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Impacts of land use change due to biofuel crops on carbon balance, bioenergy production, and agricultural yield, in the conterminous United States

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Abstract

Growing concerns about energy and the environment have led to worldwide use of bioenergy. Switching from food crops to biofuel crops is an option to meet the fast-growing need for biofuel feedstocks. This land use change consequently affects the ecosystem carbon balance. In this study, we used a biogeochemistry model, the Terrestrial Ecosystem Model, to evaluate the impacts of this change on the carbon balance, bioenergy production, and agricultural yield, assuming that several land use change scenarios from corn, soybean, and wheat to biofuel crops of switchgrass and *Miscanthus* will occur. We found that biofuel crops have much higher net primary production (NPP) than soybean and wheat crops. When food crops from current agricultural lands were changed to different biofuel crops, the national total NPP increased in all cases by a range of 0.14–0.88 Pg C yr^{-1}, except while switching from corn to switchgrass when a decrease of 14% was observed. *Miscanthus* is more productive than switchgrass, producing about 2.5 times the NPP of switchgrass. The net carbon loss ranges from 1.0 to 6.3 Tg C yr^{-1} if food crops are changed to switchgrass, and from 0.4 to 6.7 Tg C yr^{-1} if changed to *Miscanthus*. The largest loss was observed when soybean crops were replaced with biofuel crops. Soil organic carbon increased significantly when land use changed, reaching 100 Mg C ha^{-1} in biofuel crop ecosystems. When switching from food crops to *Miscanthus*, the per unit area croplands produced a larger amount of ethanol than that of original food crops. In comparison, the land use change from wheat to *Miscanthus* produced more biomass and sequestrated more carbon. Our study suggests that *Miscanthus* could better serve as an energy crop than food crops or switchgrass, considering both economic and environmental benefits.

Keywords: bioenergy, crop yield, net ecosystem production, net primary production, soil carbon, the Terrestrial Ecosystem Model

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Introduction

Increasing political, economic, and environmental security concerns are linked to energy dependence. Bioenergy is expected to play an important role in the global energy mix in the foreseeable future. Limited supply of conventional energy and rising fossil fuel carbon emissions have increased the need for renewable energy (Kim *et al*., 2009; Melillo *et al*., 2009). Bioenergy made available from materials derived from biologic sources meets the dual purpose of enhancing energy security and mitigating climate change, and is probably a reliable alternative to petroleum fuels (Kim *et al*., 2009; Beringer *et al*., 2011). A number of countries have set voluntary or mandatory biofuel targets for substituting petroleum fuels with biofuels (Fargione *et al*., 2010). Global biofuel production has increased dramatically in the last decade, especially in United States and Brazil (Carriquiry *et al*., 2011). For fuel ethanol in the United States, production increased from less than 2 billion gallons in the early 2000s to 4 billion gallons in 2005, and 13 billion gallons in 2010 (Renewable Fuels Association, 2011). According to the Energy Independence and Security Act of 2007 (The United States of America, 2007), the United States is expected to produce 36 billion gallons (136 billion liters) of renewable fuels by 2022, including 21 billion gallons (79 billion liters) of cellulosic ethanol.

However, the rapid growth of food-based biofuel is controversial, and issues of food security and potential ecological and environmental problems are often discussed. Currently, most biofuels are made from food crops like corn and soybeans. This raises major nutritional and ethical concerns, as growing crops for fuel consumes land, water, and energy resources that could

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otherwise be used in food production for humans (Pimentel *et al.*, 2010). In the United States, 13 million hectares of cropland is required to produce 37 million liters biofuel production in 2008 (Fargione *et al.*, 2010). That is about 7% of the 179 million hectares of national cropland (Lubowski *et al.*, 2006). To meet the 2022 biofuel mandate, another 15% of the cropland will be used in the United States for biofuel production. Producing biofuel from food threatens food security not only in the United States but also in other countries, especially countries that have populations suffering from malnourishment (Tilman *et al.*, 2009; Pimentel *et al.*, 2010). Studies show that conventional food-based biofuels contribute to monoculture and deforestation, which jeopardizes ecological biodiversity and ecosystem services (Fargione *et al.*, 2008; Searchinger *et al.*, 2008). Furthermore, some biofuels are not competitive with existing fossil fuels, and may result in negative energy return and net greenhouse gas (GHG) emissions (Searchinger *et al.*, 2008; Pimentel *et al.*, 2010). To date, conventional biofuels manufacture has important limitations (Evans, 2007) and the production may not be able to keep pace with fast-growing industry needs and energy demand (Hill *et al.*, 2006; Hurt *et al.*, 2006). These problems limit further development of conventional biofuels.

Second-generation biofuels are expected to help solve these problems, and to provide a considerable proportion of the fuel supply sustainably, affordably, and with greater environmental benefits. These biofuels are derived from lignocellulosic crops, and the biomass feedstock encompasses a wide range of sources such as food crop stalks, forest residues, and grass. Food crops like corn and soybeans will be consumed less for biofuel purposes. Tilman *et al.* (2006) reported that biofuels derived from well-managed energy crops provide more usable energy, higher GHG reductions, and less agrichemical pollution than conventional biofuels. Perennial energy crops like switchgrass require less water and nutrition input and produce much higher biomass per hectare for biofuel use than food crops (Hill *et al.*, 2006; Fargione *et al.*, 2010; Carriquiry *et al.*, 2011), and can be produced on degraded or abandoned agricultural land, eliminating the competition with food production for cropland, and without causing a loss of biodiversity via habitat destruction (Tilman *et al.*, 2006, 2009; Carriquiry *et al.*, 2011). Even though food-based biofuels currently eat up most of the bioenergy market, the majority of increased production after 2020 is expected to come from second-generation biofuels (IEA bioenergy, 2008; Fargione *et al.*, 2010). Furthermore, the technologies required to process cellulosic feedstocks into bioenergy are expected to be available within the next 10–20 years (Ragauskas *et al.*, 2006; Beringer *et al.*, 2011).

From the perspective of carbon mitigation, land use change for biofuel production may impact the ecosystem carbon balance both directly and indirectly. Soil and plant biomass are the two largest biologically active carbon stores in terrestrial ecosystems. Both land use and land cover (LULC) conversions will affect the carbon exchange in soil-plant-atmosphere systems. Switching from natural habitats like forests or grasslands to food crop-based biofuels creates a net 'biofuel carbon debt' by releasing more CO_2 than what is saved with the replacement of fossil fuels (Fargione *et al.*, 2008). However, making biofuels from perennial biomass grown on degraded or abandoned agricultural land seems to be applicable and beneficial to GHG reductions (Campbell *et al.*, 2008; Fargione *et al.*, 2008). While the debate continues, indirect land use change effects on carbon balance should not be neglected when accounting for GHG reduction (Fargione *et al.*, 2010). Possible deforestation and other indirect land use changes associated with biofuel production lead to uncertainties in GHG accounting (Searchinger *et al.*, 2008; Melillo *et al.*, 2009).

Considering the concerns of land source for biofuel production, crop switching from food crops to biofuel crops is one of the most promising options to meet the fast-growing need for biofuel feedstocks in the United States (Heaton *et al.*, 2008; Hoekman, 2009; Fargione *et al.*, 2010). Available lands for biofuel production are very limited; the direct sources can be classified into three categories: previously cropped land brought back into production, conversion of land used for other purposes to cropland, and crop switching (Fargione *et al.*, 2010). There are, however, many economic, social, or environmental risks involved in any of these three land sources. Cultivating the previously cropped land, the abandoned, or marginal land, for biomass production jeopardizes benefits for small-scale farmers and the rural poor. This may lead to social issues like the displacement of rural communities (Beringer *et al.*, 2011). Moreover, in the United States, available marginal land sources for biofuel production is not as encouraging when compared with other regions like Africa or China (Cai *et al.*, 2011). Converting natural ecosystems to biofuel crops will undoubtedly cause environmental problems, like increased GHG emissions and loss of biodiversity (Fargione *et al.*, 2008; Searchinger *et al.*, 2008; Melillo *et al.*, 2009; Fargione *et al.*, 2010). Crop switching from currently food-based crops to biofuels crops will directly reduce the land available for food production, and most probably lead to food reduction, discounting any agricultural, agronomical technology, or management improvement. To achieve the future bioenergy goal, any of these land sources could potentially contribute to additional biofuel production and subsequent environmental effects.

To date, many studies have investigated the direct and indirect effects of land use changes such as bringing previously cropped land back into production and converting land currently used for other purposes to cropland (Fargione *et al*., 2008; Searchinger *et al*., 2008; Melillo *et al*., 2009; Lapola *et al*., 2010). However, the effects of land use change associated with possible crop switching options for biofuel production on the ecosystem carbon balance has been less addressed (Fargione *et al*., 2010). Corn, soybeans, and wheat are three major crops with the highest production among food crops in United States, with about 220, 63, and 64 million tons per year (of the 1990s), respectively (FAOSTAT, 2011). Switchgrass (*Panicum virgatum*) and *Miscanthus* (e.g. *Miscanthus giganteus*) are two important second-generation biofuel crops with the potential to produce a large amount of biofuel feedstocks and mitigate carbon emissions (Parrish & Fike, 2005; Clifton-Brown *et al*., 2007; Tilman *et al*., 2009; Fargione *et al*., 2010; Pimentel *et al*., 2010), and they are expected to be used to meet the US bioenergy goals with less land compared with food crops (McLaughlin & Kszos, 2005; Heaton *et al*., 2008). Crop switching from corn, soybeans, or wheat to switchgrass or *Miscanthus* is foreseeable for second-generation biofuel production in the next several decades. The consequent land use change due to crop switching will enormously impact the ecosystem carbon dynamics, as well as change the regional carbon fluxes and pool sizes, which are essential for GHG estimations (Fargione *et al*., 2010).

In this study, using a biogeochemical model, the Terrestrial Ecosystem Model (TEM; Zhuang *et al*., 2003), we quantified the carbon balance due to the direct land use change associated with biofuel production in the conterminous US carbon fluxes, and pool sizes were evaluated under the land use change scenarios of LULC change from food crops (corn, soybeans, and wheat) to biofuel crops (switchgrass and *Miscanthus*). We further discussed the energy production, food yield, and land replacement associated with these land use changes.

Methods

Overview

Terrestrial Ecosystem Model was parameterized to quantitate the carbon dynamics in the agroecosystems of corn, soybeans, wheat, switchgrass, and *Miscanthus*. Regional land use change effects on biomass production, soil carbon, and ecosystem carbon balance were then estimated for the conterminous United States, followed by further discussion of land use change effects on bioenergy production, crop yield, and land use.

TEM description and parameterization

Terrestrial Ecosystem Model is a process-based global-scale ecosystem model that estimates carbon (C) and nitrogen (N) fluxes and pool sizes in terrestrial ecosystems at a monthly time step using spatial climate and ecological data (Raich *et al*., 1991; McGuire *et al*., 1992). TEM has been well-documented in simulating the C dynamics in natural ecosystems at regional and global scales (e.g. McGuire *et al*., 2001; Lu *et al*., 2009; Lu & Zhuang, 2010). TEM consists of two major C pools (vegetation carbon and soil carbon) to simulate several C fluxes as gross primary production (GPP), net primary production (NPP), net ecosystem production (NEP), autotrophic respiration (R_A), and heterotrophic respiration (R_H). In this study, TEM was coupled with freezing and thawing dynamics (Zhuang *et al*., 2003). Parameters in TEM are constant, vegetation-specific, or soil-specific, and most of them have been well defined in previous studies (e.g. Raich *et al*., 1991; McGuire *et al*., 1992; Zhuang *et al*., 2003), but some still need to be determined via calibration using site-level data of climate and ecosystem data. Previous studies using TEM mostly focused on natural ecosystems like forests or grasslands, and paid less attention to managed agricultural ecosystems. In this study, we parameterized TEM for food crop and biofuel crop ecosystems.

Climate data, together with the observed data of C and N fluxes and pool sizes, was used to calibrate the model. The monthly air temperature, precipitation, and cloudiness in the 1990s, obtained from the Climate Research Unit (CRU; Mitchell & Jones, 2005), were used for the corresponding sites. C and N fluxes and pool sizes used for corn, soybeans, wheat, *Miscanthus*, and switchgrass were documented in Table 1. TEM was calibrated for each crop to obtain the values for each limiting

Table 1 Data used to calibrate the Terrestrial Ecosystem Model for food crop and biofuel crop ecosystems*

	Corn	Soybean	Wheat	*Miscanthus*	Switchgrass
GPP	1500	550	650	3500	1200
NPP	690	270	320	1700	590
C_V	760	300	350	1900	650
N_V	21	8	10	48	28
C_S	5800	6700	5900	11200	10500
N_S	530	460	530	1000	880
N_{AV}	2.5	3.5	2.5	2.5	2.5
NPPSAT	1200	340	500	2400	1000

References and data sources: Hicke *et al*. (2004), Lu & Zhuang (2010), Qin & Huang (2010) and FAOSTAT (2011) were referenced for food crop (corn, soybean and wheat) ecosystems; Kahle *et al*. (2001), Heaton *et al*. (2004), Thomason *et al*. (2004), Liebig *et al*. (2008) and Schmer *et al*. (2011) were referenced for biofuel crop (*Miscanthus* and switchgrass) ecosystems.

*Units for annual gross primary production (GPP), net primary production (NPP), and saturation response of NPP to nitrogen fertilization (NPPSAT) are g C m^{-2} yr^{-1}. Units for vegetation carbon (C_V) and soil carbon (C_S) are g C m^{-2}. Units for vegetation nitrogen (N_V), soil N (N_S), and available inorganic N (N_{AV}) are g N m^{-2}.

parameter. Details of calibration protocols can be found in previous studies (e.g. McGuire *et al.*, 1992; Zhuang *et al.*, 2003; Lu *et al.*, 2009).

Some calculations and derivations were made from observational data in this study. Following Hicke *et al.* (2004) and Monfreda *et al.* (2008), food crop NPP was derived from the corresponding crop economic yield according to Eqn (1):

$$\mathrm{NPP}_i = \frac{\mathrm{EY}_i \times D_i \times C \times (\mathrm{RS}_i + 1)}{\mathrm{HI}_i}, \quad (1)$$

where, i is the specific food crop, either corn, soybeans, and wheat, EY is the economic yield, and NPP is the net primary production. HI refers to harvest index, which measures the proportion of total aboveground biologic yield allocated to the economic yield of the crop. D is the dry proportion of the EY, and C is the carbon content in the dry matter (usually $C = 0.45$). RS is the root to shoot ratio, which indicates the ratio of below to aboveground biomass. Even though this conversion is widely used, it still has some limitations in the way values are chosen for the parameters for each crop (Hicke *et al.*, 2004). For the instances where soil carbon in the top soil layer (0–100 cm) is not available, it is calculated from the soil carbon density at different depths according to the vertical distribution (cm) (Jobbágy & Jackson, 2000; Wang *et al.*, 2004; Qin & Huang, 2010):

$$\mathrm{SOC}_{0-20} : \mathrm{SOC}_{20-40} : \mathrm{SOC}_{40-60} = 41 : 64 : 100, \quad (2)$$

where SOC is the soil organic carbon. The subscripts indicate soil organic carbon at different depths.

Regional simulations

With spatially referenced information on climate, elevation, soil, and vegetation, the calibrated TEM was extrapolated to estimate C balance in the conterminous US carbon fluxes, and pool sizes were estimated separately for different LULC scenarios. By assuming that LULC changes from the food crops corn, soybeans, and wheat to the biofuel crops, switchgrass, and *Miscanthus*, in total, six land use change scenarios and nine simulations are included in the simulations. The land use change scenarios are namely corn to switchgrass, corn to *Miscanthus*, soybeans to switchgrass, soybeans to *Miscanthus*, wheat to switchgrass, and wheat to *Miscanthus*. The nine simulations include six for land use change scenarios plus three more for separate crops.

For regional estimation, we first ran TEM to estimate C dynamics at a grid cell level at a monthly time step from 1990 to 1999 (see protocols in McGuire *et al.*, 1992; Lu *et al.*, 2009). We then calculated the annual C fluxes and pool sizes for the whole United States. Each grid cell in TEM was assigned a certain crop ecosystem type according to the vegetation data, and calculated separately for each crop ecosystems (corn, soybeans, wheat, switchgrass, and *Miscanthus*). The regional statistics were based on the crop harvest area such that crop rotations can be considered in the estimation. C fluxes and pool sizes were estimated separately to quantitate C balance for land use change. The estimations were averaged over the 10 years of the 1990s, to alleviate the annual variations, and to be in accordance with the crop distribution information.

Spatial climate, elevation, soil, and vegetation data used in TEM were organized at a 15′ latitude × 15′ longitude (about 25 × 25 km) resolution for the conterminous United States. Specifically, the driving climate datasets, including the monthly air temperature, precipitation, and cloudiness, averaged from 1900 to 1999 based on CRU (Mitchell & Jones, 2005). For elevation, data were derived from the Shuttle Radar Topography Mission (Farr *et al.*, 2007) and resampled to the given resolution (Lu *et al.*, 2009). The 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). For each food crop, specific vegetation data describing crop distribution is used for the TEM simulation, according to the TEM vegetation classification scheme (Melillo *et al.*, 1993). The vegetation data for the conterminous United States were excerpted from a global crop harvest area database (Monfreda *et al.*, 2008). The database contains crop distribution information of year 2000 for 175 distinct crops from around the world, with a 5′ × 5′ spatial resolution in latitude by longitude (Monfreda *et al.*, 2008). Grid cells with no less than 1% crop cover by area are regarded as cropland for each crop of corn, soybeans, or wheat. The cropland data were then resampled to a specific resolution (15′ latitude × 15′ longitude) in TEM. The harvest area determined for use in this study for corn, soybeans, and wheat in the conterminous United States are 30.9, 31.2, and 23.8 M ha, respectively, (Fig. 1).

Evaluation of national energy, food and carbon sequestration

Land use change due to crop switching from food crops to biofuel crops will directly affect bioenergy production, food crop yield, and CO_2 emissions. To reach the goal of productive energy and food security, we expect to have high bioenergy production and little agricultural yield loss. We also expect less GHG emissions for environmental concern. To evaluate the balance among the trilemma of energy, food, and carbon, several variables were selected at a national scale. Biomass quantity, specifically NPP in this study, is an indicator of food or biofuel production. The food crop yield and ethanol production can be converted from crop NPP as determined using TEM. CO_2 emissions are expressed as NEP, and as soil carbon sequestration rate (SOCR) for additional carbon added into the soil as a result of land use change. A positive NEP represents annual net carbon sequestration by crop ecosystems, whereas a negative value represents that crop ecosystems are losing carbon.

Food crop yield is derived from NPP by reversing Eqn (1):

$$\mathrm{EY}_i = \frac{\mathrm{NPP}_i \times \mathrm{HI}_i}{D_i \times C \times (\mathrm{RS}_i + 1)}, \quad (3)$$

where, i is the specific food crop of corn, soybeans, or wheat, EY is the economic yield, and NPP is the net primary production. Other variables are the same as in Eqn (1). The soil organic carbon sequestration rate is used to depict the amount of soil carbon accumulated in soil annually, calculated by

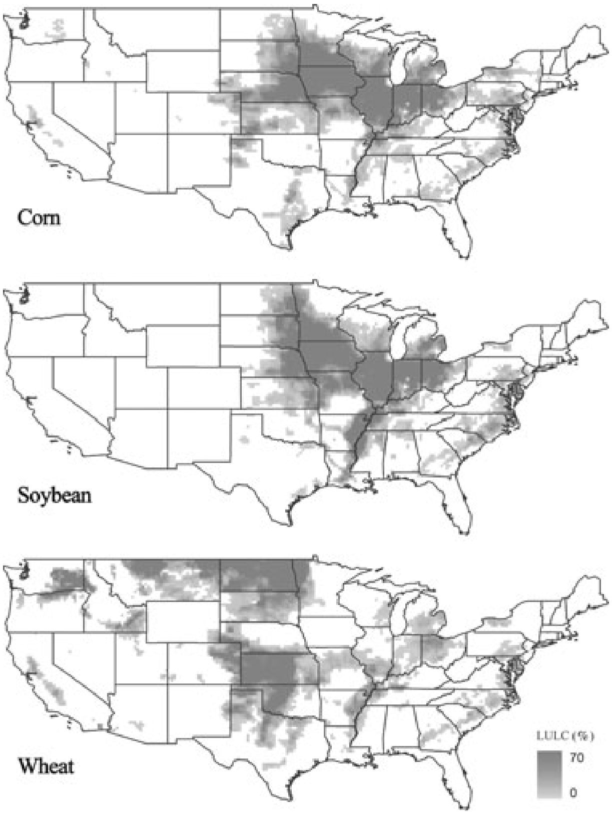

Fig. 1 Cropland distribution of corn, soybean, and wheat in the conterminous United States. Maps show the harvested area of each crop as the proportion of each grid cell. Data are derived from Monfreda *et al.* (2008).

assuming that additional SOC can be stored for a certain sequestration time (West & Six, 2007; Qin & Huang, 2010).

Results

Net primary production as a measure of biofuel feedstocks

In general, land use change from food crops to biofuel crops produce more NPP over the whole cropland in the region (Fig. 2). The NPP is weighted by crop harvest area (Monfreda *et al.*, 2008) for each grid cell. As most food crops are distributed in the Midwest, NPP patterns are similar to that of crop distribution (Fig. 1). For food crops, corn has much higher annual NPP than soybean and wheat, especially in regions with large areas of corn production, inclduing most of Iowa, Illinois, and Indiana. Most of the NPP from corn, soybeans, and wheat is intensively distributed in the Midwest region around Lake Michigan (Fig. 2). The model estimates are similar to the agricultural harvest yield data (Prince *et al.*, 2001; Monfreda *et al.*, 2008). Replacing food crops with the biofuel crops of switchgrass and *Miscanthus* provides more NPP. Specifically, switchgrass NPP is similar to that of corn, but higher than that of soybeans and wheat. *Miscanthus* has the highest NPP of all (Fig. 2), and produces about 2.5 times the NPP of switchgrass.

Nationally, biofuel crops also have higher average NPP than food crops. TEM estimates the annual NPP to be 462 g C m^{-2} yr^{-1}, or 0.4 Pg C yr^{-1}, for corn, soybeans, and wheat in total. This is close to the census NPP data of about 500 g C m^{-2} yr^{-1} in the 1990s (Hicke & Lobell, 2004). Among the food crop ecosystems, corn has the highest NPP, either in the form of harvest area averaged annual NPP, or the national total NPP, whereas wheat has the lowest national total NPP, as it has the lowest annual average NPP and smallest crop area (Table 2). Switchgrass and *Miscanthus* have an estimated average annual NPP of 620 and 1500 g C m^{-2} yr^{-1}, respectively. These values are in the range of NPP determined from the site-level observed yield (Heaton *et al.*, 2004; Heaton *et al.*, 2008; Fargione *et al.*, 2010; Schmer *et al.*, 2010). Due to land use change, the national total NPP increased by 0.14–0.88 Pg C yr^{-1} over all croplands. When switching from food crops to switchgrass, the total NPP decreased by 14% in corn, but increased by 82% and 100% in soybeans and wheat, respectively. Converting from food crops to *Miscanthus* increased the total NPP from 129% to 147% (Table 2).

Ecosystem carbon balance

Soybean and wheat ecosystems have relatively higher NEP than corn ecosystems, and have larger areas acting as C sinks. Over the food croplands, the NEP ranged mostly from −10 to 10 g C m^{-2} yr^{-1} (Fig. 3a, d, g), with a national average NEP of 0.5, 2.6, and −6.7 g C m^{-2} yr^{-1} in soybean, wheat, and corn ecosystems, respectively (Table 3). Spatially, soybean and wheat ecosystems have about half the number of grid cells acting as a C sinks, while this ratio falls to one-third for corn ecosystems. For the three crops together, the croplands act as a net C source, with a national total NEP of −1.3 Tg C yr^{-1} (Fig. 3j, Table 3).

Under LULC changes, NEP went from positive to negative for most regions. The NEP of switchgrass (Fig. 3b, e, h) and *Miscanthus* ecosystems have higher variations than those of food crop ecosystems (Fig. 3c,f, i). The soybean regions shift from a mild C sink to a C source with an annual C balance of −20 g C m^{-2} yr^{-1} in both switchgrass and *Miscanthus* ecosystems (Table 3). The corn regions become a larger C source, and the wheat regions become a net C source when replaced with switchgrass and a smaller C sink with *Miscanthus* (Table 3). Nationally, land use change causes a net carbon loss of 1.0–6.3 Tg C yr^{-1} when cropland is

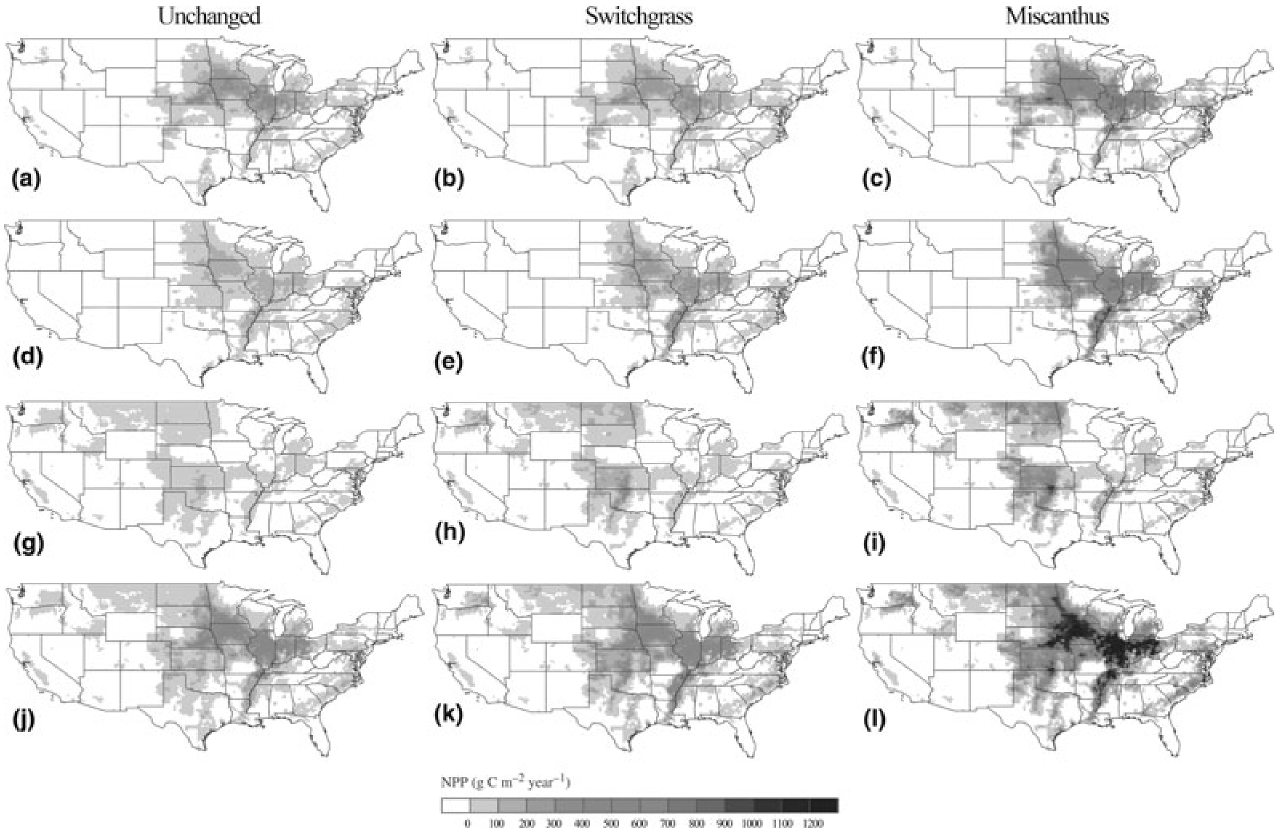

Fig. 2 Mean annual net primary production under different land use change scenarios, as determined using Terrestrial Ecosystem Model. Vegetation change from food crops (first column) to biofuel crops (*Miscanthus* and switchgrass); food crops (from top to bottom) are (a) corn, (d) soybean, (g) wheat, and (j) above three crops together.

Table 2 Mean annual net primary production before and after vegetation changed from food crops to biofuel crops over the conterminous United States

	National average (g C m^{-2} yr^{-1})			National total (Tg C yr^{-1})		
Original crops	Unchanged	Switchgrass	*Miscanthus*	Unchanged	Switchgrass	*Miscanthus*
Corn	713.0	622.2	1513.4	220.3	192.3	467.6
Soybean	355.7	648.2	1567.5	110.9	202.1	488.6
Wheat	276.2	594.5	1354.9	65.7	141.4	322.3
All	462.3	624.0	1489.1	396.9	535.8	1278.6

changed to switchgrass, and of 0.4–6.7 Tg C yr^{-1} if it is changed to *Miscanthus*. The greatest carbon loss results when soybeans are replaced with biofuel crops (Table 3). High heterotrophic respiration in biofuel crop ecosystems is the main contributor to the carbon loss experienced under these land use change scenarios. Corn ecosystems have much higher R_H than soybean and wheat ecosystems. Switchgrass has a similar amount of national average R_H as corn under all three land use change scenarios. *Miscanthus* has the highest R_H among all crops (Table 3). The higher heterotrophic respiration of biofuel crop ecosystems results in relatively lower NEP than that for food crop ecosystems. R_H is higher than corresponding NPP in both switchgrass and *Miscanthus* ecosystems, leading to these ecosystems to become greater carbon sources (Table 3). Previous studies also observed that biofuel crop ecosystems have relatively higher soil respiration than food crop ecosystems (Tufekcioglu *et al.*, 1999; Raich & Tufekcioglu, 2000), and that a greater R_H than NPP value results in a net C source for the biofuel crop ecosystems (Yazaki *et al.*, 2004).

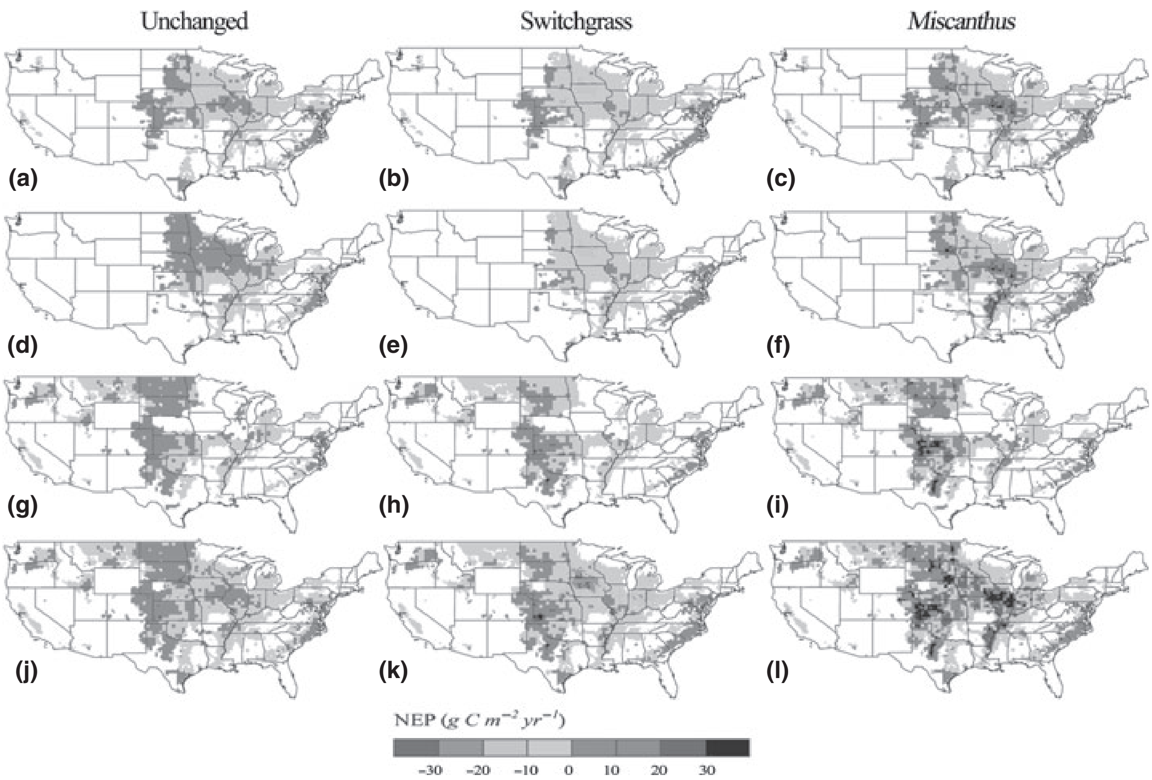

Fig. 3 Mean annual net ecosystem production under different land use change scenarios, as determined using Terrestrial Ecosystem Model. Vegetation distribution is same as in Fig. 1.

Table 3 Carbon fluxes before and after vegetation changed from food crops to biofuel crops over the conterminous United States

	National average (g C m^{-2} yr^{-1})			National total (Tg C yr^{-1})		
Original crops	Unchanged	Switchgrass	*Miscanthus*	Unchanged	Switchgrass	*Miscanthus*
Net ecosystem production						
Corn	−6.7	−15.3	−13.2	−2.1	−4.7	−4.1
Soybean	0.5	−19.6	−20.9	0.2	−6.1	−6.5
Wheat	2.6	−1.8	1.0	0.6	−0.4	0.2
Heterotrophic respiration (R_H)						
Corn	719.7	637.4	1526.5	222.4	197.0	471.7
Soybean	355.2	667.9	1588.4	110.7	208.2	495.1
Wheat	273.7	596.3	1353.9	65.1	141.9	322.1

Carbon pools in vegetation and soils

Biofuel crops have much greater carbon storage in vegetation than food crops. Crops are harvested every year, and the vegetation C pool size is almost the same as the annual NPP in agroecosystems (Lu & Zhuang, 2010). For biofuel crops, the C stored in vegetation is generally much higher than that of food crops. Specifically, switchgrass has 0.9, 1.8, and 2.1 times national average vegetation C when compared with corn, soybeans, and wheat, respectively. *Miscanthus* has even larger vegetation C pools, which are about twice that of corn, and about five times that of soybeans or wheat (Fig. 4a).

The soil organic carbon content within 1 m soil depth is 60 Mg C ha^{-1} in food crop ecosystems, without significant differences among corn, soybeans, and wheat, whereas the biofuel crop ecosystems of switchgrass and *Miscanthus* accumulate above 100 Mg C ha^{-1} in soils (Fig. 4b), which is almost double that of food crop ecosystems, and close to that of natural forest ecosystems (Heath *et al*., 2002). As a result of land use change from food crops to biofuel crops, the national total soil organic carbon increases by 1.0–1.6 Pg C in the conterminous United States, if food crops are all replaced by biofuel crops. The soil carbon of biofuel ecosystems is 13–18% of the total soil carbon storage in

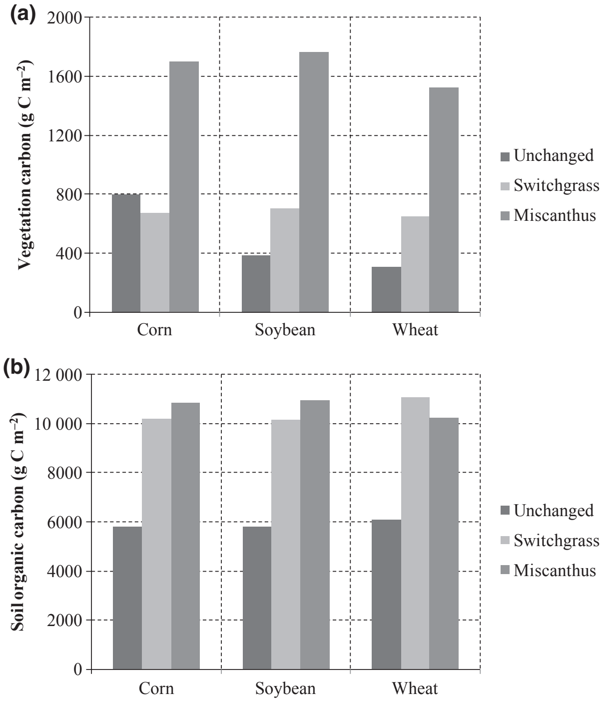

Fig. 4 Carbon pools in vegetation and soils for food crop and biofuel crop ecosystems. Unchanged are food crops without land use change. Values are harvested area weighted national average C pool sizes in vegetation (a) and soil (b).

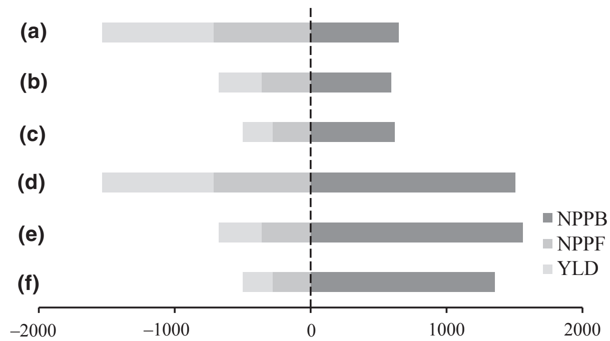

Fig. 5 Annual balance among biomass production and food yield under land use change scenarios, as determined using Terrestrial Ecosystem Model. NPPB, NPP of biofuel crops (g C m^{-2} yr^{-1}); NPPF, negative value of NPP of food crops (g C m^{-2} yr^{-1}); YLD, negative value of food crop yield (g m^{-2} yr^{-1}). Land-use change scenarios are from (a) corn to switchgrass, (b) soybean to switchgrass, (c) wheat to switchgrass, (d) corn to *Miscanthus*, (e) soybean to *Miscanthus*, and (f) wheat to *Miscanthus*.

forests of the conterminous United States (Turner *et al.*, 1995).

Discussion

Advantages of growing biofuel crops

Biofuel crops, especially *Miscanthus,* have much higher potential biomass production than food crops, and could provide a larger amount of feedstock for energy use. One factor that contributes to the high productivity of biofuel crops is high solar radiation interception and conversion efficiencies. According to the field experiments in Illinois, canopies of mature switchgrass and *Miscanthus* could intercept about 70% of the photosynthetically active radiation (PAR), and with PAR conversion efficiencies of about 1% and 3%, respectively (Heaton *et al.*, 2008). Also, owing to their low-nutrient requirements and high water use efficiencies (Stewart *et al.*, 2009; Fargione *et al.*, 2010), switchgrass and *Miscanthus* are capable of growing on sterile soils where food crops cannot survive. There are numerous reports suggesting that switchgrass and *Miscanthus* have no response to N fertilization, or only to the first 50–70 kg ha^{-1} (Lewandowski *et al.*, 2003), whereas corn responds to double or even triple the N demand (Fargione *et al.*, 2010). Water used for irrigating crops or processing biomass is far less for switchgrass and *Miscanthus* than for food crops (Fargione *et al.*, 2010). Biofuel crops are also advantageous because they can use abandoned or degraded agricultural lands to produce bioenergy, leaving more croplands available for food crops (Fargione *et al.*, 2010; Cai *et al.*, 2011).

Implications to biofuel feedstocks and agricultural yield

Under different land use change scenarios, biofuel crops produce relatively higher biomass than food crops (Fig. 5). In particular, *Miscanthus* provides significantly higher NPP than switchgrass, and even higher NPP than food crops like soybean and wheat. However, there is a net loss of food crop yield, because biofuel crops provide nothing but biomass feedstocks (Fig. 5). As determined by TEM, corn loses the highest amount of yield of 820 g m^{-2}, whereas wheat loses the lowest yield of 220 g m^{-2} (Table 4). The estimated yield loss is comparable with the FAO statistical data (Table 4). Among the six land use change scenarios, food crops changed to *Miscanthus* have a much greater net biomass gain than land changed to switchgrass. Specifically, crop switching from soybeans to *Miscanthus* gains the highest biomass of 1200 g C m^{-2}, whereas corn switching to switchgrass gains the lowest at -90 g C m^{-2}. Corn to switchgrass is the only scenario that causes a net biomass loss.

In terms of biomass production, wheat and soybeans rather than corn should be preferred for biofuel crop displacement. For the same amount of switchgrass or

Table 4 Mean annual food crop economic yield as determined by statistical data and Terrestrial Ecosystem Model (TEM) estimation for the nationwide United States

	D	HI	RS	Statistical yield (g m^{-2})	Estimated yield (g m^{-2})	Ratio of modeled to statistical yield
Corn	0.871	0.53	0.18	774	818	1.06
Soybean	0.920	0.42	0.15	247	314	1.27
Wheat	0.894	0.39	0.20	260	223	0.86

D is the dry proportion of the economic yield, HI is crop harvest index, and RS is the root to shoot ratio of each crop; data based on Prince *et al.* (2001) and Hicke & Lobell (2004). Statistical crop yield are derived from FAO statistical data (FAOSTAT, 2011), and estimated crop yield are calculated from TEM modeled NPP of the 1990s according to Eqn (1).

Miscanthus biomass, there will be less food yield loss in wheat and soybean than in corn (Fig. 5). The land use change scenario of corn to switchgrass should be considered last for crop switching for biofuel feedstocks.

Implications to bioenergy and land use

Higher biomass production does not necessarily mean higher energy production, as food crops have relatively higher energy conversion efficiencies (Fargione *et al.*, 2010). Using aboveground-to-belowground biomass ratios of 1.4 and 2.5 for switchgrass and *Miscanthus*, respectively (Meyer *et al.*, 2010), and assuming that 90% aboveground biomass is harvested, or simply using 1.17 and 1.43 times NPP (in the form of carbon) as harvestable biomass (in the form of biomass) for switchgrass and *Miscanthus*, respectively, we estimated the biofuel crops' harvestable biomass (Table 5). A unique value of 0.87 is used as the dry proportion of the yield for both corn and biofuel crops (Hicke & Lobell, 2004). The conversion efficiency of dry mass to ethanol is 416 L Mg^{-1} (110 Gal Mg^{-1}) for corn, and 380 L Mg^{-1} (100 Gal Mg^{-1}) for biomass (DOE, 2006). Soybeans and wheat are not included for ethanol production comparison, as they both produce much lower biomass than corn, and offer no superior energy conversion efficiencies.

For the same cropland area, *Miscanthus* provides the highest ethanol production among three potential bioenergy feedstocks, whereas switchgrass provides the lowest, and corn is in between. Ethanol production using corn doubles that of switchgrass with almost the same amount of total biomass (Table 5). This can be explained by the high-energy conversion efficiency of corn grain, and the low harvestable biomass of switchgrass. *Miscanthus*, owing to its high biomass productivity, triples the bioenergy of switchgrass, with the higher ethanol productivity of 2400 gal ha^{-1} (Table 5). Compared with results derived from site-level observations, the ethanol production of *Miscanthus* is reasonably lower due to the nationwide averaged biomass production, but still comparable with Heaton *et al.* (2008).

To reach the goal of ethanol production in 2022, it is important to consider that *Miscanthus* consumes much less land than corn, and especially less than switchgrass. Assuming all cellulosic ethanol comes from a sole crop, switchgrass needs 25 M ha land to produce 21 billion gallons of ethanol. This is equivalent to the current wheat harvest area in the United States. However, corn needs only 12 M ha land, meaning that about 40% of current corn fields could be enough to yield the same ethanol production. *Miscanthus* needs even less land for the same purpose, about a quarter of the current corn

Table 5 Harvestable biomass production, potential ethanol production, and land needed for different bioenergy feedstocks to reach 21 billion gallon cellulosic ethanol goal in 2022

Feedstock	Net primary production (g C m^{-2})	Harvestable biomass (Mg ha^{-1})	Ethanol production (gal ha^{-1})*	Land needed for ethanol (M ha)†	Harvested US cropland in 2000 (%)‡
Corn grain		8.2	904	23.2	18.3
Corn stover		9.0	901	23.3	18.4
Corn total	713	17.2	1805	11.6	9.2
Switchgrass	624	8.4	839	25.0	19.7
Miscanthus	1489	24.5	2447	8.6	6.8

*Calculated according to conversion efficiency of DOE (2006).
†Cropland area needed to produce 21 billion gallon ethanol in 2022.
‡Percentage of land needed to harvested US cropland in 2000 (U.S.DA, 2007).

fields, a quarter of the current soybean fields, or one-third of the current wheat fields (Table 5).

Studies show that bioethanol and biodiesel production results in a positive net energy balance when taking energy use efficiency into consideration (Farrell *et al.*, 2006; Hill *et al.*, 2006). An energy output: input ratio of 8 : 51 was observed in cellulosic crop-based ethanol production (Lewandowski & Schmidt, 2006; Scheffran & BenDor, 2009), whereas only 4 : 14 was observed in cereal crop-based ethanol production (Franzluebbers & Francis, 1995; Mandal *et al.*, 2002; Rathke *et al.*, 2007). Cellulosic crops, especially *Miscanthus* could maximize energy use efficiency and land use efficiency by replacing food crops.

Implications to carbon sequestration

Switching from food crops to biofuel crops decreases annual NEP, but increases soil carbon. For all land use change scenarios, the NEP differences are all negative, meaning a net loss of carbon at the ecosystem level (Fig. 6). Specifically, switching from soybeans to biofuel crops causes the highest ecosystem carbon loss of 20 g C m^{-2} yr^{-1}, whereas the lowest loss is observed when switching from wheat to biofuels (Fig. 6). Assuming that after the land use change, the biofuel crop ecosystems can sequestrate extra SOC for 50 years (West & Six, 2007; Qin & Huang, 2010), the SOC sequestration rate will be 80 g C m^{-2} yr^{-1}, which is even higher than that of recovering forest or grassland from agricultural lands (Post & Kwon, 2000). Our results fall into the range of SOC sequestration rates estimated elsewhere, which go from 60 (Clifton-Brown *et al.*, 2007) to more than 100 g C m^{-2} yr^{-1} (Hansen *et al.*, 2004). We estimate that when changing LULC from food crops to biofuel crops, soils will sequestrate additional CO_2, and that the amount of additional SOC will be much higher than that of the net ecosystem carbon loss due to land use change (Fig. 6). Adding up both effects of the ecosystem carbon change and soil carbon addition, the net carbon balance due to land use change will range from 67 g C m^{-2} yr^{-1} of the land use change scenario from soybeans to switchgrass (Fig. 6b), to 95 g C m^{-2} yr^{-1} of the land use change scenario from wheat to switchgrass (Fig. 6c).

Soil carbon accumulation involved in land use change is not typically included in life-cycle analyses, but has a tremendous impact on carbon balance (Anderson-Teixeira *et al.*, 2009). SOC sequestration by biofuel ecosystems, as shown here and elsewhere (e.g. Liebig *et al.*, 2005; Liebig *et al.*, 2008; Anderson-Teixeira *et al.*, 2009; Clifton-Brown *et al.*, 2007), could substantially augment the GHG mitigation benefits due to crop switching for biofuel purposes. For example, for the United States to reach its ethanol goal in 2022 by growing *Miscanthus*, 8.6 M ha of land will be required (Table 5), and about 8 Tg C yr^{-1} carbon will be sequestrated into soils, that is, one-third higher than the soil carbon accumulated due to afforestation, or two-thirds of carbon emissions caused by deforestation in the conterminous United States from 1990 to 2004 (Woodbury *et al.*, 2007).

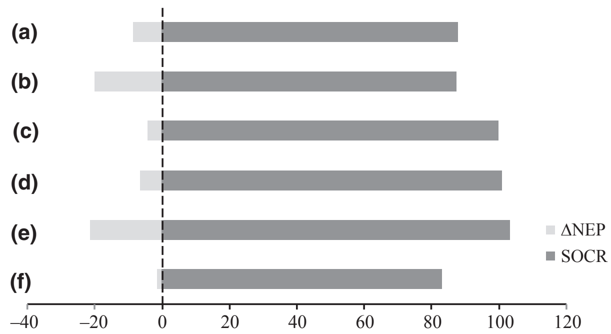

Fig. 6 Net ecosystem production and soil organic carbon change due to land use change, as determined using Terrestrial Ecosystem Model. ΔNEP, NEP difference between biofuel crop ecosystem and corresponding food crop ecosystem under land use change (g C m^{-2} yr^{-1}); SOCR, additional soil organic carbon sequestration rate due to land use change from food crop ecosystems to biofuel crop ecosystems (g C m^{-2} yr^{-1}). Land use change scenarios are from (a) corn to switchgrass, (b) soybean to switchgrass, (c) wheat to switchgrass, (d) corn to *Miscanthus*, (e) soybean to *Miscanthus*, and (f) wheat to *Miscanthus*.

Conclusions and future research

Among the six land use change scenarios considered, crop switching from wheat to *Miscanthus* is the favorite in terms of energy production, agricultural yield, and CO_2 reduction. *Miscanthus* is a promising energy crop, and could provide a considerable amount of biomass for biofuel production. *Miscanthus* produces more ethanol on a per unit area basis than corn or switchgrass, with only moderate carbon emissions at an ecosystem level compared with the soil carbon sequestration level. For crop switching, corn is too important for both food yield and ethanol production to be replaced. Soybean replacement is not efficient in terms of carbon sequestration. Wheat, on the contrary, can sequester a similar amount of soil carbon as soybeans, and when replaced with biofuel crops, release relatively less carbon and lose a relatively small amount of food yield. Our analyses here and studies elsewhere suggest that *Miscanthus* could better serve as an energy crop than food crops or switchgrass, considering both economic (Hill *et al.*, 2006; Khanna *et al.*, 2008; Smeets *et al.*, 2009) and environmental (McLaughlin & Walsh, 1998; Fan *et al.*, 2007; Heaton *et al.*, 2008; Smeets *et al.*, 2009) benefits.

Other factors should also be considered for further evaluation of bioenergy production, LULC conversion, and GHG mitigation. For example, first, conversion efficiency varies among different feedstocks. At present, conversion technology for conventional biofuel is relatively well established. In contrast, biomass conversion to second-generation biofuel is still new and the conversion efficiency is relatively low, but a potentially higher efficiency is expectable (Fargione *et al*., 2010). Second, food preference will decide the human diet and therefore food crop types, which will influence the land-use decision. Third, nitrogen oxides produced from agroecosystems due to the use of nitrogen fertilizers will also contribute to GHG emissions (Melillo *et al*., 2009). Furthermore, crop harvest should be included to estimate carbon balance in life-cycle-assessment (Davis *et al*., 2009; Fargione *et al*., 2010). Thus, a more comprehensive analysis is needed to evaluate the economic and environmental effects of land-use change to meet the bioenergy demand in the United States.

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References

Anderson-Teixeira KJ, Davis SC, Masters MD, Delucia EH (2009) Changes in soil organic carbon under biofuel crops. *Global Change Biology Bioenergy*, **1**, 75–96.

Beringer T, Lucht W, Schaphoff S (2011) Bioenergy production potential of global biomass plantations under environmental and agricultural constraints. *Global Change Biology Bioenergy*, **3**, 299–312.

Cai X, Zhang X, Wang D (2011) Land availability for biofuel production. *Environmental Science and Technology*, **45**, 334–339.

Campbell JE, Lobell DB, Genova RC, Field CB (2008) The global potential of bioenergy on abandoned agriculture lands. *Environmental Science and Technology*, **42**, 5791–5794.

Carriquiry MA, Du X, Timilsina GR (2011) Second generation biofuels: economics and policies. *Energy Policy*, **39**, 4222–4234.

Clifton-Brown JC, Breuer J, Jones MB (2007) Carbon mitigation by the energy crop, *Miscanthus*. *Global Change Biology*, **13**, 2296–2307.

Davis SC, Anderson-Teixeira KJ, Delucia EH (2009) Life-cycle analysis and the ecology of biofuels. *Trends in Plant Science*, **14**, 140–146.

DOE (2006) *Breaking the biological barriers to cellulosic ethanol: a joint research agenda*. Report from the December 2005 Workshop, DOE/SC-0095. U.S. Department of Energy Office of Science. Available at: http://www.genomicscience.energy.gov/biofuels/ (accessed May 2011).

Evans G (2007) *International Biofuels Strategy Project: Liquid Transport Biofuels - Technology Status Report*. The National Non-Food Crops Centre. Available at: http://www.nnfcc.co.uk/tools/international-biofuels-strategy-project-liquid-transport-biofuels technology-status-report-nnfcc-08-017 (accessed May 2011).

Fargione JE, Plevin RJ, Hill JD (2010) The ecological impact of biofuels. *Annual Review of Ecology, Evolution, and Systematics*, **41**, 351–377.

Fan S, Freedman B, Gao J (2007) Potential environmental benefits from increased use of bioenergy in China. *Environmental Management*, **40**, 504–515.

FAOSTAT (2011) FAOSTAT. Available at: http://www.faostat.fao.org/ (accessed May 2011).

Fargione J, Hill J, Tilman D, Polasky S, Hawthorne P (2008) Land clearing and the biofuel carbon debt. *Science*, **319**, 1235–1238.

Farr TG, Rosen PA, Caro E *et al*. (2007) The shuttle radar topography mission. *Review of Geophysics*, **45**, RG2004.

Farrell AE, Plevin RJ, Turner BT, Jones AD, O'hare M, Kammen DM (2006) Ethanol can contribute to energy and environmental goals. *Science*, **311**, 506–508.

Franzluebbers AJ, Francis CA (1995) Energy output: input ratio of maize and sorghum management systems in eastern Nebraska. *Agriculture Ecosystems and Environment*, **53**, 271–278.

Hansen EM, Christensen BT, Jensen LS, Kristensen K (2004) Carbon sequestration in soil beneath long-term *Miscanthus* plantations as determined by ^{13}C abundance. *Biomass and Bioenergy*, **26**, 97–105.

Heath LS, Birdsey RA, Williams DW (2002) Methodology for estimating soil carbon for the forest carbon budget model of the United States, 2001. *Environmental Pollution*, **116**, 373–380.

Heaton E, Voigt T, Long SP (2004) A quantitative review comparing the yields of two candidate C_4 perennial biomass crops in relation to nitrogen, temperature and water. *Biomass and Bioenergy*, **27**, 21–30.

Heaton EA, Dohleman FG, Long SP (2008) Meeting U.S. biofuel goals with less land: the potential of *Miscanthus*. *Global Change Biology*, **14**, 1–15.

Hicke JA, Lobell DB (2004) Spatiotemporal patterns of cropland area and net primary production in the central United States estimated from U.S.DA agricultural information. *Geophysical Research Letters*, **31**, L20502.

Hicke JA, Lobell DB, Asner GP (2004) Cropland area and net primary production computed from 30 years of U.S.DA agricultural harvest data. *Earth Interactions*, **8**, 1–20.

Hill J, Nelson E, Tilman D, Polasky S, Tiffany D (2006) Environmental, economic, and energetic costs and benefits of biodiesel and ethanol biofuels. *Proceedings of the National Academy of Sciences of the United States of America*, **103**, 11206–11210.

Hoekman SK (2009) Biofuels in the U.S. – challenges and opportunities. *Renewable Energy*, **34**, 14–22.

Hurt C, Tyner W, Doering O (2006) Economics of Ethanol. Purdue Extension. Available at: http://www.extension.purdue.edu/extmedia/ID/ID-339.pdf (accessed May 2011).

IEA bioenergy (2008) *From 1st- to 2nd-Generation Biofuel Technologies: an Overview of Current Industry and RD&D Activities (A Joint Task 39 and IEAHQ Report)*. IEA bioenergy. Available at: http://www.ieabioenergy.com/MediaItem.aspx?id=6060 (accessed May 2011).

Jobbágy EG, Jackson RB (2000) The vertical distribution of soil organic carbon and its relation to climate and vegetation. *Ecological Applications*, **10**, 423–436.

Kahle P, Beuch S, Boelcke B, Leinweber P, Schulten H (2001) Cropping of *Miscanthus* in Central Europe: biomass production and influence on nutrients and soil organic matter. *European Journal of Agronomy*, **15**, 171–184.

Khanna M, Dhungana B, Clifton-Brown J (2008) Costs of producing miscanthus and switchgrass for bioenergy in Illinois. *Biomass and Bioenergy*, **32**, 482–493.

Kim H, Kim S, Dale B (2009) Biofuels, land use change, and Greenhouse gas emissions: some unexplored variables. *Environmental Science and Technology*, **43**, 961–967.

Lapola DM, Schaldach R, Alcamo J *et al*. (2010) Indirect land-use changes can overcome carbon savings from biofuels in Brazil. *Proceedings of the National Academy of Sciences of the United States of America*, **107**, 3388–3393.

Lewandowski I, Schmidt U (2006) Nitrogen, energy and land use efficiencies of miscanthus, reed canary grass and triticale as determined by the boundary line approach. *Agriculture, Ecosystems and Environment*, **112**, 335–346.

Lewandowski I, Scurlock JMO, Lindvall E, Christou M (2003) The development and current status of perennial rhizomatous grasses as energy crops in the US and Europe. *Biomass and Bioenergy*, **25**, 335–361.

Liebig M, Johnson H, Hanson J, Frank A (2005) Soil carbon under switchgrass stands and cultivated cropland. *Biomass and Bioenergy*, **28**, 347–354.

Liebig M, Schmer M, Vogel K, Mitchell R (2008) Soil carbon storage by switchgrass grown for bioenergy. *BioEnergy Research*, **1**, 215–222.

Lu X, Zhuang Q (2010) Evaluating climate impacts on carbon balance of the terrestrial ecosystems in the Midwest of the United States with a process-based ecosystem model. *Mitigation and Adaptation Strategies for Global Change*, **15**, 467–487.

Lu Y, Zhuang Q, Zhou G, Sirin A, Melillo J, Kicklighter D (2009) Possible decline of the carbon sink in the Mongolian Plateau during the 21st century. *Environmental Research Letters*, **4**, 045023.

Lubowski RN, Vesterby M, Bucholtz S, Baez A, Roberts MJ (2006) Major uses of land in the United States, 2002. *Economic Information Bulletin*, No. (EIB-14). Available at: http://www.ers.usda.gov/Publications/EIB14/ (accessed May 2011).

Mandal KG, Saha KP, Ghosh PK, Hati KM, Bandyopadhyay KK (2002) Bioenergy and economic analysis of soybean-based crop production systems in central India. *Biomass and Bioenergy*, **23**, 337–345.

McGuire AD, Melillo JM, Joyce LA, Kicklighter DW, Grace AL, Moore B III, Vorosmarty CJ (1992) Interactions between carbon and nitrogen dynamics in estimating net primary productivity for potential vegetation in North America. *Global Biogeochemical Cycles*, **6**, 101–124.

McGuire AD, Sitch S, Clein JS *et al.* (2001) Carbon balance of the terrestrial biosphere in the twentieth century: analyses of CO_2, climate and land use effects with four process-based ecosystem models. *Global Biogeochemical Cycles*, **15**, 183–206.

McLaughlin SB, Kszos LA (2005) Development of switchgrass (*Panicum virgatum*) as a bioenergy feedstockin the United States. *Biomass and Bioenergy*, **28**, 515–535.

McLaughlin SB, Walsh ME (1998) Evaluating environmental consequences of producing herbaceous crops for bioenergy. *Biomass and Bioenergy*, **4**, 317–324.

Melillo JM, McGuire AD, Kicklighter DW, Moore B, Vorosmarty CJ, Schloss AL (1993) Global climate change and terrestrial net primary production. *Nature*, **363**, 234–240.

Melillo JM, Reilly JM, Kicklighter DW *et al.* (2009) Indirect emissions from biofuels: how important? *Science*, **326**, 1397–1399.

Meyer MH, Paul J, Anderson NO (2010) Competive ability of invasive *Miscanthus* biotypes with aggressive switchgrass. *Biological Invasions*, **12**, 3809–3816.

Mitchell TD, Jones PD (2005) An improved method of constructing a database of monthly climate observations and associated high-resolution grids. *International Journal of Climatology*, **25**, 693–712.

Monfreda C, Ramankutty N, Foley JA (2008) Farming the planet: 2. Geographic distribution of crop areas, yields, physiological types, and net primary production in the year 2000. *Global Biogeochemical Cycles*, **22**, GB1022.

Parrish DJ, Fike JH (2005) The biology and agronomy of switchgrass for biofuels. *Critical Reviews in Plant Sciences*, **24**, 423–459.

Pimentel D, Marklein A, Toth MA *et al.* (2010) Environmental and economic costs of biofuels. *Human Ecology*, **4**, 349–369.

Post WM, Kwon KC (2000) Soil carbon sequestration and land-use change: processes and potential. *Global Change Biology*, **6**, 317–327.

Prince SD, Haskett J, Steininger M, Strand H, Wright R (2001) Net primary production of U.S. midwest croplands from agricultural harvest yield data. *Ecological Applications*, **11**, 1194–1205.

Qin Z, Huang Y (2010) Quantification of soil organic carbon sequestration potential in cropland: a model approach. *Science China Life Sciences*, **53**, 868–884.

Ragauskas AJ, Williams CK, Davison BH *et al.* (2006) The path forward for biofuels and biomaterials. *Science*, **311**, 484–489.

Raich JW, Tufekcioglu A (2000) Vegetation and soil respiration: correlations and controls. *Biogeochemistry*, **48**, 71–90.

Raich JW, Rastetter EB, Melillo JM *et al.* (1991) Potential net primary productivity in South America: application of a global model. *Ecological Applications*, **1**, 399–429.

Rathke GW, Wienhold BJ, Wilhelm WW, Diepenbrock W (2007) Tillage and rotation effect on corn-soybean energy balances in eastern Nebraska. *Soil & Tillage Research*, **97**, 60–70.

Renewable Fuels Association (2011) *Historic U.S. Fuel Ethanol Production*. Renewable Fuels Association. Available at: http://www.ethanolrfa.org/pages/statistics#E (accessed May 2011).

Scheffran J, BenDor T (2009) Bioenergy and land use: a spatial-agent dynamic model of energy crop production in Illinois. *International Journal of Environment and Pollution*, **39**, 4–27.

Schmer MR, Mitchell RB, Vogel KP, Schacht WH, Marx DB (2010) Spatial and temporal effects on switchgrass stands and yield in the Great Plains. *BioEnergy Research*, **3**, 159–171.

Schmer MR, Liebig MA, Vogel KP, Mitchell RB (2011) Field-scale soil property changes under switchgrass managed for bioenergy. *Global Change Biology Bioenergy*, doi: 10.1111/j.1757-1707.2011.01099.x.

Searchinger TD, Heimlich R, Dong F *et al.* (2008) Use of U.S. croplands for biofuels increases greenhouse gases through emissions from land-use change. *Science*, **319**, 1238–1540.

Smeets EMW, Lewandowski IM, Faaij APC (2009) The economical and environmental performance of miscanthus and switchgrass production and supply chains in a European setting. *Renewable and Sustainable Energy Reviews*, **13**, 1230–1245.

Stewart JR, Toma Y, Fernandez FG, Nishiwaki A, Yamada T, Bollero G (2009) The ecology and agronomy of *Miscanthus* sinensis, a species important to bioenergy crop development, in its native range in Japan: a review. *Global Change Biology Bioenergy*, **1**, 126–153.

The United States of America (2007) The Energy Independence and Security Act of 2007 (H.R. 6). Available at: http://energy.senate.gov/public/index.cfm?FuseAction=IssueItems.Detail&IssueItem_ID=f10ca3dd-fabd-4900-aa9d-c19de47df2da&Month=12&Year=2007 (accessed May 2011).

Thomason WE, Raun WR, Johnson GV, Taliaferro CM, Freeman KW, Wynn KJ, Mullen RW (2004) Switchgrass response to harvest frequency and time and rate of applied nitrogen. *Journal of Plant Nutrition*, **27**, 1199–1226.

Tilman D, Hill J, Lehman C (2006) Carbon-negative biofuels from low-input high-diversity grassland biomass. *Science*, **314**, 1598–1600.

Tilman D, Socolow R, Foley JA *et al.* (2009) Beneficial biofuels- the food, energy, and environment trilemma. *Science*, **17**, 270–271.

Tufekcioglu A, Raich JW, Isenhart TM, Schultz RC (1999) Fine root dynamics, coarse root biomass, root distribution, and soil respiration in a multispecies riparian buffer in Central Iowa, U.S.A. *Agroforestry Systems*, **44**, 163–174.

Turner DP, Koerper GJ, Harmon ME, Lee JJ (1995) A carbon budget for forests of the conterminous United States. *Ecological Applications*, **5**, 421–436.

U.S.DA (2007). *Major Land Uses*. U.S.DA. Available at: http://www.ers.usda.gov/Data/MajorLandUses/ (accessed May 2011).

Wang S, Huang M, Shao X, Mickler RA, Li K, Ji J (2004) Vertical distribution of soil organic carbon in China. *Environmental Management*, **33**, S200–S209.

West TO, Six J (2007) Considering the influence of sequestration duration and carbon saturation on estimates of soil carbon capacity. *Climatic Change*, **80**, 25–41.

Woodbury PB, Heath LS, Smith JE (2007) Effects of land use change on soil carbon cycling in the conterminous United States from 1900 to 2050. *Global Biogeochemical Cycles*, **21**, GB3006.

Yazaki Y, Mariko S, Koizumi H (2004) Carbon dynamics and budget in a *Miscanthus sinensis* grassland in Japan. *Ecological Research*, **19**, 511–520.

Zhuang Q, McGuire AD, Melillo JM *et al.* (2003) Carbon cycling in extratropical terrestrial ecosystems of the Northern Hemisphere during the 20th century: a modeling analysis of the influences of soil thermal dynamics. *Tellus*, **55B**, 751–776.