in NPP, and significantly declined with increasing switchgrass conversion, with no significant change between the Aggressive and Aggressive with Pasture scenarios. Carbon fluxes in the UMRB from all crops harvested showed no significant difference between climate scenarios, whereas increased switchgrass conversion lead to significantly higher harvest rates than fixed land use. Losses of carbon decreased as natural lands converted to switchgrass, while at the same time fluxes from land-use change were highest in the Aggressive and Aggressive with Pasture scenarios assuming 20th century climate (Table S4). Only the 20th century Baseline scenario showed positive NBP; however, the interannual variation of NBP in this scenario was large and over half the simulated years resulted in a net carbon source to the atmosphere across the UMRB (Figure 4; Table 3).

Live vegetation biomass stocks of natural ecosystems showed larger responses to climate change than to land-cover and land-use change (Figure 5a). As of 2005 the average standing stock of vegetation carbon in the UMRB was found to be ~1,200 Tg. By comparing our no land-use change scenarios (Baseline) under various climates we could isolate the impact of climate and CO₂, thus also allowing us to then isolate the impact of land-use change. Similarly, when we look at the impact of a given land-use scenario, we compare it to the Baseline scenario under the same climatic conditions. By 2090 assuming the Baseline (B) scenario of no land-use change, natural ecosystems lost
~400 Tg C stored in vegetation under RCP 2.6 conditions and ~875 Tg C under RCP 8.5. In contrast a smaller, ~90 Tg C increase was modeled in live vegetation under climate conditions followed similar patterns to those in late 20th century climate conditions and fixed atmospheric CO2 (Figure 5; Table 4). By 2090 losses in vegetation carbon from Aggressive switchgrass planting ranged from 475 Tg C under the Recycled climate conditions, to 250 Tg C under RCP 2.6 and 125 Tg C under RCP 8.6 climatic conditions. While the most extreme RCP losses appear to be smaller but this is largely because there was less ecosystem carbon to lose during conversion. Planting additional switchgrass on pasture and grazing lands only lead to an additional loss of ~5–15 Tg under by the end of the 21st century under the respective climate scenarios, bringing total stocks of vegetation carbon down to a 200 Tg C planting scenario (Figure 5a; Table 4). When critical ecological areas were protected from conversion, losses from conversion were approximately halved.

Declines in litter and soil carbon through the 21st century followed similar trends to vegetation carbon. Litter and soil carbon showed an initial increase in pools as mortality increased, unlike vegetation carbon, which declined relatively steadily over time (Table 4). Litter carbon showed similar declines in response to climate change and intensified land-use change to switchgrass. When assuming a late 20th century climate, losses in litter carbon from switchgrass expansion ranged from 250 under the conservative scenario to 480 Tg under the most Aggressive with Pasture land-use scenario. Similar losses of 250–425 Tg C of litter from climate change

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**TABLE 3** Ten year average cumulative net biome productivity (NBP) beginning in 2005. Units are Tg C/year

| Year          | Climate scenario | Land management scenario | Baseline (B) | Conservation (C) | Aggressive (A) | Aggressive with pasture (AP) |
|--------------|------------------|--------------------------|--------------|-----------------|---------------|-----------------------------|
| 2030 (2025–2034) | Recycle          |                          | −180.8       | −389.7          | −491.9        | −332.8                      |
|               | RCP 2.6          |                          | −145.2       | −347.3          | −448.2        | −299.5                      |
|               | RCP 8.5          |                          | −246.1       | −469.9          | −577.5        | −419.5                      |
| 2050 (2025–2034) | Recycle          |                          | −102.6       | −799.1          | −1162.6       | −1020.5                     |
|               | RCP 2.6          |                          | −409.8       | −1081.5         | −1438.2       | −1283.1                     |
|               | RCP 8.5          |                          | −485.9       | −1185.5         | −1543.9       | −1343.7                     |
| 2070 (2025–2034) | Recycle          |                          | 206.1        | −934.6          | −1554.7       | −1618.3                     |
|               | RCP 2.6          |                          | −634.7       | −1647.6         | −2191.8       | −2191.9                     |
|               | RCP 8.5          |                          | −1353.1      | −2206.8         | −2684.1       | −2610.5                     |
| 2090 (2025–2034) | Recycle          |                          | 336.9        | −1115.1         | −1935.3       | −2238.0                     |
|               | RCP 2.6          |                          | −840.4       | −2075.2         | −2765.2       | −2965.8                     |
|               | RCP 8.5          |                          | −1894.6      | −2931.8         | −3535.2       | −3565.3                     |

The italicized value indicates the mean across years.
| Year       | Scenario                        | Vegetation (Tg C) | Litter (Tg C) | Soil (Tg C) |
|------------|---------------------------------|-------------------|--------------|-------------|
|            |                                 | Recycle | RCP2.6 | RCP8.5 | Recycle | RCP2.6 | RCP8.5 | Recycle | RCP2.6 | RCP8.5 |
| 2030 (2025–2034) | Baseline (B)                    | 107.6   | 1,014.1 | 955.8 | 1,258.7 | 1,274.7 | 1,266.3 | 10,534.2 | 10,615.7 | 10,581.5 |
|            | Conservation (C)                | 979.7   | 931.3   | 868.2 | 1,208.2 | 1,217.0 | 1,205.1 | 10,472.0 | 10,554.1 | 10,506.5 |
|            | Aggressive (A)                  | 921.2   | 877.7   | 813.3 | 1,189.2 | 1,194.0 | 1,181.2 | 10,447.3 | 10,529.7 | 10,477.6 |
|            | Aggressive with Pasture (AP)    | 1,013.1 | 955.7   | 897.2 | 1,223.1 | 1,235.3 | 1,218.4 | 10,480.6 | 10,559.1 | 10,514.5 |
| 2050 (2045–2054) | Baseline (B)                    | 1,118.9 | 918.8   | 812.3 | 1,246.5 | 1,187.4 | 1,192.0 | 10,573.1 | 10,541.9 | 10,575.4 |
|            | Conservation (C)                | 937.5   | 785.9   | 688.6 | 1,069.4 | 1,014.6 | 1,011.6 | 10,239.9 | 10,177.1 | 10,179.3 |
|            | Aggressive (A)                  | 811.3   | 638.3   | 601.0 | 986.1   | 926.8   | 917.8   | 10,089.1 | 10,011.3 | 10,001.5 |
|            | Aggressive with Pasture (AP)    | 861.2   | 720.6   | 637.8 | 1,017.8 | 965.3   | 970.1   | 10,152.3 | 10,094.1 | 10,114.4 |
| 2070 (2065–2074) | Baseline (B)                    | 1,264.7 | 883.7   | 508.7 | 1,293.8 | 1,148.6 | 975.9   | 10,664.7 | 10,435.3 | 10,179.3 |
|            | Conservation (C)                | 1,018.4 | 732.8   | 438.3 | 1,043.8 | 932.7   | 817.0   | 10,012.6 | 9,758.7  | 9,547.2  |
|            | Aggressive (A)                  | 844.8   | 628.4   | 389.0 | 913.1   | 812.6   | 719.7   | 9,690.7  | 9,422.0  | 9,211.3  |
|            | Aggressive with Pasture (AP)    | 848.0   | 623.8   | 386.7 | 873.3   | 786.9   | 718.8   | 9,653.7  | 9,433.3  | 9,278.3  |
| 2090 (2085–2094) | Baseline (B)                    | 1,291.1 | 792.6   | 327.0 | 1,332.2 | 1,077.1 | 906.3   | 10,762.2 | 10,339.6 | 9,921.7  |
|            | Conservation (C)                | 1,014.6 | 648.7   | 253.0 | 1,076.3 | 869.0   | 760.0   | 9,843.6  | 9,456.7  | 9,104.8  |
|            | Aggressive (A)                  | 815.5   | 544.5   | 204.0 | 934.1   | 751.2   | 670.3   | 9,364.8  | 8,988.2  | 8,640.1  |
|            | Aggressive with Pasture (AP)    | 799.5   | 532.8   | 198.9 | 847.8   | 696.7   | 662.3   | 9,164.3  | 8,854.4  | 8,623.1  |
alone were observed by 2090 when no switchgrass conversion was considered. Carbon losses were highest for the soil stocks, for which land management decisions, such as removal of switchgrass and its residue via harvest, had approximately double the impact on stored soil carbon than climate. Losses of carbon attributed to climate alone ranged from 422 to 850 Tg C, while losses from land-use change alone ranged from 925 to 1,600 Tg C under the 20th century climate scenario (Figure 5; Table 4).

3.5 | Ecosystem water balance

Mean annual evapotranspiration (ET) across the UMRB averaged ~340 mm/year with an interannual variability ranging from 210 to 425 mm/year. There was a negligible impact of land use on ET differences between land-use scenarios when climate is held constant (Figure 6). While no trend was detected in the RCP 2.6 or baseline scenario, ET decreased under the RCP 8.5 scenario and
averaged ~320 mm/year across the second half of the 21st century. Breaking ET down into its individual components of evaporation and transpiration showed larger differences between simulations as higher evaporation and lower transpiration were simulated as switchgrass fractions increased on the landscape (Figure 6). Under the most aggressive switchgrass planting scenarios, we observed a near reversal of dominance from transpiration to evaporation across the UMRB by the end of the century (Figure S9). Average annual runoff was much lower than either evaporation or transpiration and averaged ~55 mm/year under baseline 20th century climate conditions with a range of 25–95 mm/year. Runoff generally increased with increasing switchgrass conversion on natural lands, but the influence of climate was larger with mean runoff dipping to 40 mm/year under the RCP 8.5 baseline scenario (Figure S9). LPJmL simulates a shift in peak transpiration and ET to earlier times of the year along with a significant decline in both fluxes summer and early fall (Figure S10). With more intense land-use change due to switchgrass conversion, a significant increase in evaporation was detected throughout the year with the largest peak in April and May, as well as smaller increases in runoff.

4 | DISCUSSION

4.1 | Bioenergy harvest and net CO₂ reduction potential

Large portions of the UMRB, including eastern Montana, western North Dakota and South Dakota and dispersed areas across Wyoming, have been highlighted as some of the best suited lands in the contiguous United States for BECCS development due to proximity of CO₂ storage basins, infrastructure, and land bioenergy production potentials (Baik et al., 2018). Bioenergy production potential often focuses on near-term opportunities (2030–2040) within existing managed agricultural, pasture, and forest lands (Downing et al., 2011; Efroymson & Langholtz, 2017; Langholtz, Stokes, & Eaton, 2016; Perlack et al., 2005), and here we attempt to characterize long-term potential of dedicated switchgrass bioenergy crops on “marginal” lands. We also characterize the spatial and temporal variability and trends in switchgrass yield due to changes in climate, atmospheric CO₂ and the trade-offs for carbon and water budgets associated with land-use and land-cover change.

Previous studies have shown large variability (1–35 Mg dry biomass/ha) in yields of lignocellulosic energy grasses such as switchgrass that depend on climate and management (Krause et al., 2017; LeBauer et al., 2017; Schmer et al., 2008; Searle & Malins, 2014). Similarly, we found large variability in our rainfed switchgrass yields across the UMRB (0.5–12 Mg/ha), as well as large spatial gradients in yield, with the eastern UMRB showing highest and most consistent yields from year-to-year. The higher interannual variability in switchgrass yields in the western UMRB raises concerns that biomass sources may not be reliable for providing sustained energy production (Downing et al., 2011). We suggest that interannual variability in climate be considered in addition to infrastructure limitations when estimating baseloads for power production. The yield increases and sensitivity to climate variability may be minimized to some extent as technological advances alter crop varieties and their response to flash droughts and temperature extremes. Additional research on technological advances toward achieving yield increases highlighted in the 2016 Billion-Ton report of the US Department of Energy (Efroymson & Langholtz, 2017; Langholtz et al., 2016) assume either 1% or 3% increase over 2015 yields for agriculture. This is within the range of McLaughlin, Kiniry, Taliaferro, and Torre Ugarte (2006) who consider an annual yield gain of 1.5% year in the U.S. Northern Plains, and report a mean 2003 baseline yield for this region at 7.8 Mg/ha (4.5–12.3 Mg/ha). They also projected increases in average bioenergy production to 8.9 and 10.1 Mg/ha in 2015 and 2025 respectively due mainly through the breeding of higher-yielding cultivars. Importantly, in agreement with previous research on forage and crops in the region, we found a decline in switchgrass yields over time due to increasing temperatures, confirmed by Conant et al. (2018), that exceeded any marginal gains from increasing CO₂. The lack of CO₂ fertilization effect is due in part to the C4 photosynthetic pathway of switchgrass that already concentrates CO₂ around the Rubisco enzyme (Ehleringer & Bjorkman, 1977; Ehleringer, Cerling, & Helliker, 1997; Sage & Kubien, 2007).

We chose not to assess the effects of specific management practices on switchgrass such as soil management and fertilization, whereas we allowed for irrigation on food crops that minimized climate-related water stress (Rosenberg et al., 1999). Field studies have found that switchgrass responds positively to nitrogen fertilizer additions; Mulkey, Owens, and Lee (2006) found that fertilizing at 56 kg N/ha did lead to an increase in biomass, no additional benefit was observed for further additions and the carbon cost of conventional fertilizer due to the Haber–Bosch process needs to be considered for full carbon accounting (Izaurralde, McGill, Rosenberg, & Schlesinger, 2000). Similar results were obtained for switchgrass grown in a mixture with big bluestem (Andropogon gerardi Vitman), a native C4 grass, while noting that nitrogen application did not always result in significant increases in production (Mulkey, Owens, & Lee, 2008). Ecosystem carbon stocks can also be enhanced using lower energy-intensive agricultural practices, such as conservation agriculture (Valdez et al., 2017). Switchgrass grown in species mixtures in the field have been shown to have lower yields and little to no...
reaction to nitrogen inputs and irrigation as compared to monocultures; switchgrass grown in a mixture could still provide a valuable feedstock for bioenergy as well as increasing environmental and biodiversity benefits (Searle & Malins, 2014) and achieved switchgrass yields were not significantly lower than monocultures when grown with legumes (Wang, Lebauer, & Dietze, 2010).

Whereas the US Department of Energy’s Billion-Ton Report focuses on the use of existing lands in active agricultural and forest management, we focused on lands traditionally defined as marginal. Searle and Malins (2014) suggest that government policies prioritize safeguards to preserve existing cropland for food production to address food security concerns over food shortages and price hikes like those seen in 2008 caused by increased biofuel production (Searle & Malins, 2015). Our Aggressive land-cover and land-use change scenario for the UMRB most closely resembles the definition used by Searle and Malins (2015) where forest, wetlands, permanent urban ice/water/rock, and cropland are excluded from switchgrass production. We evaluated various other datasets to define marginal land over our domain and found similar results to Emery, Mueller, Qin, and Dunn (2017) who assessed the potential of marginal land for cellulosic feedstock production and concluded that very few land parcels in the United States comply with multiple definitions of marginal land. While it is unrealistic that all these lands would be converted, we believe that our estimates provide an upper limit of switchgrass energy crop production, particularly the spatial patterns of yield that agree with empirical studies, that is, Gelfand et al. (2013). For example, second-generation crops may eventually compete with food crops if climate and population growth drive increasing food demand and expansion of irrigated agriculture into marginal lands (Ray, Ramankutty, Mueller, West, & Foley, 2012).

Tribal Lands include between 20% and 25% of the potential lands that could be converted to second-generation bioenergy production in the UMRB. We decided to include lands managed by Tribal Nations in our assessments given their diversity of land management strategies and ongoing leadership in alternative energy projects (Doris, Lopez, & Beckley, 2013; Fiato et al., 2013). Tribal Lands comprise some 2% of land in the United States by area, but some 5% of the renewable energy resources (NREL, 2012). Like all land transitions, any bioenergy development needs to be balanced with societal and cultural values (Stoy et al., 2018).

Based on our modeling assumptions, we estimated the energy production potential of switchgrass in the context of powering existing coal-fired power plants in the UMRB. We determined net energy content of harvested dry mass using conversion net energy recovery rates of 18 GJ/Mg of dry mass as measured by McLaughlin et al. (1999; Table 5). After 10 years of planting we estimated the mean potential energy of harvest material to be between 0.6 and 1.1 EJ/year (Table 2). During the last two decades of the 21st century, at which point lands converted to switchgrass had stabilized, the annual potential energy of modeled switchgrass harvest averaged from 1.1 to 1.7 EJ/year for the Conservative scenario under all different climate scenarios investigated here. Potential energy increased to 1.6–2.6 EJ/year under the Aggressive scenario and 2.3–3.7 EJ/year when conversion of pastureland to switchgrass was allowed in the Aggressive with Pasture (Figure 3; Table 2). No significant difference was found in the mean energy potential during the last decade of the 21st century between the RCP 2.6 and RCP 8.5 scenarios (Figure S9). Assuming perfect conversion efficiency, the range of estimates could provide power plants with a combined capacity of 33,500–118,700 MW/year towards the end of the 21st century. To estimate a more realistic production capacity of the switchgrass harvested, we further considered loss in DM yield due to storage at 20%, which has typically been found to be lower than Miscanthus (McLaughlin & Kszos, 2005; Searle & Malins, 2014). Further we considered a conservative efficiency factor of 0.27 for power plants using biomass equipped with CCS (Baik et al., 2018), bringing our estimates down to an average 7,200–28,000 MW/year by the end of the 21st century. At the same time assuming no infrastructure constraints to CCS and assuming a 90% CO₂ capture efficiency (Eldardiry & Habib, 2018) in addition to the 20% loss in biomass during storage, the carbon capture rate could average 21–34 Tg C/year under a conservation-based scenario to 47–75 Tg C/year under an aggressive conversion scenario that included grazing and pasture lands. However, this range is still quite optimistic for all scenarios as it assumes all harvestable biomass is used for energy conversion independent of location within the UMRB, and also assumes no infrastructure constraints.

To put this in regional context, we compare the results to the emissions from the Colstrip power plant in Montana, the largest coal-powered power plant in the UMRB and currently one of the largest emitters of CO₂ east of the Mississippi (~4 Tg C/year; Table S5, with a generating capacity of just over 2,000 MW). We found that even our most conservative scenario fit the production needs of the plant by 2030 for generating a year’s worth of electricity while considering losses in feedstock from storage and plant efficiency. As previously discussed, interannual variation in biomass production is higher in the Western portion of the UMRB and needs to be considered as the range of potential generation capacity of switchgrass harvested during low years could drop to 1,900 MW or rise to 9,000 MW. Losses from crop transport are not considered in our analysis such that analyses here should be considered a potential approximation of bioenergy supply. At global scales, Beringer, Lucht, and Schaphoff (2011) estimated that between 26 and 116 EJ/year can be generated via rainfed bioenergy crop production (Beringer et al., 2011); this amount could be increased with irrigation to
52–174 EJ/year. In this global context we found that switchgrass bioenergy production grown on marginal lands within the UMRB could provide around 1%–3% of the upper estimated global bioenergy supply from bio-plantations.

4.2 Ecosystem services of native and managed grasslands

There are multiple trade-offs to consider in expanding bioenergy production that include changes to carbon and water budgets, effects on habitat for biodiversity, ecosystem health, societal values and more. While we could not account for all of these trade-offs with our current model set up alone, we did find that land-use change leads to a loss of 75–490 Tg C of aboveground biomass by the end of the century. Climate impacts on live biomass are larger and accounted for an additional loss of 255–965 Tg C between RCP 2.6 and 8.5. When adding carbon stored in soil and litter, the UMRB could lose up to 1,000–2,600 Tg C from conversion of current lands to switchgrass while 1,200–2,200 Tg C could be lost under the RCP 2.6 and RCP 8.5 scenarios by the last decade of the 21st century with no switchgrass conversion. The soil carbon losses are due to decreased carbon inputs as switchgrass is harvested and moved off site. The largest potential for loss of carbon storage from intensified switchgrass conversion occurred under the late 20th century climate scenario, for which the Baseline land-use scenario showed a slight gain in vegetation and ecosystem carbon storage over the 21st century. Thus, more vegetation C was lost per unit area converted to switchgrass as compared to the other climate scenarios in which carbon losses were independent of land-use change. Converting existing pasture and grazing lands to switchgrass production for bioenergy had little additional ecosystem C impact on average. Previous studies suggesting that grazing and managing switchgrass for bioenergy production in the great plains can occur at profitable rates if slight declines to both the grazing intensity and total switchgrass yield are accepted (Biermacher, Haque, Mosali, & Rogers, 2017). The impact of switchgrass conversion on ecosystem carbon storage could be as large as the loss from carbon stored in natural lands, highlighting need for earlier actions to lessen possibilities of most extreme climate change. We also find that switchgrass expansion also affects the hydrologic cycle by reducing transpiration and increasing evaporation, which can further exacerbate regional warming by changing latent energy fluxes (Gerken, Bromley, & Stoy, 2018). Runoff was also shown to increase under switchgrass conversion because of a combination of longer periods of the year with no vegetation cover and because of reduced transpiration. When conservation of ecosystems was not prioritized, ~10 million ha of lands flagged as having very high ecological value in our study region by TNC for reasons such as wildlife habitat, biodiversity reservoirs, or water quality protection services were converted by the end of the century. If no action is taken and climate and atmospheric carbon concentrations follow trajectories similar to those explored in this study, there could be significant ecological impacts independent of a mitigation strategy such as bioenergy production.

5 CONCLUSIONS

The UMRB has high potential for second-generation bioenergy production given the large area of marginal lands in private or Tribal ownership available for planting. Without constraints from infrastructure and transport, local land-use policies, and assuming high-efficiency in converting biomass to energy, the UMRB could play a major role within the United States in driving a BECCS-type climate mitigation strategy. However, we find that the switchgrass expansion scenarios drive losses in ecosystem carbon, in both vegetation and soil, that lead to cumulative losses of 1–2.5 Pg C (Figure 5b; Table 5), which increase to 3.5 Pg C losses if climate change is considered. These carbon losses must be considered with the carbon sequestered through direct air capture, which would likely accompany any BECCS-related energy activity. Expansion on switchgrass production also alters the hydrologic system of the UMRB, reducing water storage by increasing runoff in already water-limited ecosystems. Further work evaluating how additional ecosystem processes and services would be impacted include habitat for biodiversity and changes to the nitrogen cycle (Kim et al., 2018; Krause et al., 2017).

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**SUPPORTING INFORMATION**

Additional supporting information may be found online in the Supporting Information section.