GLOBAL CHANGE BIOLOGY
BIOENERGY

GCB Bioenergy (2014), doi: 10.1111/gcbb.12212

Bioenergy crop productivity and potential climate change mitigation from marginal lands in the United States: An ecosystem modeling perspective

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Abstract

Growing biomass feedstocks from marginal lands is becoming an increasingly attractive choice for producing biofuel as an alternative energy to fossil fuels. Here, we used a biogeochemical model at ecosystem scale to estimate crop productivity and greenhouse gas (GHG) emissions from bioenergy crops grown on marginal lands in the United States. Two broadly tested cellulosic crops, switchgrass, and *Miscanthus*, were assumed to be grown on the abandoned land and mixed crop-vegetation land with marginal productivity. Production of biomass and biofuel as well as net carbon exchange and nitrous oxide emissions were estimated in a spatially explicit manner. We found that, cellulosic crops, especially *Miscanthus* could produce a considerable amount of biomass, and the effective ethanol yield is high on these marginal lands. For every hectare of marginal land, switchgrass and *Miscanthus* could produce 1.0–2.3 kl and 2.9–6.9 kl ethanol, respectively, depending on nitrogen fertilization rate and biofuel conversion efficiency. Nationally, both crop systems act as net GHG sources. Switchgrass has high global warming intensity (100–390 g CO_2eq l^{-1} ethanol), in terms of GHG emissions per unit ethanol produced. *Miscanthus*, however, emits only 21–36 g CO_2eq to produce every liter of ethanol. To reach the mandated cellulosic ethanol target in the United States, growing *Miscanthus* on the marginal lands could potentially save land and reduce GHG emissions in comparison to growing switchgrass. However, the ecosystem modeling is still limited by data availability and model deficiencies, further efforts should be made to classify crop-specific marginal land availability, improve model structure, and better integrate ecosystem modeling into life cycle assessment.

Keywords: biofuel, global warming potential, greenhouse gas emission, land use change, life cycle assessment, *Miscanthus*, nitrous oxide, switchgrass

Received 24 April 2014; accepted 5 June 2014

Introduction

Bioenergy, an important renewable energy produced from biological materials, is becoming an increasingly attractive energy choice in the context of economic development, energy security, and climate change. One hand, with increasing world population and rapidly growing regional and global economy, conventional fossil fuel-based energy alone is not likely to provide essential and sufficient support to the functioning of modern economies, due to its limited supply, high or volatile fossil fuel prices, and concerns about national energy independence (Field *et al.*, 2008; Hill *et al.*, 2009). On the other hand, the society is increasingly aware of the destructive impacts of conventional energy use on the environment and climate change, and looking for alternative sources of energy that are renewable and sustainable (Tilman *et al.*, 2009; Fargione *et al.*, 2010). Biofuels, compared with fossil fuels, could potentially support state energy goals, increase domestic energy supplies to reduce dependence on foreign oil and its potential disruptions, and yet reduce GHG emissions and other air pollutants (USDOE, 2011). In the United States, only about 10% of total primary energy consumption is from renewable energy sources, but biomass-derived energy makes up about half of the total renewable energy (EIA, 2012). Compared with some other renewable energy alternatives (e.g. wind, solar power), bioenergy may be one of the most viable options to adopt in the near term (USEPA, 2009).

To meet the mandate targets for biofuel production (US Congress, 2007), a large amount of land will be

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needed to grow energy crops for biomass feedstocks. Among lands that can be used for production of biofuel feedstocks, marginal lands were often introduced as a promising land option for energy cropping purpose, considering that switching food crops to biofuel crops to produce biomass on currently available croplands may raise concerns about food security, ethic issues, and unsustainable farming practices (Field *et al.*, 2008; Tilman *et al.*, 2009; Fargione *et al.*, 2010; Gramig *et al.*, 2013), while converting lands occupied by natural ecosystems (e.g. forest) to biofuel cropland could inevitably cause environmental and ecological problems such as deforestation, biodiversity loss, habitat fragmentation, and land use change induced GHG emissions (Searchinger *et al.*, 2008; Melillo *et al.*, 2009; Dauber *et al.*, 2010). Marginal land refers to those lands where a cost-effective production is not possible under given environmental conditions, cultivation techniques, agricultural management as well as other economic and legal conditions (Wiegmann *et al.*, 2008; Gopalakrishnan *et al.*, 2011), including lands such as idle or fallow cropland, abandoned or degraded cropland, and abandoned pastureland (Cai *et al.*, 2011; Gopalakrishnan *et al.*, 2011). Compared with cropland, marginal land normally has lower inherent agricultural productivity, due to its less fertile soils and often less favorable water, climate, and possibly other environmental conditions. However, certain energy crops with high resource-use-efficiencies are still capable of growing on these lands where traditional food crops may not thrive (Bandaru *et al.*, 2013; Gelfand *et al.*, 2013). For example, some perennial cellulosic crops, such as switchgrass and *Miscanthus*, could provide abundant biomass but require relatively less nutrient than food crops (Lewandowski *et al.*, 2003; Heaton *et al.*, 2004; Stewart *et al.*, 2009). These crops could therefore be used to grow biomass feedstock and produce cellulosic ethanol by using the less favored lands, and thus avoid competing with food crops for cropland (Bandaru *et al.*, 2013).

Field experiments suggested that, cellulosic energy crops or herbaceous vegetation, once well established, could produce considerable biomass feedstocks and have direct GHG emissions mitigation capacity that rivals that of conventional food crops. Switchgrass and *Miscanthus*, for example, can produce comparable or even higher biomass than traditionally used biofuel crop – maize (Fike *et al.*, 2006; Heaton *et al.*, 2008; Nikièma *et al.*, 2011). These perennial cellulosic crops normally have high conversion efficiency of photosynthetically active radiation and are able to enhance carbon (C) accumulation in a wide range of soil and climate conditions because of C4 metabolism (Heaton *et al.*, 2008). A considerable amount of C is assimilated and stored in the belowground biomass and soils, which fosters benefits for carbon dioxide (CO_2) sequestration (Don *et al.*, 2012; Monti *et al.*, 2012). In addition, cellulosic crops generally require only a very limited amount of nutrients (e.g. nitrogen fertilizer) due to their high nutrient-use efficiency, and therefore could possibly reduce fertilization induced nitrous oxide (N_2O) emissions (Lewandowski *et al.*, 2003; Monti *et al.*, 2012). Soil methane (CH_4) fluxes were negligible in these ecosystems (Drewer *et al.*, 2012). Gelfand *et al.* (2013) recently also reported in their comparative experiments that, if grown on marginal lands, successional herbaceous crops could still produce sizeable amounts of biomass and concurrently mitigate GHG emissions due to significant C sequestration in soils and reduction in N_2O emissions.

However, biomass productivity and GHG emissions regarding large-scale bioenergy expansion on marginal lands are rarely studied (Gelfand *et al.*, 2013). During the past several decades, modeling was used extensively to study regional or global scale C, nitrogen (N) dynamics, and GHG emissions of both natural (e.g. forest, grassland) and managed ecosystems (e.g. cropland) (Raich *et al.*, 1991; Bondeau *et al.*, 2007; Huang *et al.*, 2009). More recently, models were increasingly used to assess agroecosystems related to bioenergy crops, either by incorporating agricultural modules into natural ecosystem models, e.g. Agro-BGC(Di Vittorio *et al.*, 2010) and LPJml (Bondeau *et al.*, 2007), or by developing crop-specific models, e.g. ALMANAC (Kiniry *et al.*, 1992) and MISCANMOD (Clifton-brown *et al.*, 2004). These models can be applied to a large region to estimate biomass production or/and GHG emissions (Thomas *et al.*, 2013). As most previous modeling studies concentrated on the land use change due to conversion of natural ecosystems to agroecosystems, or crop switching from food crops to energy crops on cropland (Fargione *et al.*, 2008; Searchinger *et al.*, 2008; Melillo *et al.*, 2009), another land use scenario of growing energy crops on marginal lands was also important but less studied (Qin *et al.*, 2011; Gelfand *et al.*, 2013). Along with the biomass production, GHG emissions produced from or mitigated by marginal lands could significantly affect the total GHG budget in the lifecycle assessment of biofuel production, and therefore additional effort should be made to study potential C and N dynamics and GHG fluxes of these biofuel ecosystems. Here, we use a modeling approach to conduct such a study assuming switchgrass and *Miscanthus* grown on the marginal lands in the conterminous United States. The spatial estimates are made for biomass production, net carbon balance, nitrous oxide emissions, and therefore the total GHG emissions. Biofuel productivity, land use, and global warming potential are further analyzed at regional scales to meet the United States national biofuel mandate by year 2022.

Materials and methods

Energy crops

Switchgrass and *Miscanthus* were introduced as energy crops for biomass production purpose due to their considerable productivity and stress tolerance to unfavorable environments (McLaughlin & Adams Kszos, 2005; Heaton *et al.*, 2008). Switchgrass is a perennial cellulosic crop native to North America, with biomass productivity of 5–20 Mg (1 Mg = 1 t) dry matter (DM) per hectare land. It was widely tested for biomass production across the conterminous United States (Fike *et al.*, 2006; Heaton *et al.*, 2008; Wright & Turhollow, 2010). *Miscanthus* refers to a genus of several perennial grass species mostly native to the subtropical and tropical areas of Asia (Stewart *et al.*, 2009). Its yield could normally reach 20–30 Mg DM ha^{-1} if well cultivated (Heaton *et al.*, 2008). These two perennial crops could be potential biomass sources for cellulosic ethanol production. In this study, Switchgrass and *Miscanthus* are assumed to be grown on marginal lands in the United States to produce biofuel feedstocks.

Model description

AgTEM is a biogeochemical model designed for agroecosystems, by incorporating ecophysiological, biogeochemical, and management related processes into the framework of the Terrestrial Ecosystem Model (TEM) (Raich *et al.*, 1991; McGuire *et al.*, 1992; Zhuang *et al.*, 2003, 2010). The model can be used to simulate C and N dynamics of agroecosystems (Figure S1) at a daily time step, by using spatially explicit forcing data describing climate, soil, vegetation, and agronomic conditions (Qin *et al.*, 2013a,b).

In AgTEM, all algorithms related to C and N fluxes and pools are governed by five equations describing changes of ecosystem states regarding vegetation and soil (Qin *et al.*, 2013a). C cycling in the agroecosystems is modeled as following [Eqn (1)]: atmospheric CO_2 is preliminarily assimilated by plants through photosynthesis and stored in the vegetation. In the model, net primary production (NPP) is the rate at which the plants produce net useful chemical energy. It is the difference between the rate at which the plant produces useful chemical energy (GPP, gross primary production) and the rate at which some of that energy is used during autotrophic respiration. NPP represents the total available biomass of the ecosystem produced, which is partly harvested as harvestable biomass (HBIO), partly used during heterotrophic respiration and partly allocated to soil organic carbon (SOC) and belowground biomass (as in perennial crops). C of HBIO is eventually released as CO_2 through biofuel production and use. The net C balance in the ecosystem is modeled as net carbon exchange (NCE) which accounts for all C fluxes into or out of the system. A positive NCE indicates net ecosystem CO_2 sink while a negative value indicates a CO_2 source (Qin *et al.*, 2013a).

$$CO_2 \Rightarrow GPP \overset{CO_2\uparrow}{\Longrightarrow} \underbrace{NPP \overset{CO_2\uparrow}{\Longrightarrow} \left\{ \begin{array}{l} HBIO \overset{CO_2\uparrow}{\Longrightarrow} \\ \overset{CO_2\uparrow}{\Longrightarrow} C_{SOC} \end{array} \right.}_{NCE} \tag{1}$$

Modeled N_2O accounts for soil N_2O fluxes from both nitrification and denitrification, as in [Eqn (2)] (Qin *et al.*, 2013a):

$$\left. \begin{array}{l} NH_4^+ \overset{NO}{\Longrightarrow} N_2O_{ntf} \\ NO_3^- \overset{N_2}{\Longrightarrow} N_2O_{dtf} \end{array} \right\} \Longrightarrow N_2O \tag{2}$$

where N_2O_{ntf} is N_2O produced from the nitrification process of the biological oxidation of ammonia (NH_4^+) with oxygen, and N_2O_{dtf} is N_2O produced from soil nitrate (NO_3^-) through denitrification process; N2O is the total N_2O fluxes of N_2O_{ntf} and N_2O_{dtf}. Nitric oxide (NO) and nitrogen (N_2) are also produced from the processes of nitrification and denitrification, respectively, but they are not quantified in this study.

The original version of AgTEM 1.0 was calibrated at site levels and applied at regional scales to assess regional C dynamics (Qin *et al.*, 2011), biomass production (Qin *et al.*, 2012), and water balance (Zhuang *et al.*, 2013). The further developed AgTEM 2.0 incorporated processes such as biomass allocation, N cycling and agricultural management (Qin *et al.*, 2013a). In the model, most parameters describing and constraining generic ecosystem processes were either inherited from TEM or predefined in previous studies (e.g. Zhuang *et al.*, 2003, 2010; Qin *et al.*, 2011, 2012). However, some additional vegetation-specific or soil-specific parameters were defined and calibrated for certain ecosystems or processes not previously included in the model. For example, the temperature threshold parameters were determined separately for switchgrass and *Miscanthus* to describe plant photosynthesis and crop phenology. Many additional variables and parameters were included in AgTEM 2.0 to represent nitrification and denitrification processes (Qin *et al.*, 2013a). The parameterized and calibrated model was then used to estimate site-level biomass and N_2O emissions, and they were validated against field observations. The results suggested that the AgTEM 2.0 well reproduced the observations (Qin *et al.*, 2013a) and can be applied to region-level estimations (Qin *et al.*, 2013b). More information concerning AgTEM can be found in previous studies (e.g. Qin *et al.*, 2011, 2013a,b). In this study, the AgTEM 2.0 was used.

Model simulations and regional analyses

By assuming that switchgrass and *Miscanthus* will be grown on available marginal lands in the conterminous United States (Figure S2), we applied the AgTEM 2.0 separately for these two crop systems, to simulate ecosystem C and N dynamics along with crop growth, using spatially referenced data describing climate, soil, vegetation, atmospheric CO_2, and agricultural management. Model estimates were then used to assess spatial distribution of output variables of interest, including NPP, HBIO, NCE, and N_2O. Spatial analyses were finally conducted

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to estimate spatial and national biomass/biofuel production, CO_2 mitigation, N_2O emissions, and total GHG emissions.

For spatial simulations, model was run grid-by-grid to estimate C and N dynamics at a daily time step with available forcing data from 1989 to 2008. First, we initialized the model by running AgTEM to equilibrium using the first year data. The model was then spun up for 100 years repeatedly using the first 10 years' data to reach equilibrium state. We then ran the transient simulations continuously from 1989 to 2008 using transient forcing data. Spatial forcing data were organized at a 0.25° latitude × 0.25° longitude resolution for the study region. Specifically, climate data describing temperature, precipitation, cloudiness were obtained from the ECMWF (European Centre for Medium-Range Weather Forecasts) Data Server (www.ecmwf.int) and organized at a temporal resolution of 1 day from 1989 to 2008. Annual atmospheric CO_2 concentrations were collected from the NOAA Mauna Loa CO_2 record (www.esrl.noaa.gov/gmd/ccgg/trends/). The elevation data were derived from the Shuttle Radar Topography Mission (SRTM) (Farr *et al.*, 2007) and soil texture data were based on the Food and Agriculture Organization/Civil Service Reform Committee (FAO/CSRC) digitization of the FAO/UNESCO soil map of the World (1971). N fertilization was set at four input rates as 0 (N0), 50 (N1), 100 (N2), and 150 kg N ha^{-1} (N3) for both switchgrass and *Miscanthus* systems, according to field experiments (Fike *et al.*, 2006; Heaton *et al.*, 2008; Propheter *et al.*, 2010; Nikièma *et al.*, 2011). In Cai *et al.*'s (2011) study, global marginal lands were identified according to marginal agricultural productivity based on land suitability indicators such as topography, climate conditions, and soil productivity. The scenario 1 in Cai *et al.* (2011) includes marginal lands from abandoned land and mixed crop and vegetation land, and yet without sacrificing large amounts of cropland and natural lands (forest and grassland) (Figure S2). This scenario was considered as initial land use condition for the modeling purpose in this study, to represent the spatial distribution of marginal lands in the United States. The data in Cai *et al.* (2011) were reorganized at a 0.25° latitude × 0.25° longitude resolution according to the proportion of marginal lands in each pixel.

Spatial analyses were conducted for each crop ecosystem based on model simulations, using geographic information system techniques. Regional analyses based on grid outputs were presented as average of the 1990s. NPP and HBIO were computed for both spatial and national levels as primary and harvested biomass production, respectively. Using biomass-to-biofuel conversion efficiencies, biofuel production was further calculated from HBIO results. Under current technologies, the efficiency of converting biomass-to-biofuel is estimated to be about 282 l ethanol Mg^{-1} DM (Lynd *et al.*, 2008). The potential efficiency could reach about 399 l ethanol Mg^{-1} DM if advanced technologies would be available (Lynd *et al.*, 2008). Net CO_2 balances (NCE) and total N_2O emissions (N2O) were also computed to estimate spatial and national GHG emissions in terms of global warming potential (GWP). The GWP of N_2O was calculated in units of CO_2 equivalent (CO_2eq) over a 100-year time horizon. In addition, GWP was related to energy production by computing global warming intensity (GWP_i) in terms of total GWP relative to biofuel production (Qin *et al.*, 2013b).

Results

Biomass and biofuel production on marginal lands

With increasing use of N fertilizer, the biomass production at ecosystem scale also increases, in both switchgrass and *Miscanthus* ecosystems (Fig. 1). At N0 level, the switchgrass produces NPP (harvest-area weighted) of less than 400 g C m^{-2} in most areas (Fig. 1a). With N addition, the NPP production increases dramatically, especially in those areas with intense cropping, e.g. Wisconsin (Fig. 1b–d). When the N rate reaches N2 (Fig. 1c) and N3 (Fig. 1d) levels, most of the southern areas have NPP of 400–800 g C m^{-2}. In terms of biomass harvested (Table 1), switchgrass produces a national average of 3.5 Mg DM ha^{-1} each year without N application, with additional 1.4 Mg DM ha^{-1} if applied 50 kg N ha^{-1} (N1). The average HBIO could reach 5.7–5.9 Mg DM ha^{-1} with sufficient N fertilizer. *Miscanthus* generally has higher biomass productivity than corresponding switchgrass at the same N application levels (Fig. 1e–h). Without N application, the NPP reaches over 600 g C m^{-2} in most intense cropping areas (Fig. 1e), with a national average HBIO production of about 10 Mg DM ha^{-1} (Table 1). With each additional kg of N application, the *Miscanthus* HBIO increases about 50 kg DM ha^{-1} each year on average, with highest increase of 64 kg DM ha^{-1} from N0 to N1 level and lowest increase of 28 at DM ha^{-1} from N2 to N3 level. When the N rate reaches N3, *Miscanthus* produces the highest HBIO of 17.2 Mg DM ha^{-1}, which almost triples the switchgrass production (Table 1).

Production of cellulosic ethanol using the harvested biomass is highly dependent on biomass-to-biofuel conversion technologies (Table 1). Under currently available technology, switchgrass could produce about 1.0–1.7 kl ethanol from each hectare of marginal land, depending on N application and biomass production. *Miscanthus*, however, could produce 2.9–4.9 kl ethanol ha^{-1} land due to its high biomass productivity. With advanced technology available, the biofuel conversion efficiency could increase by 41.5%. Switchgrass harvested from marginal lands could therefore produce 1.4–2.3 kl ethanol ha^{-1} land and productive *Miscanthus* could produce 4.1–6.9 kl ethanol ha^{-1} land. Generally, with advanced technology and application of high-rate N fertilizer, cellulosic crops grown on marginal lands could have a considerably higher land use efficiency, in terms of biofuel production on given land, than otherwise with current technology and less use of N. *Miscanthus*, in particular, has a higher land use efficiency than switchgrass at each technology × N application level scenario.

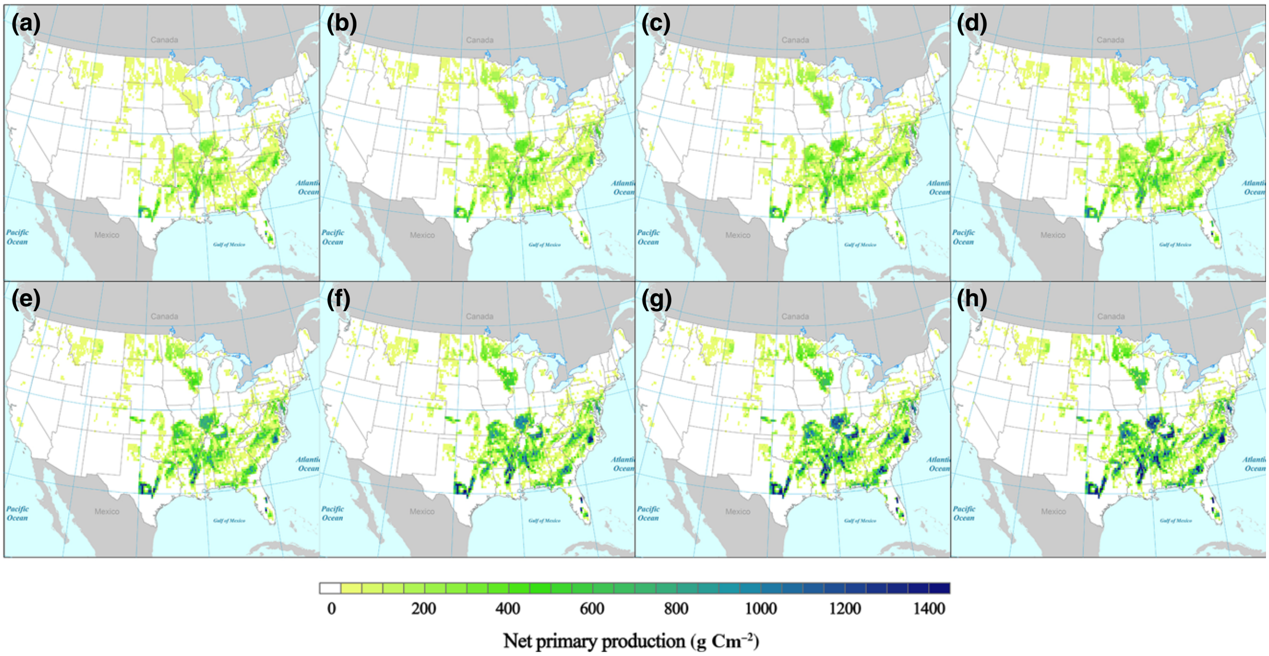

Fig. 1 Modeled net primary production from marginal lands. Area weighted estimates were made for switchgrass grown under nitrogen application levels of (a) N0, (b) N1, (c) N2, and (d) N3, and *Miscanthus* grown under (e) N0, (f) N1, (g) N2, and (h) N3.

Table 1 Estimated harvestable biomass and biofuel production from energy crops grown on marginal lands under different nitrogen application scenarios

Energy crops	Nitrogen application*	Estimated harvestable biomass production (Mg DM ha^{-1} land)	Estimated biofuel production (kl ethanol ha^{-1} land)	
			Current level†	Potential level‡
Swithchgrass	N0	3.5 (0.3)	1.0 (0.1)	1.4 (0.1)
	N1	4.9 (0.5)	1.4 (0.1)	1.9 (0.2)
	N2	5.7 (0.6)	1.6 (0.2)	2.3 (0.2)
	N3	5.9 (0.6)	1.7 (0.2)	2.3 (0.2)
Miscanthus	N0	10.2 (1.0)	2.9 (0.3)	4.1 (0.4)
	N1	13.4 (1.3)	3.8 (0.4)	5.3 (0.5)
	N2	15.8 (1.7)	4.5 (0.5)	6.3 (0.7)
	N3	17.2 (2.0)	4.9 (0.6)	6.9 (0.8)

*Nitrogen fertilization was set at four input rates as 0 (N0), 50 (N1), 100 (N2), and 150 kg N ha^{-1} (N3).
†Current and ‡potential levels of biofuel production are estimated based on current and potential biomass-to-biofuel conversion efficiencies, respectively (Lynd *et al.*, 2008). Values were averaged for the 1990s, with standard deviation in parentheses.

Greenhouse gas emissions in bioenergy ecosystems

GHG emissions (in terms of GWP) are determined by the effects of both ecosystem CO_2 and N_2O emissions. Our model experiments indicate that most of the cropping areas in the southern United States act as net sources of GHG emissions, and the estimated *Miscanthus* GWP (Fig. 2e–h) has a much higher variation than the corresponding switchgrass GWP (Fig. 2a–d) at any specific location. Specifically, in the switchgrass cropping systems, with increasing use of N fertilizer, the GHG emissions increase markedly especially in the intense cropping areas in the middle United States (Fig. 2a–d). For example, after increasing use of N, net GHG sinks in some areas become GHG sources, e.g. Texas (Fig. 2a, b), and some GHG sources become even larger sources, e.g. South Illinois (Fig. 2b, c). In the *Miscanthus* systems, however, the GHG emissions do not necessarily increase with increasing use of N (Fig. 2e–h). It is evident that, for those areas that are already GHG sources without N

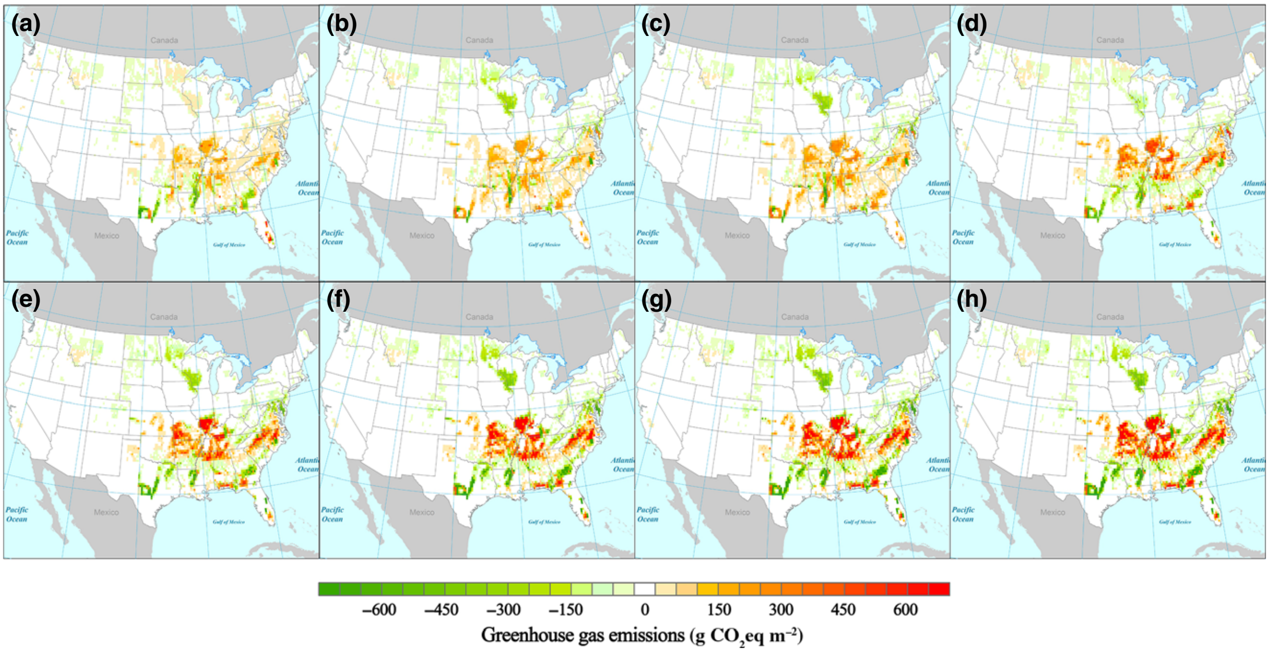

Fig. 2 Modeled GHG emissions from marginal lands. Maps show area weighted total emissions of CO_2 and N_2O (GWP) for switchgrass grown under nitrogen application levels of (a) N0, (b) N1, (c) N2, and (d) N3, and *Miscanthus* grown under (e) N0, (f) N1, (g) N2, and (h) N3. A positive value indicates a net GHG sink while a negative value indicates a net GHG source.

fertilization, e.g. Missouri, Kentucky and Tennessee in the middle of the United States (Fig. 2e), the net GWP tends to be larger after use of N fertilizer (Fig. 2f–h); but for the areas that are originally GHG sinks, e.g. Texas and Louisiana in the South United States (Fig. 2e), their GWP become even smaller, suggesting these areas become even larger GHG sinks.

From the perspective of national average GHG emissions, the changes of net GWP are simply the results of GWP changes in both CO_2 and N_2O. Both ecosystems act as GHG sources at national level and at all N application levels (Fig. 3a). Switchgrass and *Miscanthus* have a similar amount of N_2O emissions at each N application rate, and even similar C sinks at lower N rates (N0, N1). But *Miscanthus* has a much larger C sink than switchgrass at higher N rates (N2, N3). For instance, in the switchgrass systems, with increasing use of N, both N_2O emissions and CO_2 mitigation increase, but the former has a relatively larger value than the latter, resulting in a net source of GHG emissions. This is especially true when the N rate reaches N2 and N3 levels and where the total GHG emissions

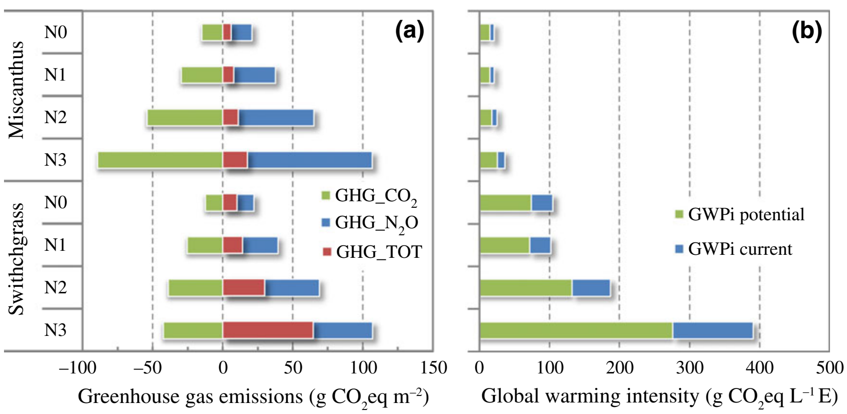

Fig. 3 National average GHG emissions from switchgrass and *Miscanthus* grown on marginal lands. (a) Contributions of CO_2 (GHG_CO2) and N_2O (GHG_N2O) to total GHG emissions (GHG_TOT) under different nitrogen application levels; (b) global warming intensity (GWPi), in terms of GWP relative to ethanol (E) production under current or potential conversion efficiencies.

reach 30 and 65 g CO_2eq m^{-2} respectively, when compared with 10 g CO_2eq m^{-2} at N0 level. By contrast, N_2O emissions and CO_2 mitigation do not change much in *Miscanthus* systems, even when the N rate increases. For example, the GWP (N0) of CO_2 and N_2O are −15 and 20 g CO_2eq m^{-2} respectively, making the net ecosystem GHG emission only 6 g CO_2eq m^{-2}. The GWP of CO_2 and N_2O reaches up to −89 and 107 g CO_2eq m^{-2}, respectively, when the N application gets to the N3 level, but the net ecosystem GHG emission is still only 18 g CO_2eq m^{-2} – about 27% of switchgrass GWP at the same N level. For these cellulosic systems, the emitted N_2O-N accounts for 1.38–1.68% of N applied, which is slightly higher than the IPCC reported default emission factors with a total of 1.325% (IPCC, 2006).

By relating GHG emissions to biofuel production, our model results show that, *Miscanthus* has much smaller global warming intensities than switchgrass, at all N application levels (Fig. 3b). Under currently available technologies, for each liter of ethanol produced, the *Miscanthus* system releases 21–36 g CO_2eq of GHG; with increasing N application, the GWPi also increases. Switchgrass, however, releases much more GHG per unit biofuel than *Miscanthus*, with lowest GWPi of about 100 g CO_2eq l^{-1} at N0 and N1 levels and highest GWPi of 390 g CO_2eq l^{-1} at N3 level. To produce the same amount of ethanol, the switchgrass systems on average release 4–10 times more GHG than *Miscanthus* systems. With advanced conversion technology, the GWPi can be lowered for both systems by reducing about 40% GHG release relative to current GWPi levels. But still, *Miscanthus* ecosystem has significantly lower GWPi than switchgrass ecosystem.

By considering the energy content in ethanol, which is 76 330 British thermal units per gallon (Btu Gal^{-1}) or 21.29 Megajoule per liter (MJ l^{-1}) (GREET, 2012), the global warming intensity can be translated into energy-based GHG emissions which can be further used to calculate carbon intensity in life cycle assessment (LCA) (Table 2). Depending on agricultural management and biofuel conversion efficiency, switchgrass could release 1.4–13.0 g CO_2eq for each MJ of energy produced. *Miscanthus*, however, has a relatively lower GHG intensity, ranging from 0.3 g CO_2eq MJ^{-1} under low N input and high biofuel conversion efficiency to 1.2 g CO_2eq MJ^{-1} under N3 input level and current biofuel conversion efficiency (Table 2). If we assume marginal land to be carbon neutral, the GHG emissions due to cropping of switchgrass and *Miscanthus* would result in a net carbon source in the marginal land. For switchgrass, especially when grown with high N input in the marginal land, the GHG intensity could exceed the earlier estimates for land use changes from cropland, grassland, and forest to cropping switchgrass (Dunn *et al.*, 2013; Elliott *et al.*, 2014) (Table 2).

Discussion

Cellulosic crops as biomass feedstocks

Cellulosic crops, such as switchgrass and *Miscanthus*, normally have higher nutrient-use efficiency

Table 2 Estimated GHG emissions of switchgrass- and *Miscanthus*-based ethanol in ecosystem modeling and life cycle assessment

References	Scenarios	Swtichgrass	*Miscanthus*
		g CO_2eq MJ^{-1}	
Ecosystem modeling			
This study*	N0 input (0 kg N ha^{-1})	1.4~3.5	0.3~0.7
	N1 input (50 kg N ha^{-1})	1.4~3.4	0.3~0.7
	N2 input (100 kg N ha^{-1})	2.6~6.2	0.3~0.8
	N3 input (150 kg N ha^{-1})	5.4~13.0	0.5~1.2
Life cycle assessment†			
Dunn *et al.*, 2013;	Domestic land use change‡	−3.9~13	−12~−3.8
	Total land use change¶	2.7~19	−10~−2.1
Elliott *et al.*, 2014;	Direct land use change‡	−0.13~0.21	0.89~2.35
	Total land use change¶	0.47~3.03	1.28~3.18

*The GHG emissions depend on technology levels, with a lower bound under advanced technology and an upper bound under current technology.

†LCA can estimate both domestic/direct and international/indirect land use change impacts on GHG emissions, and it considers all possible land conversion types.

‡Domestic/Direct land use change refers to conversions due to biofuel cropping within the United States. Cropland, grassland, and forest were normally considered for land use change.

¶Total land use change includes domestic/direct land use change, and international/indirect land use change which refers to land conversions occurred outside the United States because of biofuel cropping in the United States.

(Lewandowski *et al.*, 2003; Fargione *et al.*, 2010) and possibly higher water use efficiency than food crops (Stewart *et al.*, 2009; Zhuang *et al.*, 2013). They could therefore grow on marginal lands instead of competing with food crops for fertile croplands. However, the results here and elsewhere (Gelfand *et al.*, 2013) also show that biomass production from marginal lands may be lower than that from croplands. Our previous studies suggested that, an average of about 5–8 Mg DM ha^{-1} of switchgrass or around 20 Mg DM ha^{-1} of *Miscanthus* could be produced from cropland (Qin *et al.*, 2013b), which is higher than those grown on marginal lands even with high N input (Table 1). This may be partly because that besides nutrient (e.g. N) other factors could also affect biomass production on marginal lands, for example, water availability, climate conditions, and soil fertility (Cai *et al.*, 2011).

N application affects not only biomass production but also the ecosystem GHG emissions. One hand, use of N fertilizer could improve soil nutrient condition and therefore stimulate crop growth. With an increasing rate of N application, for each unit of N use, biomass production increment decreases gradually (Fig. 4a, c). Marginal HBIO production, the change in HBIO arising with each unit change in N input, d(HBIO)/d(N), decreases with N addition (Fig. 4b, d). On the other hand, increasing use of N leads to more N losses through gaseous emissions, leaching, and runoff. With increasing N application, the GHG release also increases (Fig. 4a, c), the marginal GHG emissions i.e. change of GHG arising with each unit change in N input, d(GHG)/d(N) increase with N addition (Fig. 4b, d). It is therefore very important to analyze how N use affects the benefits (e.g. biomass or biofuel production) and costs (e.g. GHG emissions) in marginal lands in our future studies.

Land use and GHG emissions regarding 2022 biofuel target

The Energy Independence and Security Act (US Congress, 2007) established a target of 136 billion liters (36 billion gallons) of renewable fuels in the United States by 2022, including 79 billion liters (21 billion gallons) of cellulosic ethanol. To reach the cellulosic ethanol target, a total of about 280 million tons of cellulosic biomass will be required under current biofuel conversion technology. If switchgrass was to be grown on marginal lands for biofuel feedstocks, a total of 48–81 Mha of land would be required (Fig. 5). According to estimates made by Cai *et al.* (2011), large areas of cropland or natural ecosystems might have to be sacrificed for this purpose. In addition, 8–31 Tg CO_2eq of GHG would be released due to cropping, depending on N input levels (Fig. 5). However, if *Miscanthus* were grown, a large quantity of land could be saved compared with growing switchgrass, only 16–28 Mha of available marginal

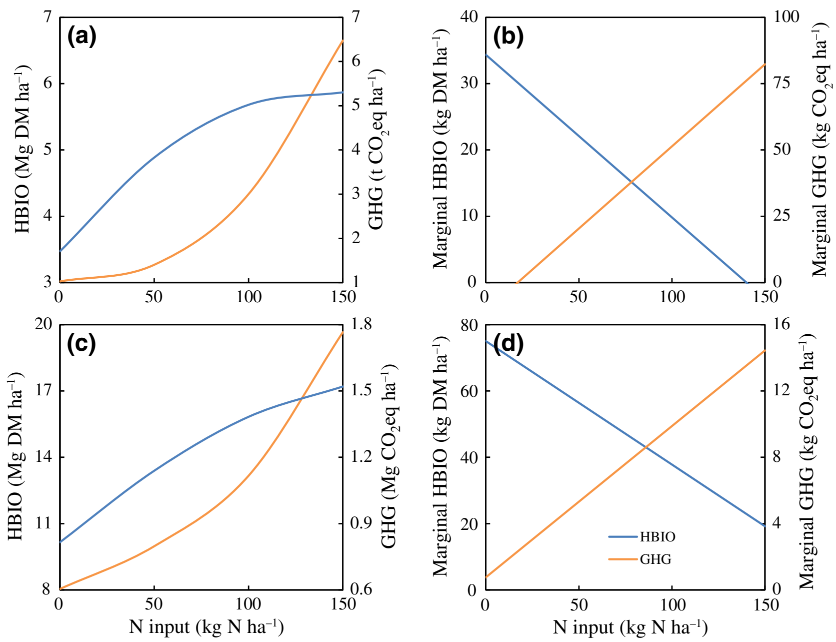

Fig. 4 Modeled change in HBIO and GHG from marginal lands with increasing use of N. Estimates for HBIO and GHG of (a) switchgrass and (b) *Miscanthus* were based on national average results (Table 1); marginal HBIO and marginal GHG of (c) switchgrass and (d) *Miscanthus* were based on polynomial (order 2) relationships between HBIO or GHG and N input.

 GCB Bioenergy, doi: 10.1111/gcbb.12212

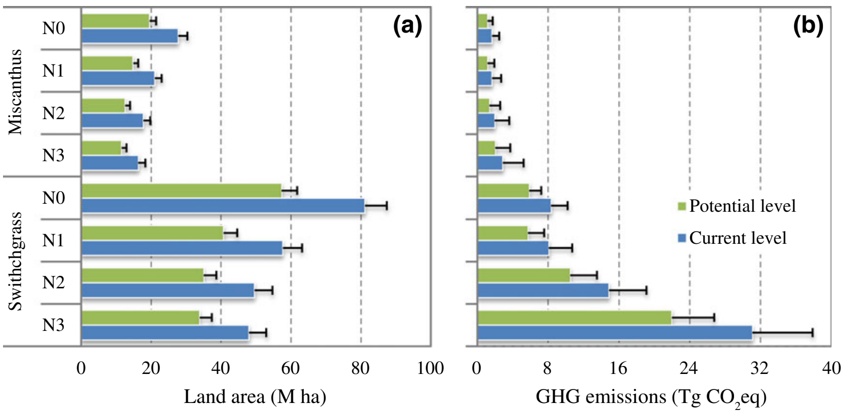

Fig. 5 Estimated demand of marginal lands and GHG emissions to achieve the 2022 biofuel mandate of 79 billion liters of cellulosic ethanol. Model estimates of (a) land demand and (b) GHG emissions were made for switchgrass and *Miscanthus* under current and potential biofuel conversion efficiencies.

lands could be sufficient to produce the required biofuel feedstocks. More importantly, using *Miscanthus* could reduce a considerable amount of GHG emissions; only a total of 1.7–2.9 Tg CO_2eq of GHG would be released from the ecosystem to meet the 2022 target (Fig. 5).

If biofuel conversion efficiency could be improved, i.e. from 282 to 399 l ethanol Mg DM (Lynd *et al.*, 2008), the biomass demand would be dramatically reduced to 200 million ton of dry matter. The land demand and GHG emissions could also be reduced to 71% of those under current technology, for both switchgrass and *Miscanthus* systems. Considering biofuel productivity alone, *Miscanthus* grown under N3 level has the highest land use efficiency. Under this scenario, only 11.6 Mha of marginal lands will serve the purpose of producing 79 million liters of ethanol (Fig. 5). However, if minimizing GHG emissions is the primary concern, then *Miscanthus* grown under the N0 level releases the smallest amount of GHG of just 1.2 Tg CO_2eq, but yet requires 19.6 Mha of land (Fig. 5).

By comparing with previous estimates for biomass produced from cropland (Qin *et al.*, 2012, 2013b), we find that, cellulosic crops have lower productivity grown on marginal lands, and therefore require relatively more land to reach the 2022 target, than if they were grown on fertile cropland. However, compared with maize grown on cropland (Qin *et al.*, 2012, 2013b), marginal land – based *Miscanthus* requires comparable or even less land resources and releases remarkably less amount of GHG, irrespective of N application and technology.

Limitations and future needs

Modeling studies are often limited by data availability and model deficiencies. In this study, data of climate, soil, and vegetation were used to initialize the model and make regional estimates. Most of these data (e.g. temperature, precipitation) are derived or reanalyzed from site/field observations, which inevitably introduce uncertainties into the spatially referenced model simulations due to observation errors, spatial heterogeneity, and possible interpretation biases (Huang *et al.*, 2009; Melillo *et al.*, 2009). In particular, due to lack of spatial data, the fertilization rate was assumed to be constant throughout the whole United States. Even with several different N rates (N0–N3), the fertilization scenario may not necessarily reflect real management practices, mainly because soil fertility is spatially heterogeneous and the fertilization rate can be adjusted accordingly. Also, the N fertilization impacts on N allocation (e.g. Guretzky *et al.*, 2011) and possible N fixation of certain crops (e.g. Davis *et al.*, 2010) are still open to discussion; our modeling experiments did not fully consider these issues due to insufficient mechanism understanding and data unavailability for certain cellulosic crop systems. Another major uncertainty regarding marginal land distribution should be further examined when data are available. This study did not consider crop-specific environmental constraints, such as possible water and temperature limitations, economic profitability, agronomic practicality as well as societal concerns about switchgrass and *Miscanthus*. It is very likely that certain regions may not be suitable for growing switchgrass or *Miscanthus* in the first place. Therefore, it is important to recognize that the land use scenarios considered here did not suggest actual land conversion practices. Crop-specific marginal land distribution and classification data should be developed to assist future modeling decisions with regard to land availability.

In addition, the AgTEM version used in this study did not specifically model the cropping impacts on other environmental factors, especially water quantity (Le *et al*., 2011) and quality (Ng *et al*., 2010). As observational and spatial data become available and our understanding regarding bioenergy ecosystems advances, we shall factor these components into AgTEM modeling and regional analysis with higher accuracy. Soil C fluxes were considered as part of the ecosystem C cycling but the SOC was not specifically reported in this study. When more observational data are available to support prediction of SOC under various types of marginal land, we shall further estimate SOC dynamics due to land use change.

As for cost-benefit analysis of energetic, environmental, and economic aspects regarding large-scale bioenergy development (e.g. Hill *et al*., 2006), LCA will be needed to account for energy system processes along with cellulosic ethanol's life 'from-cradle-to-grave' (e.g. Davis *et al*., 2009; Scown *et al*., 2012). The ecosystem analysis in this study is only one segment of the whole LCA chain, and estimates only those processes occurring inside specific ecosystems. As shown in Table 2, the estimated GHG intensity only assesses those GHG emissions during crop growth and harvest, irrespective of previous land use type. The LCA, however, has much broader system boundaries. Besides ecosystem, the LCA also assesses GHG emissions from other system processes, such as transportation, manufacturing, and biofuel use. Even for the assessment of GHG emissions in ecosystem, the LCA analyses could have their specific definitions and boundaries. For example, many LCA studies (e.g. Dunn *et al*., 2013; Elliott *et al*., 2014) included GHG emissions induced by land use change (e.g. converting cropland, grassland, or forest to bioenergy crops), which could assess possible C pool changes during land conversion. However, the LCA may not necessarily consider net change in C and/or N pools and fluxes in current (e.g. the estimates in this study) and previous ecosystems. Therefore, caution should be exercised when using the ecosystem modeling results in LCA assessment; especially the system boundary should be clarified to match ecosystem models with LCA processes.

Acknowledgement

The authors thank Dr. Jennifer Dunn and anonymous reviewers for insightful comments on an earlier version of this manuscript. Computing was supported by Rosen Center for Advanced Computing (RCAC) at Purdue University. This study is supported through projects funded by the NASA Land Use and Land Cover Change program (NASA-NNX09AI26G), Department of Energy (DE-FG02-08ER64599), the NSF Division of Information & Intelligent Systems (NSF-1028291), and the NSF Carbon and Water in the Earth Program (NSF-0630319).

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Supporting Information

Additional Supporting Information may be found in the online version of this article:

Figure S1. A schematic flow of carbon (C) and nitrogen (N) in the AgTEM 2.0. The AgTEM is based on the Terrestrial Ecosystem Model (TEM), and the C and N cycling in the model follows the general structure in the TEM (Raich *et al.*, 1991; McGuire *et al.*, 1992). Square blocks show state variables of C and N in vegetation and soils. Arrows indicate C and N fluxes; the dashed arrow shows C and N fluxes due to possible harvest (*H*). GPP, gross primary production; C_V, vegetation C; R_A, autotrophic respiration; L_C, C in litterfall; R_C, C in residue return; C_S, soil C; R_H, heterotrophic respiration; N_V, vegetation N; L_N, N in litterfall; RN, N in residue return; NS, soil N; N_{UPTAKE}, N uptake by vegetation; NETNMIN, net rate of soil N mineralization; N_{INPUT}, N inputs from outside ecosystem; N_{LOST}, N losses from ecosystem; NO_X, nitrogen oxides. More details about AgTEM can be found in Qin *et al.*, 2013a,b.

Figure S2. Area of marginal lands (%) capable of growing energy crops. Data were derived from Scenario 1 of Cai *et al.* (2011). Marginal lands were identified according to marginal agricultural productivity based on land suitability indicators such as topography, climate conditions, and soil productivity. Fuzzy Logic Modeling method was used to determine land productivity (Cai *et al.*, 2011).