ECOSPHERE

Biogeochemical consequences of regional land use change to a biofuel crop in the southeastern United States

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Abstract.   The United States has mandated the production of 80 billion liters of second-generation biofuel by 2022, and several approaches to meet this target focus on using ligno-cellulosic ethanol from perennial grasses and non-food crops. The large-scale deployment of biofuel agronomy should consider high-yielding crops that meet ethanol production goals, choose appropriate landscapes for biofuel crops from a climate and food production standpoint, and a full consideration of the environmental impact of large-scale land use change. The southeastern United States has a long growing season conducive for producing high-yielding crops, and is relatively less important to US food production than the rain-fed Midwestern states that have been extensively studied for biofuel crops. We use the DayCent biogeochemical model to run simulation experiments to test the hypotheses that converting a large swath of traditional agriculture in the southeastern United States that is already utilized for bioenergy production (assuming 35% of current corn-soy, and 10% of grazed pasture hectares; ~950,000 ha) to energy cane will result in greater biomass production, increased soil C storage, decreased soil N losses and lower greenhouse gas emissions than a landscape of corn-soy rotations and interspersed grazed pasture. Our simulations suggest that energy cane above-ground productivity on former pasture and corn-soy fields would be between 52-59 million Mg dry mass per year, resulting in 21.1–23.7 billion liters of ligno-cellulosic ethanol, or ~28% of the 2022 US government mandate. DayCent did not predict significant changes in soil C flux from land conversion to energy cane, but simulations predicted lower rates of N loss compared to current agriculture. GHG emissions from energy cane landscapes were substantially higher on former pasture, but an order of magnitude lower when compared to corn-soy hectares. While further study is needed to ascertain the full economic and industrial feasibility of converting nearly 1,000,000 ha of land to energy cane production, our results suggest that such an undertaking could meet a sizeable fraction of the US ethanol mandate, reduce N pollution and GHG emissions, and avoid compromising land devoted to food production in the southeastern United States.

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INTRODUCTION

The Energy Independence and Security Act of 2007 commits the United States to produce 80 billion liters of advanced biofuel by 2022 (US Congress 2007). There were approximately 50 billion liters of bio-ethanol produced in the United States in 2012 (United States Energy Information Administration 2012), but only ~40 million liters (<0.01%) came from second generation or advanced biofuels (Federal Register 2012). By definition, advanced biofuels are essentially any fuels not derived from plant sugars such as corn, sugar beet or sugar cane, and are instead formulated from biomass-derived ethanol converted from cellulose and lignin (Federal Register 2012). However, there is considerable uncertainty regarding what crops will be used as biomass feedstock and what landscapes will be converted to maximize energy crops' potential to produce fuel. In addition to providing a domestic source of energy, there is interest in developing environmentally beneficial crops that can maximize soil C storage, minimize nitrogen losses and mitigate greenhouse gas (GHG) emissions (Tilman et al. 2006, Fargione et al. 2010, Somerville et al. 2010, Davis et al. 2012).

Most bio-ethanol produced in the United States is derived from corn grain (Zea maize; USDA Economic Research Service [USDA-ERS] 2012). Corn grain and grain derivatives are ingredients in many human food and animal feed products, and since ~40% of all corn grown in the US goes to making ethanol, using corn as a bio-ethanol feedstock has direct implications for food costs (USDA-ERS 2012). However, some estimates suggest that up to 12% of corn used for ethanol re-enters the market as animal feed in the form of distiller's grain (National Corn Growers Association 2012). Because rain-fed agriculture in the midwestern United States is important for both food production and bio-energy production, there is a concerted effort to understand the environmental impact of replacing corn with different biofuel crops (Heaton et al. 2008, Davis et al. 2010, 2012). An alternative is to expand biofuel crop production into regions that are less important to food production.

The southeastern United States holds great potential for expansion of biofuel crops. This region has a longer growing season than northern latitudes in the corn belt of the US, and there is generally more corn failure in the SE US, which leaves open the possibility that converting corn land in this region will result in more efficient land use in addition to biofuel production. For example, 94% of corn planted in Midwestern states was actually harvested in 2011, compared to 83% of planted hectares harvested in the SE US states (NASS 2012).

Instead of corn, high yielding C_4 perennial grasses like sugarcane have been explored as feedstock for ligno-cellulosic ethanol (Sladden et al. 1991, Duff and Murray 1996, Cheng et al. 2008). Sugarcane varieties grown for food sugar are grown in limited areas of the SE USA. However, a cold tolerant variety of sugarcane known as energy cane is being considered as a biofuel crop (Mark et al. 2009, Kim and Day 2011), with potential for high yields (20–70 $Mg \cdot ha^{-1} \cdot yr^{-1}$ dry mass) and may be grown in more northern latitudes in the US compared to traditional sugarcane. "Energy cane" has been touted as an energy crop since at least the 1980s (Alexander 1985). Energy cane is a hybrid of the traditional sugar cane grown for food sugar, Saccharum officinarum and its close relative, the wild grass S. spontaneum, and has been developed for maximum biomass at the cost of reduced sucrose content (Matsuoka et al. 2014). Energy cane hybrids produce greater numbers of tillers than traditional sugar cane, and have more established rhizomes, which facilitate resilience under cultivation pressures like compaction from harvesting. Furthermore, heartier rhizomes may increase the longevity of the crop life span in a perennial harvesting system (Matsuoka et al. 2014). Indeed, our previous modeling work has shown that in an area of Florida at the northern extent of traditional sugarcane cultivation, simulated energy cane yields ranged from 46 to 72 dry mass $Mg \cdot ha^{-1} \cdot yr^{-1}$ (Duval et al. 2013). Relative to the grazed pasture it displaced, we also predict that planting energy cane will decrease GHG exchange with the atmosphere, depending on soil type (Duval et al. 2013).

Another benefit of sugarcane varieties for biofuel feedstock is the long history of sugarcane cultivation and research in the SE US (Greenland 2005, Gilbert et al. 2006, Kim and Day 2011). Deploying a sugarcane variety for large-scale

biofuel production is economically sensible because the plant's nutrient, light and water requirements are well understood, as are its pathogens (Vallis et al. 1996, James and Olivares 1997, Hoy et al. 1999, Morris and Gilbert 2005). Therefore, best practices for sugarcane agronomy can be followed for energy cane, and the crop is a suitable candidate for simulation modeling because parameters related to its growth and impact on soil systems are understood (Vallis et al. 1996, Duval et al. 2013).

In the absence of long-term data on the environmental impacts of converting traditional agriculture to energy cane, modeling experiments can be used to develop hypotheses about the viability of large-scale land use conversion to biofuel crops (sensu Greenland 2005). We therefore based *in silico* experiments on the USDA projections that to meet EISA standards for 2022, that 35% of corn-soy rotations in the SE US will be converted to energy cane production (R. Steiner, *personal communication*). In addition to 35% of current corn-soy hectares, we also assume that some portion of grazed pasture will be converted to biofuel crops, and we conservatively chose 10% of existing pasture for possible energy cane production (Appendices A and B).

Here, we use modeling experiments to provide a range of predictions of changes to C, N and GHG biogeochemistry following the conversion of row crop and grazed pasture land in the SE USA to energy cane production. We use the DayCent biogeochemical model to run our experiments, as it has accurately predicted regional and global yield, C, N, and GHG fluxes in agricultural systems (Parton et al. 1998, Del Grosso et al. 2002, 2010, Hartman et al. 2011), and has been validated for grazed pasture and energy cane in the SE US (Duval et al. 2013). We hypothesize that: (1) Land use conversion from traditional agriculture (corn-soy rotations and grazed pasture) to energy cane will result in greater harvestable biomass, and therefore higher potential ethanol production than existing landscapes in the SE US. (2) Nitrogen (specifically NO_3^-) losses will be lower after land use conversion because of the higher N efficiency of energy cane compared to row-crop agriculture. (3) Greenhouse gas emissions, including nitrous oxide (N_2O), in the SE US region will be substantially reduced after conversion of a portion of traditional agriculture to a perennial biofuel crop.

METHODS

DayCent parameterization and validation

We employed the process based biogeochemical model DayCent to simulate the effects of a large-scale land use change from traditional agriculture to energy cane (Del Grosso 2002, Davis et al. 2010, Parton et al. 2010). DayCent has been used to simulate sugarcane production in Brazil (Galdos et al. 2009, 2010), Australia (Vallis et al. 1996) and energy cane in Florida (Duval et al. 2013). These studies show that the DayCent soil organic matter sub-model correctly simulates the impacts of burning, fertilizer, irrigation and organic matter additions on plant productivity, soil carbon levels and surface litter decay for sugar cane and energy cane. We additionally parameterized DayCent by using empirically measured plant and soil traits (C and N content) from plant tissue (stems and foliar tissue) and soil cores taken from an energy cane farm in central Florida (Duval et al. 2013). The principal changes were to adjust leaf and stem C:N based on field collections and lab analysis of C and N, and adjusting the parameter for C allocation to stems in DayCent, to reflect the lower C content of stems relative to N for energy cane (Duval et al. 2013). Parameters altered from the CENTURY sugarcane values used in Vallis et al. (1996) display an "*" in Appendix A. There are seven major groups of parameters which were changed to represent the growth of the sugarcane crop. The seven groups include:

(1) the maximum growth rate of the sugarcane plant (PFDX(1));
(2) the maximum and minimum carbon to nitrogen ratios for the different plant parts (CERFOR(1, 1, 1));
(3) the fraction of carbon allocated to the growth of the sugarcane stems (FCFRAC(4, 1));
(4) the fraction of carbon allocated for the growth of fine roots (TFRTCN(1), etc.) depending on the nitrogen and water stress;
(5) the monthly death rate for the sugarcane live leaves;
(6) the model parameters which control live

leaf area of the sugarcane plant (BTOLAI, MAXLAI, and KLAI);

(7) the symbiotic nitrogen fixation rate for the sugarcane plant (SNFXMX(2)).

These parameters were altered to match the growth of the different sugarcane plant parts and nitrogen inputs to the system from sugarcane symbiotic N fixation. Observed sugarcane field data was used to estimate the C/N ratios for the different plant parts, carbon allocation for plant growth for the different plant parts and the death rate of the live leaves. The maximum growth rate and symbiotic N fixation rates were altered in order to match the sugarcane crop yield data (Duval et al. 2013).

Pasture simulations were validated with productivity data for 15 sites throughout the SE USA (Fig. 1A; NASS 2012, Duval et al. 2013). Corn-soy rotations were modeled using variables previously used to model these crops in the SE US (Del Grosso et al. 2006). Parameters for corn-soy rotations in this part of the US have successfully modeled aboveground production, N_2O and net GHG flux, so we relied on existing corn and soybean parameterizations for DayCent (Del Grosso et al. 2002, 2006).

Aboveground biomass data, consisting of stem plus foliar tissue in the model compared to dry mass reported in the literature, were used to validate DayCent, as this variable has been measured widely across a range of sites, and validation based on productivity for other crops reliably predicts trace gas flux (Valentine et al. 1994, Del Grosso et al. 2002, 2006, Adler et al. 2007). The literature values used in validation came from the geographical range of sugarcane and potential energy cane production presented here. We observed a strong correlation ($r^2 = 0.82$; Fig. 1B) between our modeled biomass from DayCent and literature values.

We collected data on the extent of corn, soy and pasture hectares in the counties listed as plant hardiness zones 8b or higher in Alabama, Arkansas, Florida, Georgia, Louisiana, Mississippi, South Carolina and Texas from the National Agricultural Statistics Service database (Appendix B; NASS 2012). We populated DayCent with soil parameters from county level soil data from STATSGO (http://water.usgs.gov/GIS/metadata/usgswrd/XML/ussoils.xml). The soil characteristics for each county represent the modal soil values for the area of the county that is classified as an agricultural region by the STATSGO data set (Hartman et al. 2011). Our previous model parameterization for soil types used two soil types common to agricultural areas in central Florida, a low organic matter Spodosol, and a Histosol that contained extremely high quantities of organic matter (Duval et al. 2013). Results for energy cane grown in the model on both of these soil types are included in the validation regressions (Fig. 1), and highlight the facility of the model to reasonably predict energy cane production on disparate soil types. Daily weather data for each county were obtained from VEMAP (http://www.cgd.ucar.edu/vemap/) and DAYMET (https://daymet.ornl.gov; Parton et al. 1998, 2010, Kittel et al. 2004, Natural Resources Conservation Service Soil Survey 2010).

We ran a 2000-year spin-up period in DayCent to stabilize soil carbon and mimic likely landscapes in the SE US prior to widespread conversion to agriculture. We used a combination of savanna tree-prairie as the initial system prior to initiating corn-soy rotations and then energy cane cultivation. Simulations for each county in our database (Appendix B) for both corn-soy rotations and for grazed pasture for a 15-year period prior to land use change to energy cane. The corn-soy rotation was based on previous DayCent simulations for the United States (Del Grosso et al. 2006).

The pasture simulated in the experiment was grazed every year between March and October at a mid-level grazing intensity, with 10% of live shoot and 1.0% of standing dead shoot removed by grazing. We do recognize that subtle changes in cattle diets alter the gaseous N emission profile from manure (i.e., N_2O flux versus NH_3 volatilization; Arndt et al. 2015), but it is beyond the scope of this modeling effort to encapsulate the range of potential feeding strategies and therefore cattle emissions on a regional level. However, to ensure cattle impact was included in our greenhouse gas assessment, we assumed a cattle stocking density of 1 animal per hectare, and for our regional greenhouse gas assessment, we assumed enteric CH_4 emissions to be 95 $kg \cdot animal^{-1} \cdot yr^{-1}$. Prior to planting energy cane, we modeled a plowing event to a depth of 30 cm to remove the pasture or residual corn-soy
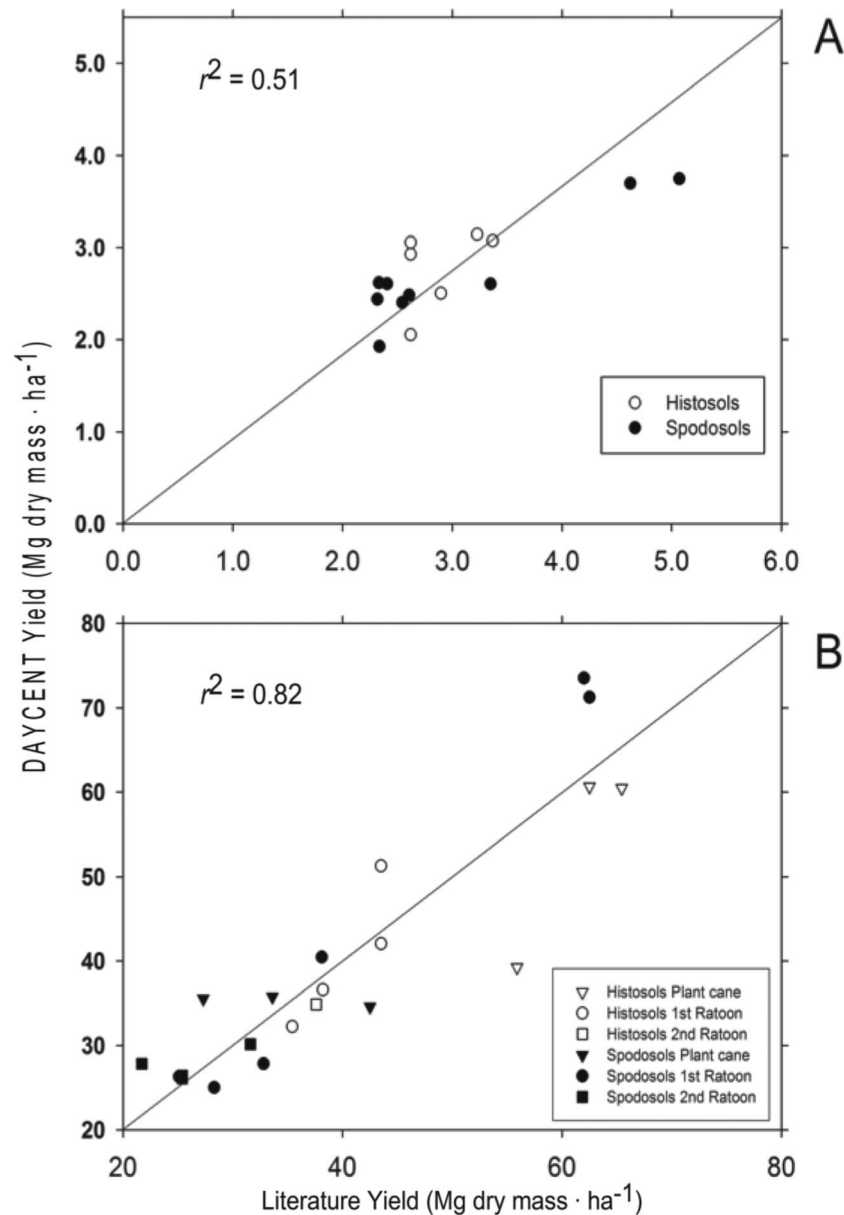
Fig. 1. DayCent Validation for (A) pasture yield and (B) energy cane yield. Data from the literature and the National Agricultural Statistics Service are regressed against DayCent output for grasslands and energy cane in the counties where data were available. Line represents 1:1 correspondence with literature and model.

vegetation, and initiate the land use change to energy cane cultivation.

The simulated cycle of energy cane planting and harvest was based on accepted sugarcane agronomy (Glaz and Ulloa 1993, Vallis et al. 1996, Wiedenfeld and Encisco 2008) and communications with sugarcane agronomists (R. Gilbert and B. Glaz, *personal communication*). Because it is a

perennial C_4 grass, energy cane could be grown for several years and perhaps nearly a decade without replacement (Matsuoka et al. 2014). However, our modeling exercise focused on understanding the implications of a regional agronomic shift toward using this plant for bioenergy. Thus, we chose a conservative approach of re-planting energy cane every 3 years,

to minimize variation across the region that we could not easily account for when determining management practices. Longer rotation intervals would likely further enhance increases in soil organic carbon and reduce GHG emissions from perennial crops (Hudiburg et al. 2014).

Energy cane simulations began January of the first year (2013), followed by a two-year ratoon (crop regenerated from remaining biomass) from which 80% of the above ground biomass was harvested in December. At the end of the second ratoon (third year of the cycle), the crop was removed and the land plowed before planting a new seed crop (Glaz and Morris 2010). Model runs for energy cane were simulated from 2013 to 2041.

To test the effect of irrigation, one set of simulations was made on energy cane that was not artificially irrigated, i.e., precipitation was the only external water source to soils. For irrigated energy cane, watering events were scheduled every month throughout the dry season, and every two months during the rainy season to maintain soil water at field capacity. Fertilizer (NH_4^+-NO_3^-) was applied to both irrigated and non-irrigated energy cane in mid-February and mid-June of each year of the simulation, at a rate of 102 kg N·ha^{-1} per fertilization event. This fertilization regime for both irrigated and non-irrigated cane was based on studies that suggest that split fertilization at this rate maximizes sugarcane yield (Allen et al. 2010). Fertilizing above this level does not necessarily increase yield but significantly increases soil N_2O efflux (Vallis et al. 1996, Muchovej and Newman 2004). This cycle of ratooning and planting was repeated in the simulation for 27 years following conversion from pasture; i.e., nine three-year cycles in total.

Statistical analysis

Our DayCent simulations for 289 counties offered several opportunities for calculating the biogeochemical effect of changing land use from pasture or corn-soy rotations to energy cane and testing the effect of irrigation and duration of ratooning. DayCent calculates values for biomass, C and N flux and GHG flux on a per land area basis, and output is expressed as element mass per square meter. This is useful for comparing land changes and assessing variance across a region, however, since there was high variability in the area extent of traditional agriculture (corn-soy and pasture) from county to county, we also scaled the output for those land uses by the proportion of land that would theoretically be converted to energy cane. For our regional summary, we scaled the per county simulated output by multiplying those per area values by 35% of the most recent (typically 2013 data) existing data for corn-soy rotation hectares, and 10% of the reported pasture hectares (NASS 2012; R. Stiener, *personal communication*).

We also compared biomass and biogeochemical parameters for energy cane that were simulated on land converted from pasture versus energy cane grown on former corn-soy hectares. For the regional assessment, values were summed; as the scaling for the number of hectares converted from those previous land uses had already been corrected.

Data were homogeneous for variance among treatments (tested via the Flinger-Killeen test); we therefore used a two-way ANOVA model to examine productivity, C and N cycling differences as a result of ratoon duration and irrigation, and interactions between those factors. Tukey's HSD was used as a post-hoc determination of differences between groups (Crawley 2007). Statistical tests were performed using JMP v.7.1 (SAS Institute, Cary, North Carolina, USA) and R (R Development Core Team 2013).

RESULTS

Biomass

DayCent simulated significantly higher annual biomass yields from energy cane than traditional agriculture in the SE USA (Fig. 2, Table 1). Indeed, predicted energy cane production was an order of magnitude higher than grazed pasture systems, and a factor of four higher than corn-soy rotation productivity, irrespective of irrigation (Table 1). For both irrigated and non-irrigated energy cane, there was no significant yield difference for crops grown on former corn-soy rotation fields compared to crops grown on former pasture. While not statistically significant, the increase in energy cane production under an irrigation management system exceeded that of non-irrigated cane by nearly the same annual output of the previous pasture system (~200 g C·m^{-2}·yr^{-1}).

A

B
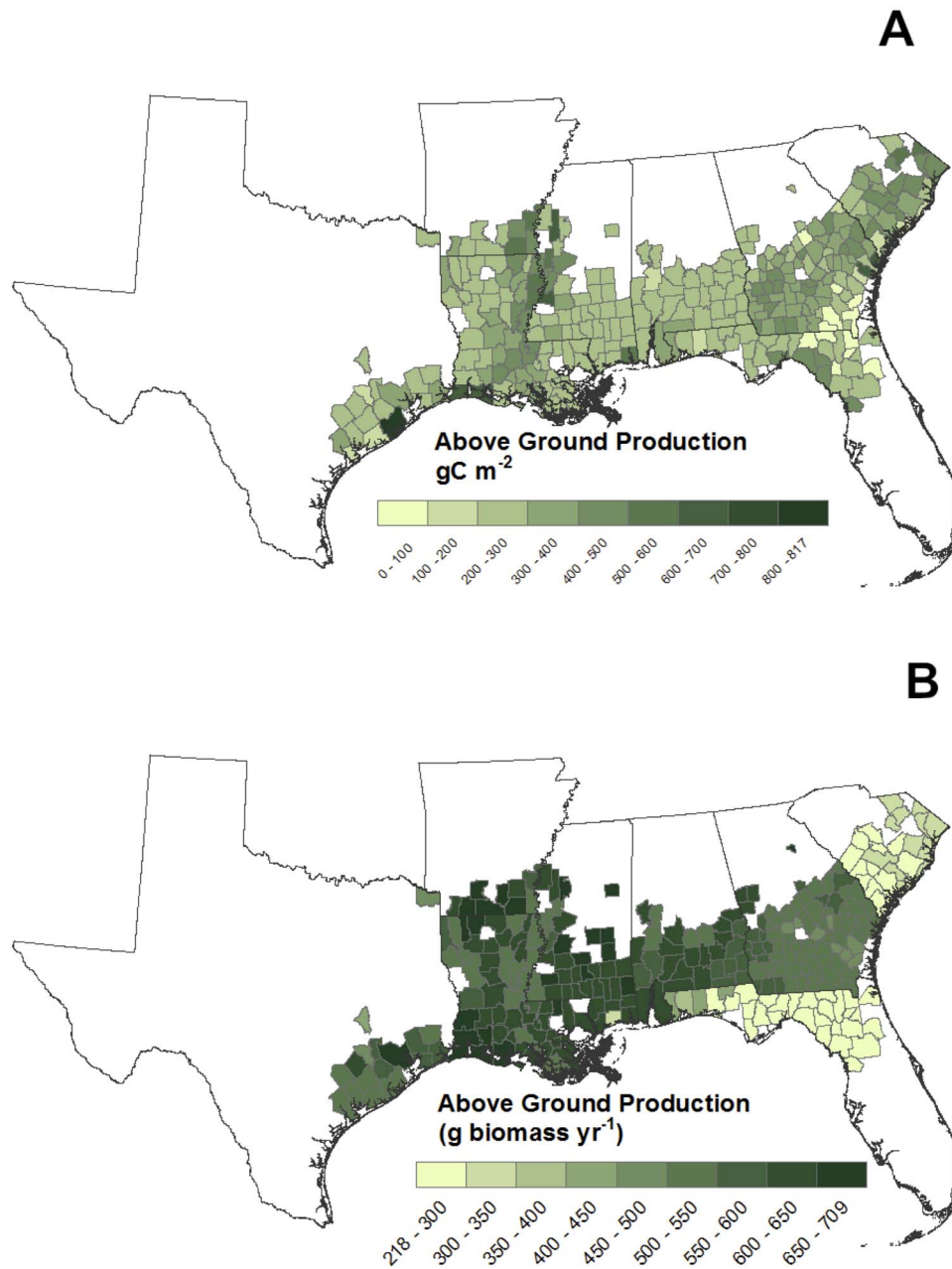

Fig. 2. Above ground biomass (g C m^{-2} yr^{-1}) after land use conversion to energy cane from (A) corn-soy rotation + pasture, (B) energy cane grown under irrigated conditions. Data are presented at the county level.

Nitrogen cycling

There was significantly less NO$_3^-$ leached from land converted to energy cane than traditional agriculture (Fig. 3, Table 1; Wilcoxon test, $\chi^2 =$ 519, $P < 0.001$). Within energy cane simulations, non-irrigated energy cane lost slightly more N via leaching than irrigated cane (Table 1).

However, both non-irrigated and irrigated cane leached significantly more N when planted on former corn-soy fields than when planted on pasture (non-irrigated, $F_{1, 603} = 38.07$, $P < 0.001$; irrigated cane, $\chi^2 = 27.32$, $P < 0.01$). Over the course of the simulation, the conversion of either corn-soy or pasture to energy cane caused a

Table 1. Per area values averaged over the entire southeastern US of biomass, leached N, and total greenhouse gas flux predicted by DayCent simulations as a consequence of converting traditional agriculture (pasture land + corn-soy rotations) to energy cane in the Southeastern United States. Change in soil C and N values are the difference between the last year of the simulation minus the first year of the simulation. Values are means (± SEM).

Output variable	Pasture	Corn-soy rotation	Non-irrigated energy cane	Irrigated energy cane
Aboveground production (g C m^{-2} yr^{-1})	221 (1)	531 (7)	2063 (11)	2262 (9)
Soil carbon (g C m^{-2} yr^{-1})†	1395 (30)	1738 (47)	1625 (25)	1588 (25)
Change in total system carbon (g C m^{-2} yr^{-1})‡	9 (4)	−84 (8)	931 (8)	1032 (9)
Leached nitrogen (g N m^{-2} yr^{-1})§	0.46 (0.01)	6.83 (0.29)	1.78 (0.04)	1.64 (0.04)
N$_2$O emissions (g N m^{-2} yr^{-1})	57 (1)	845 (24)	137 (2)	150 (2)
Net annual GHG emissions (g CO$_2$eq m^{-2} yr^{-1})	317 (2)	878 (24)	28 (3)	33 (3)

† Average annual organic C.
‡ Change in total ecosystem organic matter C, last year—first year of crop rotation or ratoon cycle.
§ Average annual leaching of NO$_3$.

reduction in soil inorganic N. Both corn-soy rotations and pasture showed gains in soil N, while the energy cane fields lost 26.0–27.1 g N·m^2 after 27 years (Wilcoxon test, $\chi^2 = 705$, $P < 0.001$).

Greenhouse gas flux

On a per area basis, there was a significant reduction in greenhouse gas production from energy cane following conversion from traditional agriculture (Fig. 4). Indeed, while both non-irrigated and irrigated energy cane systems were still a net source of GHG's, they produced greenhouse gas fluxes to the atmosphere at a rate nearly a factor of 10 lower than pasture systems, and produced only ~5% of the corn-soy rotation emissions (Table 1). Nitrous oxide flux to the atmosphere was the major driver for overall GHG emissions in corn-soy (96% of total emissions when converted to CO$_2$eq), and N$_2$O flux was offset in energy cane due to those fields serving as CO$_2$ and CH$_4$ sinks (Table 1).

Regional scaling

To evaluate the regional biogeochemical impact of land conversion to energy cane, we also calculated the total productivity, soil C and N fluxes and net GHG flux based on the area of land converted in our simulations in the SE USA (Table 2). Assuming that 10% of current pasture land would be converted resulted in 457,653 ha of land for energy cane production, and adding a 35% conversion of land in corn-soy rotation resulted in an additional 491,362 ha for a total of 949,015 ha of land converted from traditional agriculture to energy cane (Table 2).

We calculate that the total above ground production of pasture was ~2.5 million Mg dry mass·yr^{-1} for this area, and corn-soy production to be ~8.2 million Mg dry mass·yr^{-1} (Table 2). Energy cane productivity on this same land would be between 52 and 59 million Mg C yr^{-1}, which could potentially yield 21.1–23.7 billion liters of lingo-cellulosic ethanol. In addition to greater biomass production after the conversion to energy cane, our simulations suggest that there would be a greenhouse gas emissions increase of between 0.15 and 0.26 million Mg CO$_2$eq for converting existing pasture, but a reduction of ~4.5 million Mg CO$_2$eq emissions from changing corn-soy rotations to energy cane (Table 2).

Discussion

Meeting US biofuel mandates by 2022 will require advances in conversion technology and improvements in agronomy and crop production efficiency. From an environmental perspective, meeting this mandate also necessitates thought as to what types of land will be converted to biofuel crops because of the explicit challenge in growing plants for fuel and minimizing the potential competition for land between fuel and food crops (Tilman et al. 2009, Anderson-Teixeira et al. 2012). The large biomass differences we predicted between traditional agriculture and land converted to energy cane suggests that energy cane can be a viable energy crop in a part of the United States that has a long growing season, while developing this fuel-based agricul-
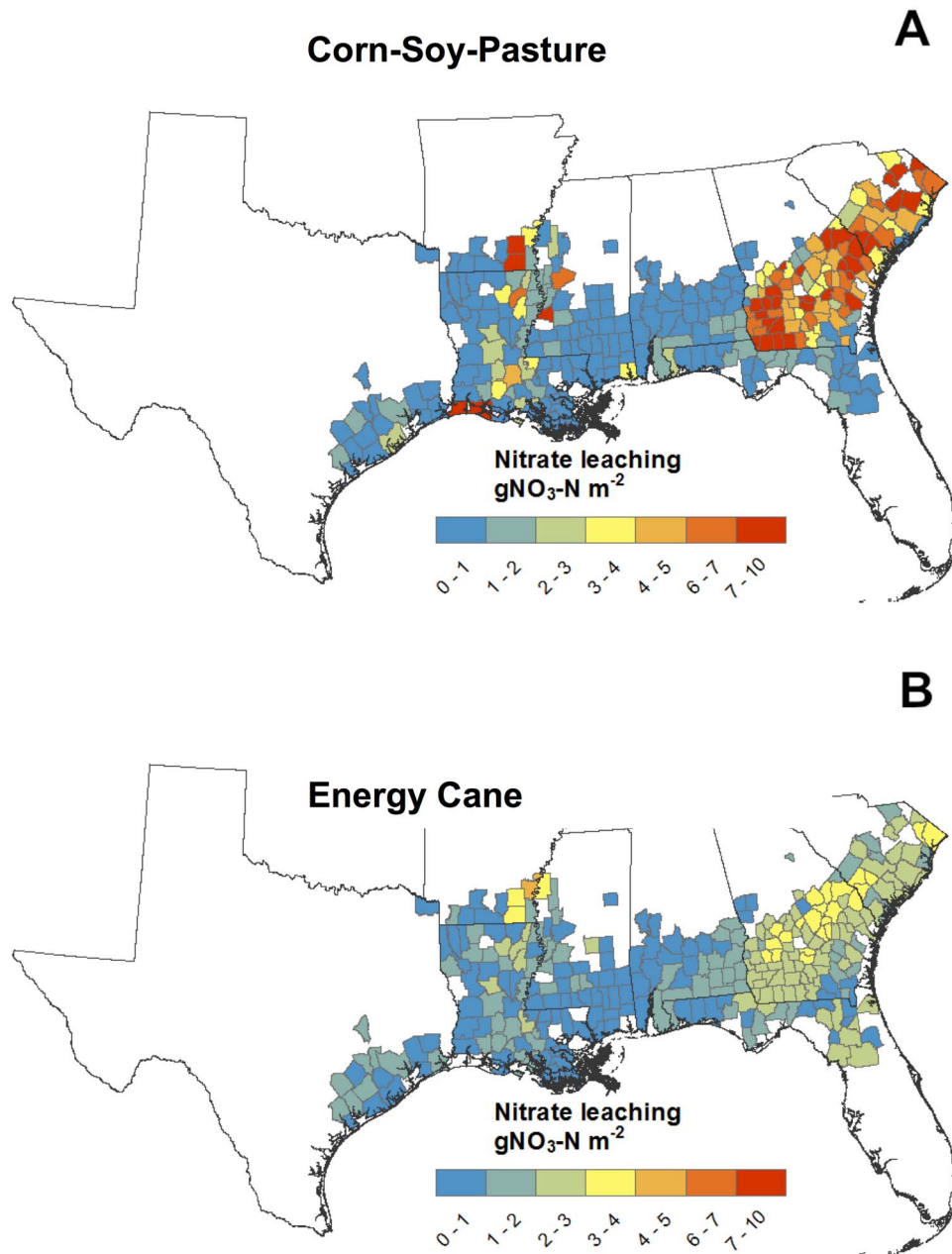
Corn-Soy-Pasture

A

B

Energy Cane

Fig. 3. Leached nitrogen (g NO_3^--N m^{-2} yr^{-1}) after land use conversion to energy cane from (A) corn-soy rotation + pasture, (B) energy cane grown under irrigated conditions. Data are presented at the county level.

ture in a region that is less productive for corn and soybean production than the rain-fed Midwestern United States (NASS 2012). This result supports our first hypothesis that converting land to energy cane will result in significantly higher biomass production than the grazed pasture or corn-soy agriculture that energy cane replaces.

Our simulations only convert a portion of corn-soy rotations to energy cane (35%), and a much smaller (10%) fraction of SE USA grazed pasture hectares, a scenario that portends the possibility of large-scale energy crop production in conjunction with, and not opposed to, food
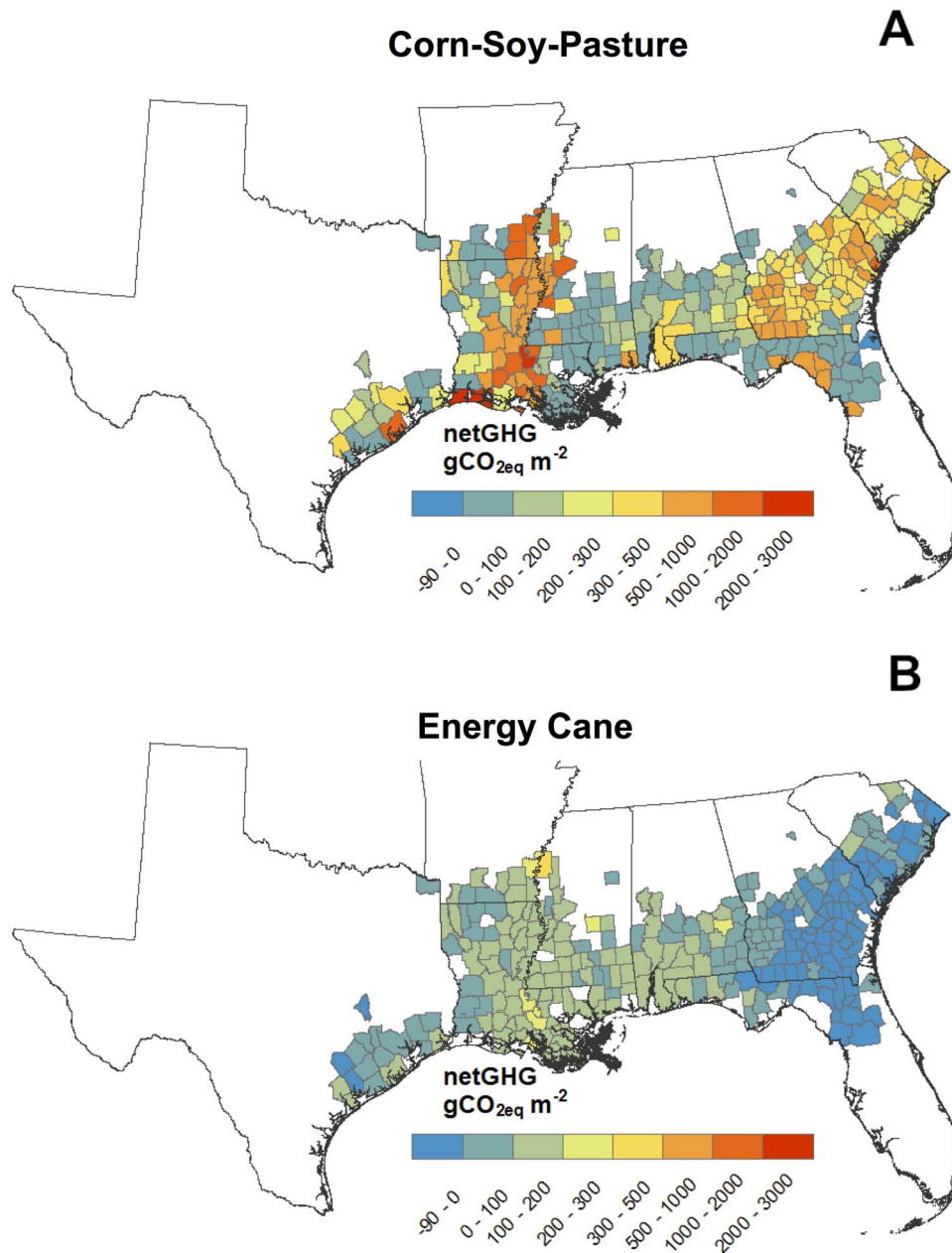
Corn-Soy-Pasture

A

netGHG
gCO$_{2eq}$ m^{-2}

-90 – 0   0 – 100   100 – 200   200 – 300   300 – 500   500 – 1000   1000 – 2000   2000 – 3000

B

Energy Cane

netGHG
gCO$_{2eq}$ m^{-2}

-90 – 0   0 – 100   100 – 200   200 – 300   300 – 500   500 – 1000   1000 – 2000   2000 – 3000

Fig. 4. Total greenhouse gas flux (g CO$_2$eq m^{-2} yr^{-1}) after land use conversion to energy cane from (A) corn-soy rotation + pasture, (B) energy cane grown under irrigated conditions. Data are presented at the county level.

producing agriculture (Tilman et al. 2009). Indeed, 40% of US corn production is currently used for ethanol (National Corn Growers Association 2012), which leaves open the possibility of replacing that area of land with a higher-yielding, and more environmentally favorable (i.e., lower GHG emissions and less N pollution)

crop (Davis et al. 2012).

Considering nitrate pollution in a large-scale biofuel crop conversion program is critical because NO$_3^-$ runoff from agriculture is a massive environmental problem in the SE USA. Nitrate pollution reduces water quality and negatively impacts aquatic biodiversity via nu-

Table 2. Total values of biomass, leached N, and total greenhouse gas flux predicted by DayCent simulations as a consequence of converting land in traditional agriculture (pasture land + corn-soybean rotations) to energy cane in the southeastern United States. Values for land area assume 35% of corn-soy rotations and 10% of current pasture is converted to energy cane across 289 counties and parishes in Alabama, Arkansas, Georgia, Florida, Louisiana, Mississippi, South Carolina and Texas, USA.

Land use	Crop area (ha)	Aboveground dry mass (Mg·yr^{-1})	EtOH yield (ML EtOH·yr^{-1})†	Net GHG flux (Mg CO$_2$eq·yr^{-1})
Pasture	457653	2489516	988	318597
Corn-soy	491362	8211093	3276	5096654
Non-irrigated energy cane	949015	52686452	21127	475221
Irrigated energy cane	949015	59085556	23693	576416

† Calculation of ethanol yields assume 401 L EtOH·Mg^{-1} dry biomass for pasture and energy cane (Somerville et al. 2010), and that all corn-soy biomass on land dedicated to conversion to energy cane (35%) is converted to grain based EtOH (399 L EtOH·Mg^{-1} dry biomass; Somerville et al. 2010).

trient induced anoxia (Rabalais et al. 2002, Vaquer-Sunyer and Duarte 2008). Consistent with our second hypothesis, our results show land use conversion to energy cane caused a five-fold decline in leached N compared to corn-soy agriculture in the counties bordering the Mississippi River, the Gulf of Mexico coasts of Alabama, Florida, Louisiana, Mississippi and Texas, as well as the Atlantic Ocean coasts of South Carolina, Georgia and Florida (Fig. 3). Mechanistically, this is explained by the much greater nitrogen uptake efficiency of energy cane than corn, as the simulations predicted six times greater harvested biomass from energy cane compared to corn-soy, but the harvested N from energy cane was roughly nine times greater. There was a slight increase in nitrate loss from non-irrigated energy cane, but this is likely due to greater biomass production under irrigated conditions, and therefore higher overall mass of N incorporated from the soil into energy cane tissues and less N substrate for nitrification and NO$_3^-$ leaching.

While quantifying the impact of land use change on ocean anoxia or impacts to marine biodiversity are beyond the scope of this experiment, our results suggest that conversion of annual crop land and pasture to high-yielding energy cane would greatly reduce the detrimental effects of N leaching. Therefore, this type of land use change could also be a strategy for N management, and suggests the need to calculate a regional N budget related to biofuel crops in the SE USA (Donner and Kucharik 2008, David et al. 2010).

Net GHG emissions significantly declined as a result of conversion from corn-soy to energy cane. However, within the energy cane simulations, GHG emissions from energy cane land that formerly was grazed pasture were higher than from former corn-soy. The reduction of GHG efflux from converting corn-soy to energy cane was likely due to the perennial nature of energy cane because some biomass is left on the field to regenerate resulting in less soil disturbance compared to corn, which is re-planted annually. The higher rates of GHG emissions from former pasture was attributable to pasture lands having little history of soil disturbance, which were then necessarily tilled to facilitate energy cane production. Mechanical disturbance of the soil increases oxygen in pore spaces deeper in the soil profile and promotes heterotrophic respiration, inducing a flux of CO$_2$ to the atmosphere (Paustian et al. 2000, Guo and Gifford 2002).

Our third hypothesis, that converting traditional agriculture to energy cane would result in substantial net GHG reductions was supported by our simulations. These reductions were driven by irrigation practices and fertilization. Irrigating energy cane to field capacity every month of the growing season increased yield, but at the expense of increased GHG emissions. Non-irrigated energy cane approached 90% of the potential ethanol yield as irrigated cane (Table 2), while producing 100,000 fewer tons of GHG emissions in our simulations (Table 2). Consistent water availability likely drove production increases in the model simulations, but in concert with reasonably high levels of inorganic N fertilization (~200 kg·ha^{-1}·yr^{-1}), irrigation also creates a favorable environment for N$_2$O pro-

duction because residual inorganic N can be transformed into gaseous N species via microbial nitrification/denitrification pathways (Schlesinger 1997). Irrigation also depressed CH_4 oxidation, leading to greater aggregate GHG emissions from irrigated energy cane compared to non-irrigated cane.

Another large-scale modeling effort in the rainfed Midwestern US that simulated converting traditional agriculture to biofuel crops (switchgrass and *Miscanthus*), observed a shift to that region becoming a net GHG sink (Davis et al. 2012). *Miscanthus* is similar to energy cane in some respects, as both are large, fast growing tropical C_4 grasses, but energy cane has high N requirements (Muchovej and Newman 2004) and the high fertilization rates we simulated to achieve maximum biomass would exacerbate N_2O emissions for the reasons stated above (Matson et al. 1996).

Scaling our results to the entire SE USA region suitable for energy cane shows that converting a portion of both grazed pasture and corn-soy results in a GHG emission benefit of >4 million Mg CO_2eq yr^{-1} (Table 2), or to use another metric, the equivalent of avoiding the consumption of ~1.7 billion liters of petroleum gasoline (Environmental Protection Agency 2015).

This study is not a full life cycle analysis of land use conversion from traditional agriculture to energy cane (Adler et al. 2007), and thus does not consider the full economic and industrial feasibility of converting nearly 1 million hectares of land from traditional agriculture to energy cane production. However, our results suggest that such an undertaking could meet demand for over a quarter of the US ethanol mandate per the 2022 EISA (US Congress 2007). From an environmental standpoint, this land use conversion will potentially reduce N leaching into the Gulf of Mexico and Atlantic coastal areas of the SE USA. Furthermore, converting a portion of traditional agriculture to energy cane in this region will reduce GHG emissions, maintains the proportion of corn-soy hectares devoted to food, and 90% of grazed pasture, intact in the SE USA.

Acknowledgments

Literature Cited

Adler, P. R., S. J. Del Grosso, and W. J. Parton. 2007. Life-cycle assessment of net greenhouse-gas flux for bioenergy cropping systems. Ecological Applications 17:675–691.

Alexander, A. G. 1985. The energy cane alternative. Elsevier, Amsterdam, Netherlands.

Allen, D. E., G. Kingston, H. Rennenberg, R. C. Dalal, and S. Schmidt. 2010. Effect of nitrogen fertilizer management and waterlogging on nitrous oxide emission from subtropical sugarcane soils. Agriculture Ecosystems and Environment 136:209–217.

Anderson-Teixeira, K. J., B. D. Duval, S. P. Long, and E. H. DeLucia. 2012. Biofuels on the landscape: Is "land sharing" preferable to "land sparing"? Ecological Applications 22:2035–2048.

Arndt, C., J. M. Powell, M. Aguerre, and M. M. Wattiaux. 2015. Performance, digestion, nitrogen balance, and emission of manure ammonia, enteric methane, and carbon dioxide in lactating cows fed diets with varying alfalfa silage-to-corn silage ratios. Journal of Dairy Science 98:418–430.

Cheng, K. K., B. Y. Cai, J. A. Zhang, H. Z. Ling, Y. J. Zhou, J. P. Ge, and J. M. Xu. 2008. Sugarcane bagasse hemicellulose hydrolysate for ethanol production by acid recovery process. Biochemical Engineering Journal 38:105–109.

Crawley, M. J. 2007. The R book. John Wiley and Sons, West Sussex, UK.

David, M. B., L. E. Drinkwater, and G. E. McIsaac. 2010. Sources of nitrate yields in the Mississippi River basin. Journal of Environmental Quality 39:1657–1667.

Davis, S. C., W. J. Parton, S. J. Del Grosso, C. Keough, E. Marx, P. Adler, and E. H. DeLucia. 2012. Impacts of second-generation biofuel agriculture on greenhouse gas emissions in the corn-growing regions of the US. Frontiers in Ecology and the Environment 10:69–74.

Davis, S. C., W. J. Parton, F. G. Dohleman, C. M. Smith, S. Del Grosso, A. D. Kent, and E. H. DeLucia. 2010. Comparative biogeochemical cycles of bioenergy crops reveal nitrogen-fixation and low greenhouse gas emissions in a *Miscanthus × giganteus* agro-

ecosystem. Ecosystems 13:144–156.

Del Grosso, S. J., S. M. Ogle, W. J. Parton, and F. J. Breidt. 2010. Estimating uncertainty in N₂O emissions from U.S. cropland soils. Global Biogeochemical Cycles 24:GB1009.

Del Grosso, S. J., D. S. Ojima, W. J. Parton, A. R. Mosier, G. A. Peterson, and D. S. Schimel. 2002. Simulated effects of non-irrigated cropping intensification on soil organic matter and GHG exchanges using the DAYCENT ecosystem model. Environmental Pollution 116:S75–S83.

Del Grosso, S. J., W. J. Parton, A. R. Mosier, M. K. Walsh, D. S. Ojima, and P. E. Thornton. 2006. DAYCENT national scale simulations of N₂O emissions from cropped soils in the USA. Journal of Environmental Quality 35:1451–1460.

Donner, S. D., and C. J. Kucharik. 2008. Corn-based ethanol production compromises goal of reducing nitrogen export by the Mississippi River. Proceedings of the National Academy of Sciences USA 105:4513–4518.

Duff, S. J. B., and W. D. Murray. 1996. Bioconversion of forest products industry waste cellulosics to fuel ethanol: a review. Bioresource Technology 55:1–33.

Duval, B. D., K. J. Anderson-Teixeira, S. C. Davis, C. Keogh, S. P. Long, W. J. Parton, and E. H. DeLucia. 2013. Predicting greenhouse gas emissions and soil carbon from changing pasture to an energy crop. PLoS ONE 8(8):E72019.

Environmental Protection Agency. 2015. Energy and the Environment. http://www2.epa.gov/energy

Fargione, J., R. Plevin, and J. Hill. 2010. The ecological impact of biofuels. Annual Review of Ecology Evolution and Systematics 41:351–377.

Federal Register. 2012. No. 58, Regulation of fuels and fuel additives: 2012 renewable fuel standards, EPA, final rule. https://www.federalregister.gov

Galdos, M. V., C. C. Cerri, C. E. P. Cerri, K. Paustian, and R. Van Antwerpen. 2009. Simulation of soil carbon dynamics under sugarcane with the CENTURY model. Soil Science Society of America Journal 73:802–811.

Galdos, M. V., C. C. Cerri, C. E. P. Cerri, K. Paustian, and R. Van Antwerpen. 2010. Simulation of sugarcane residue decomposition and aboveground growth. Plant and Soil 326:243–259.

Gilbert, R. A., J. M. Shine, Jr., J. D. Miller, R. W. Rice, and C. R. Rainbol. 2006. The effect of genotype, environment and time of harvest on sugarcane yields in Florida, USA. Field Crops Research 95:156–170.

Glaz, B., and D. R. Morris. 2010. Sugarcane responses to water-table depth and periodic flood. Agronomy Journal 102:372–380.

Glaz, B., and M. F. Ulloa. 1993. Sugarcane yields from plant and ratoon sources of seed cane. Journal of the American Society of Sugar Cane Technologists

13:7–13.

Greenland, D. 2005. Climate variability and sugarcane yield in Louisiana. Journal of Applied Meteorology 44:1655–1666.

Guo, L. B., and R. M. Gifford. 2002. Soil carbon stocks and land use change: a meta analysis. Global Change Biology 8:345–360.

Hartman, M. D., E. R. Merchant, W. J. Parton, M. P. Gutmann, S. M. Lutz, and S. A. Williams. 2011. Impact of historical land-use changes on greenhouse gas exchange in the U.S. Great Plains, 1883–2003. Ecological Applications 21:1105–1119.

Heaton, E. A., F. G. Dohleman, and S. P. Long. 2008. Meeting US biofuel goals with less land: the potential of Miscanthus. Global Change Biology 14:2000–2014.

Hoy, J. W., M. P. Grisham, and K. E. Damann. 1999. Spread and increase of ratoon stunting disease of sugarcane and comparison of disease detection methods. Plant Disease 83:1170–1175.

Hudiburg, T. W., S. C. Davis, W. Parton, and E. H. DeLucia. 2014. Bioenergy crop greenhouse gas mitigation potential under a range of management practices. Global Change Biology Bioenergy 7:366–374.

James, E. K., and F. L. Olivares. 1997. Infection and colonization of sugarcane and other graminaceous plants by endophytic diazotrophs. Critical Reviews in Plant Sciences 17:77–119.

Kim, M., and D. F. Day. 2011. Composition of sugarcane, energy cane and sweet sorghum suitable for ethanol production at Louisiana sugar mills. Journal of Industrial Microbiology and Biotechnology 38:803–807.

Kittel, T. G. F., et al. 2004. VEMAP phase 2 bioclimatic database. I. Gridded historical (20th century) climate for modeling ecosystem dynamics across the conterminous USA. Climate Research 27:151–170.

Mark, T., P. Darby, and M. Salassi. 2009. Energy cane usage for cellulosic ethanol: estimation of feedstock costs. Southern Agricultural Economics Association Annual Meeting, Atlanta, Georgia, 31 January–3 February 2009. http://ageconsearch.umn.edu/handle/43497

Matson, P. A., C. Billow, S. Hall, and J. Zachariassen. 1996. Fertilization practices and soil variations control nitrogen oxide emissions from tropical sugar cane. Journal of Geophysical Research 101(D13):18533–18545.

Matsuoka, S., A. J. Kennedy, E. G. D. dos Santos, A. L. Tomazela, and L. C. S. Rubio. 2014. Energy cane: its concept, development, characteristics, and prospects. Advances in Botany 2014:597275.

Morris, D. R., and R. A. Gilbert. 2005. Inventory, crop use, and soil subsidence of Histosols in Florida. Journal of Food Agriculture and Environment

3:190–193.

Muchovej, R. M., and P. R. Newman. 2004. Nitrogen fertilization of sugarcane on a sandy soil II: soil and groundwater analysis. Journal American Society Sugar Cane Technologists 24:225–240.

NASS. 2012. Quick Stats. USDA National Agricultural Statistics Service. http://quickstats.nass.usda.gov/results/2605081D-D5DC-3489-8CBE53319BCF41FD

National Corn Growers Association. 2012. World of corn. http://www.ncga.com/worldofcorn

Natural Resources Conservation Service. 2010. Web soil survey. http://websoilsurvey.sc.egov.usda.gov/App/HomePage.htm

Parton, W. J., P. J. Hanson, C. Swanston, M. Torn, S. E. Trumbore, W. Riley, and R. Kelly. 2010. ForCent model development and testing using the Enriched Background Isotope Study experiment. Journal of Geophysical Research 115:G04001.

Parton, W. J., M. D. Hartman, D. S. Ojima, and D. S. Schimel. 1998. DAYCENT and its land surface submodel: description and testing. Global and Planetary Change 19:35–48.

Paustian, K., J. Six, E. T. Elliott, and H. W. Hunt. 2000. Management options for reducing CO_2 emissions from agricultural soils. Biogeochemistry 48:147–163.

Rabalais, N. N., R. E. Turner, and W. J. Wiseman. 2002. Gulf of Mexico hypoxia, a.k.a. "The dead zone.". Annual Review of Ecology and Systematics 33:235–263.

R Development Core Team. 2013. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Schlesinger, W. H. 1997. Biogeochemistry: an analysis of global change. Academic Press, San Diego, California, USA.

Sladden, S. E., D. I. Bransby, G. E. Aiken, and G. M. Prine. 1991. Biomass yield and composition, and winter survival of tall grasses in Alabama. Biomass

and Bioenergy 1:123–127.

Somerville, C., H. Youngs, C. Taylor, S. C. Davis, and S. P. Long. 2010. Feedstocks for lignocellulosic biofuels. Science 329:790–792.

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

Tilman, D., R. Socolow, J. A. Foley, J. Hill, E. Larson, L. Lynd, S. Pacala, J. Reilly, T. Searchinger, C. Somerville, and R. Williams. 2009. Beneficial biofuels-the food, energy and environment trilemma. Science 325:270–271.

United States Congress. 2007. The Energy Independence and Security Act of 2007 (H.R. 6). https://www.congress.gov/bill/110th-congress/house-bill/6/text

USDA Economic Research Service (USDA-ERS). 2012. http://www.ers.usda.gov/data-products/us-bioenergy-statistics.aspx

United States Energy Information Administration. 2012. http://www.eia.gov/todayinenergy/detail.cfm?id=4110#

Valentine, D. W., E. A. Holland, and D. S. Schimel. 1994. Ecosystem and physiological controls over methane production in northern wetlands. Journal of Geophysical Research 99:563–1571.

Vallis, I., W. J. Parton, B. A. Keating, and A. W. Wood. 1996. Simulation of the effects of trash and N fertilizer management on soil organic matter levels and yields of sugarcane. Soil and Tillage Research 38:115–132.

Vaquer-Sunyer, R., and C. M. Duarte. 2008. Thresholds of hypoxia for marine biodiversity. Proceedings of the National Academy of Sciences USA 105:15452–15457.

Wiedenfeld, B., and J. Enciso. 2008. Sugarcane responses to irrigation and nitrogen in semiarid south Texas. Agronomy Journal 100:665–671.

SUPPLEMENTAL MATERIAL

ECOLOGICAL ARCHIVES

Appendices A and B are available online: http://dx.doi.org/10.1890/ES15-00546.1.sm