Ecological Applications, 21(4), 2011, pp. 1055–1067

The biogeochemistry of bioenergy landscapes: carbon, nitrogen, and water considerations

G. Philip Robertson,[1,2,6] Stephen K. Hamilton,[1,3] Stephen J. Del Grosso,[4,5] and William J. Parton[4]

[1]*W. K. Kellogg Biological Station, Michigan State University, Hickory Corners, Michigan 49060 USA,* and
DOE Great Lakes Bioenergy Research Center, Michigan State University, East Lansing, Michigan 48824 USA
[2]*Department of Crop and Soil Sciences, Michigan State University, East Lansing, Michigan 48824 USA*
[3]*Department of Zoology, Michigan State University, East Lansing, Michigan 48824 USA*
[4]*Natural Resource Ecology Laboratory, Colorado State University, Fort Collins, Colorado 80521 USA*
[5]*Agricultural Research Service, U.S. Department of Agriculture, Fort Collins, Colorado 80512 USA*

Abstract. The biogeochemical liabilities of grain-based crop production for bioenergy are no different from those of grain-based food production: excessive nitrate leakage, soil carbon and phosphorus loss, nitrous oxide production, and attenuated methane uptake. Contingent problems are well known, increasingly well documented, and recalcitrant: freshwater and coastal marine eutrophication, groundwater pollution, soil organic matter loss, and a warming atmosphere. The conversion of marginal lands not now farmed to annual grain production, including the repatriation of Conservation Reserve Program (CRP) and other conservation set-aside lands, will further exacerbate the biogeochemical imbalance of these landscapes, as could pressure to further simplify crop rotations.

The expected emergence of biorefinery and combustion facilities that accept cellulosic materials offers an alternative outcome: agricultural landscapes that accumulate soil carbon, that conserve nitrogen and phosphorus, and that emit relatively small amounts of nitrous oxide to the atmosphere. Fields in these landscapes are planted to perennial crops that require less fertilizer, that retain sediments and nutrients that could otherwise be transported to groundwater and streams, and that accumulate carbon in both soil organic matter and roots. If mixed-species assemblages, they additionally provide biodiversity services.

Biogeochemical responses of these systems fall chiefly into two areas: carbon neutrality and water and nutrient conservation. Fluxes must be measured and understood in proposed cropping systems sufficient to inform models that will predict biogeochemical behavior at field, landscape, and regional scales. Because tradeoffs are inherent to these systems, a systems approach is imperative, and because potential biofuel cropping systems and their environmental contexts are complex and cannot be exhaustively tested, modeling will be instructive. Modeling alternative biofuel cropping systems converted from different starting points, for example, suggests that converting CRP to corn ethanol production under conventional tillage results in substantially increased net greenhouse gas (GHG) emissions that can be only partly mitigated with no-till management. Alternatively, conversion of existing cropland or prairie to switchgrass production results in a net GHG sink. Outcomes and policy must be informed by science that adequately quantifies the true biogeochemical costs and advantages of alternative systems.

Key words: bioenergy; biofuels; cellulosic biomass; climate mitigation; climate stabilization; ecosystem modeling; global warming potential; greenhouse gases; nitrogen; nitrous oxide; soil carbon; water quality.

Introduction

U.S. corn-based ethanol production has increased dramatically in the past year and promises to continue to consume a substantial fraction of U.S. corn production in coming decades. In 2009, over 30% of total U.S. production (U.S. Department of Agriculture 2010) was used to produce some 41 billion liters (10.8 billion gallons) of ethanol (U.S. Department of Energy 2010). By 2015 well over 50% of today's crop will be needed to meet the 57 billion liter (15 billion gallon) mandate embedded in the U.S. Energy Independence and Security Act of 2007. Market prices have responded accordingly, with biofuel feedstock demand responsible for about one-third of the 37% increase in corn prices in 2007 (Lazear 2008).

With rising prices comes rising demand for additional crop production, and rising pressure to farm existing cropland more intensively and to bring idled farmland—

Manuscript received 18 March 2009; revised 19 January 2010; accepted 5 February 2010; final version received 1 March 2010. Corresponding Editor: A. R. Townsend. For reprints of this Invited Feature, see footnote 1, p. 1037.

[6] E-mail: robertson@kbs.msu.edu

either abandoned from agriculture or in conservation set-aside programs—back into production (Secchi et al. 2008). The environmental implications of such changes are substantial (e.g., Fargione et al. 2008): the conversion of marginal lands to grain-based agriculture or the further intensification of agriculture on marginal lands will only exacerbate the environmental imbalances associated with intensive cropping (Howarth et al. 2009).

Biogeochemistry is but one piece of a biofuel sustainability puzzle that includes economic, environmental, and social components. But of the environmental pieces, biogeochemistry is central: if we don't get the biogeochemistry right then a major rationale for developing a so-called bioeconomy—climate stabilization—goes away. And there are other biogeochemical pieces that come into play as well, most notably those related to nitrogen and phosphorus cycling, with the potential for further environmental harm from agriculture to inland surface waters and to marine coastal zones (Costello et al. 2009), and to further greenhouse gas loading of the atmosphere via indirect emissions (e.g., nitrous oxide emissions downstream of agricultural fields) (Crutzen et al. 2008).

For all the same reasons that intensive grain-based food crops are environmentally challenged, then, grain-based biofuels are equally problematic. In fact to the extent that economic pressures lead to intensified production on existing farmland or bring abandoned farmland back into production, biofuel crops will exacerbate existing problems. Land that has not been profitable to farm in recent years becomes profitable as grain prices climb substantially, and by definition this lower-fertility land is less capable of buffering the inputs required to produce high yields. So as we see U.S. corn acreage increasing at the expense of land now enrolled in set-aside programs such as the Conservation Reserve Program (CRP), we can expect to see amplified the environmental signals that we've been trying hard to attenuate over the past several decades: soil erosion (Lal 1998), marine hypoxia (Donner and Kucharik 2008), atmospheric nitrous oxide loading (Crutzen et al. 2008), and rising groundwater nitrate concentrations (Nolan et al. 1997), to name a few. Add to this the carbon debt associated with tropical land conversion in response to higher commodity prices (Fargione et al. 2008, Gibbs et al. 2008), and the biogeochemical cost of grain-based biofuels begins to substantially offset their putative CO_2 benefit. Increasingly, grain-based biofuels are seen only as a transitional step to cellulosic fuels (National Research Council 2009).

Cellulosic biofuels provide the potential for an alternative outcome: agricultural landscapes that accumulate carbon, that conserve nitrogen, and that do not much alter fluxes of the non-CO_2 greenhouse gases. Done right (Robertson et al. 2008, National Research Council 2009), cellulosic biofuels can provide biogeochemical and biodiversity (Fletcher et al. 2010) benefits not now realized in most agricultural landscapes. Mostly these benefits stem from the fact that cellulose biorefineries can be designed to accept a wide variety of feedstocks, and in particular biomass from perennial vegetation. This provides the potential for improved environmental performance relative to grain-based systems with their substantially more-leaky nitrogen cycle and reliance on external chemical subsidies (Robertson and Vitousek 2009).

The current drawback to cellulosic refineries is expense: harvested biomass requires heat, acid, or other pretreatment to expose cellulose and hemicellulose to enzymes that convert these complex biopolymers to fermentable sugars. At present the expense of pretreatment and enzymes hinders large-scale commercialization, but costs are expected to come down with further research advances, and financial incentives in the 2008 U.S. Farm Bill will speed deployment, as will higher oil and grain prices. The U.S. Energy Independence and Security Act of 2007 mandates 80 billion liters (21 billion gallons) of cellulosic or other non-grain sources of ethanol by 2022; this together with the grain-based ethanol mandate of 57 billion liters (15 billion gallons) represents about 25% of 2008 U.S. gasoline consumption of 511 billion liters (135 billion gallons; U.S. Department of Energy 2010). U.S. Environmental Protection Agency (2007) and National Resources Defense Council (2007) project a U.S. ethanol demand of about 300 billion liters (80 billion gallons) by 2050 as part of a strategic portfolio to eliminate gasoline use in the U.S. transportation sector (lowered use of gasoline through efficiency and smart growth are also key to reaching this goal).

Apart from the potential for liquid fuel production, cellulosic biomass can also be burned directly to produce electricity or heat (Richter et al. 2009), with substantial efficiency savings as no energy is lost to biorefining processes (Samson et al. 2008) nor, if used to power electric vehicles, to internal combustion in vehicle engines (Ohlrogge et al. 2009). These more efficient uses of biomass should be encouraged (Howarth et al. 2009) and may eventually displace the need for liquid fuels, but will not displace the need for biomass sustainably produced. Moreover, Wise et al. (2009) note that even in the absence of the value of biomass for energy production, following the successful development of carbon capture and storage (CCS) technologies, biomass combustion + CCS may be the most efficient means for drawing down atmospheric CO_2 levels in the future.

Biomass is thus likely to be a major part of any future U.S. energy portfolio, and two questions logically follow: how would our agricultural portfolio need to change to meet the demand for cellulosic feedstock, and what are the likely biogeochemical repercussions?

Sources of Cellulosic Feedstock

If 57 billion liters of the projected demand were provided by grain-based ethanol (the 2007 Energy

Independence and Security Act production cap), then the remainder of the 300 billion liters of projected U.S. demand would need to be met by cellulosic ethanol. Commercial ethanol yields from cellulosic feed stocks are expected to be ~0.4 L/kg biomass (Renewable and Appropriate Energy Laboratory 2007); current yields in pilot plants are ~0.3 L/kg. To produce 243 billion liters, then, will eventually require $\sim 608 \times 10^6$ metric tons (608×10^6 Mg or 608 Tg) biomass.

Perlack et al. (2005) estimate that 109×10^6 metric tons of forest products are currently available for conversion to ethanol. This includes 41×10^6 metric tons of logging residues (50–65% of residues generated annually); 60×10^6 metric tons of forest thinnings for fuel reduction (15–20% of that available over a 30-yr average rotation, or 0.5–0.7% per year); and 8×10^6 metric tons of 161×10^6 metric tons of mill residue byproducts, most of which are now used on-site for energy and other uses. Because the use of logging residues and forest thinnings for biofuel may have little impact on atmospheric CO_2 mitigation—removing residues can reduce soil carbon stores and removing trees releases carbon that would otherwise (in the absence of fire) remain sequestered in biomass for many years—the biogeochemical value of these feedstocks is questionable, and may not be eligible for carbon credit accounting (Searchinger et al. 2009).

Municipal solid waste, on the other hand, is residue that would otherwise be incinerated or landfilled, such that burning it to offset fossil fuel use would have clear CO_2 benefits. The National Research Council (2009) estimates that, in the United States, 90×10^6 metric tons could be available for biofuel use, of a 140×10^6 metric tons total.

Likewise, some proportion of annual crop residue would be expected to decompose over the year following crop harvest, and this too would be a creditable biofuel if instead it were used to offset fossil fuel use. Graham et al. (2007) estimate that about 110×10^6 metric tons of corn stover could be harvested without risk of erosion from a national corn crop that yields $\sim 196 \times 10^6$ metric tons of grain. This represents about 55% of the stover produced by the record 2005 U.S. corn crop, and assumes widespread no-till adoption. Wilhelm et al. (2007) question that this level of stover retention is sufficient to maintain soil carbon levels, and the National Research Council (2009) suggests that, at most, 76×10^6 metric tons of stover can be removed without harming soil carbon stocks. Wilhelm et al. (2007) estimate more conservatively that for no-till soils, over eight times more stover is needed to protect against carbon loss than to protect against soil erosion.

Subtracting from total biomass needed the available wood waste (109×10^6 metric tons) and a more conservative 50% (55×10^6 metric tons) of Graham et al.'s. (2007) estimate of available stover leaves a remaining feedstock need of 354×10^6 metric tons biomass. If it is shown that more logging residue and stover needs to be retained on-site to protect against soil carbon loss, and if forest thinnings for fire reduction become ineligible for carbon credits, this value could be as large as 518×10^6 metric tons.

Even at moderately high rates of aboveground net primary productivity, the land needed to provide this much feedstock is substantial, and the potential biogeochemical impact is correspondingly significant. For example, average on-farm rates of switchgrass (*Panicum virgatum*) production in the northern Great Plains are ~7.5 metric tons/ha (3.2 tons/acre; Schmer et al. 2008), which translates to an annual harvest requirement of over 47×10^6 ha in order to provide 354×10^6 metric tons biomass. This represents about 25% of current U.S. cropland (178×10^6 ha; Economic Research Service 2008), which in any case would compete with food production, and most grassland/rangeland (240×10^6 ha) is too arid to be this productive without irrigation.

Alternatively, fallow lands could be brought back into production, including $\sim 24 \times 10^6$ ha of cropland that appears to have reverted to secondary succession since 1982 (Economic Research Service 2005) and the 15×10^6 ha of cropland now enrolled in CRP. Neither of these sources would compete with current food production, and their abandoned or set-aside status implies lower inherent fertility and a greater environmental vulnerability were they to be used for grain production in the future. Other land in nonnative vegetation on degraded soils or on abandoned crop or pastureland might be equally suitable for cellulosic feedstock production, whether with grasses and other herbaceous plants or with short rotation trees such as poplars and willows, and a comprehensive analysis awaits attention.

Climate Forcing Considerations

Calculations of energy return on investment (EROI) provide a rough measure of carbon neutrality. EROI is the ratio of energy in a quantity of renewable fuel to the non-reusable energy required to produce it, and represents the amount of net new energy provided by a specific energy source. Grain-based ethanol, for example, yields on average about 1.4 MJ of ethanol for every 1 MJ petroleum invested in its manufacture (Table 1; Hammerschlag 2006). Put in carbon terms, 1 kg of fossil-fuel C is required to produce 1.4 kg of ethanol C offset (or put another way, a liter of corn-grain ethanol contains 71% fossil carbon equivalents). In this calculation are embedded most of the carbon costs of ethanol manufacture, ranging from the diesel needed to power tractors to the natural gas needed to heat fermentation vats. Disparities in this value (recent estimates range from <1 [Pimentel and Patzek 2005] to >1.6 [Kim and Dale 2005]) generally result from the degree to which different authors bound the system or credit residual co-products such as dry distillers grain (Farrell et al. 2006). In any given cropping system, however, there is also wide latitude in the extent to

TABLE 1. Energy return on investment (EROI; see *Climate forcing considerations*) and net energy yield for ethanol produced from corn grain and cellulosic feedstocks.

Feedstock	EROI		Net energy yield (MJ/L)	
	Range	Median	Range	Median
Corn grain	0.8–1.65	1.4	−6.1–8.9	3.9
Cellulosic biomass	0.7–6.6	4.5		23

Notes: EROI values are as compiled by Hammerschlag (2006) for six corn grain studies and four cellulosic feedstock studies. Net energy yields are for the seven corn grain scenarios and one cellulosic scenario analyzed by Farrell et al. (2006); hence no range is available for cellulosic biomass.

which different practices affect the system's net carbon cost (Robertson et al. 2000), as discussed further below.

In contrast to grain-based ethanol, EROI for cellulosic ethanol (Hammerschlag 2006) is ~4.5 (Table 1): an investment of 1 unit of fossil energy yields about 5 units of new energy. Put another way, cellulosic ethanol contains about 20% fossil fuel equivalents. The positive biogeochemical impact is correspondingly high: 5 kg of atmospheric carbon offsets are generated for every kg of fossil carbon invested in its production. Much of the carbon efficiency of cellulosic systems is due to lower use of fuel, fertilizer, and other agronomic inputs plus a lignin co-product that can be burned to power biorefinery operations and to generate excess electricity.

EROI includes only those carbon costs that are energy related. Other carbon fluxes can also be significant. Soil carbon change, for example, can be a more important flux of C than fuel use in grain-based cropping systems (Robertson et al. 2000); in no-till systems the accumulation of soil C can more than offset the C in diesel fuel used to plant, spray, fertilize, and harvest the crop. Most cellulosic crops are perennial and thus no-till by definition; additionally, significant carbon can accumulate in perennial, living roots (see Plate 1).

Agricultural lime (mainly calcium and magnesium carbonates) used as a soil amendment represents another anthropogenic C flux in most row crop systems (West and McBride 2005). As the added carbonates are consumed in soils, some portion is released as CO_2 (Hamilton et al. 2007). The magnitude of this CO_2 loss varies by soil type and liming frequency, but can rival that from fuel use (Robertson et al. 2000). Alternatively, at higher soil pH levels, liming can cause net sequestration of CO_2 in the form of bicarbonate alkalinity generated by carbonic acid dissolution of the lime and its transport to groundwater reservoirs. At the landscape scale, CO_2 sequestration from the generation of alkalinity in some soils may offset the CO_2 released when carbonates are attacked by strong acids in others, but our knowledge of the relative importance of these reactions is too incomplete to generalize at this point (Hamilton et al. 2007).

Non-CO_2 greenhouse gases also play a major role in cropping system greenhouse gas budgets. Both nitrous oxide (N_2O) and methane are more potent greenhouse gases than CO_2 (Intergovernmental Panel on Climate Change 2007), and for both gases agriculture is a major global source of anthropogenic fluxes (Robertson 2004). In annual grain crops, N_2O production can be the single greatest source of radiative forcing (Robertson et al. 2000, Mosier et al. 2005).

Annual N_2O flux rates can differ by a factor of two between annual and perennial cropping systems (Fig. 1), which is another reason that cellulosic biofuels can result in substantially less climate forcing than grain-based systems. Indeed, for the major grain-based biofuel crops, Crutzen et al. (2008) have estimated that consideration of N_2O emissions alone could largely negate any global warming reductions based on C balances. Melillo et al. (2009) warn that if cellulosic crops were to be intensively fertilized, resultant increases in N_2O emissions could negate any climate forcing benefits from improved C balances.

Additionally, if land is used for biofuels that would otherwise be used for food or fiber production, forcing land elsewhere to be cleared and planted to make up for lost production, the indirect climate forcing produced by that new land use change must also be factored in the system's net climate forcing (Fargione et al. 2008,

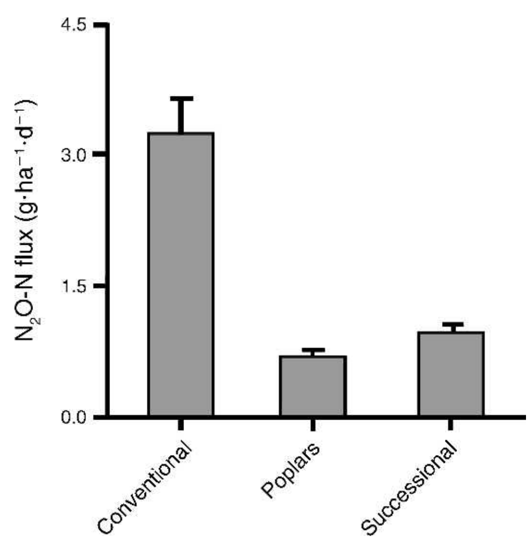

FIG. 1. N_2O production in a conventional corn–soybean–wheat rotation, a poplar biofuel crop, and early successional vegetation at a site in Michigan, USA (Robertson et al. 2000, Grandy et al. 2006). Error bars show +SE.

TABLE 2. Radiative forcing costs of field crop activities at a northern corn belt location.

Cropping system	Soil carbon change	Fuel use	N-fertilizer production	Lime dissolution	N_2O	CH_4	Net balance
Grain-based							
Corn–soybean–wheat	0	16	27	1	52	−4	102
Cellulosic							
Poplar (*Populus* sp.) trees	−117	2	5	0	10	−5	−105
Early successional vegetation	−220	2	0	0	15	−6	−211

Notes: All units are (g CO_2 equivalents)$\cdot m^{-2} \cdot yr^{-1}$. A negative net balance indicates greenhouse gas (GHG) mitigation (more CO_2 equivalents are sequestered than emitted). The table is from Robertson et al. (2000).

Searchinger et al. 2008). Carbon lost from soil and vegetation, including the lost opportunity for future sequestration (Field et al. 2008), in addition to changes in N_2O fluxes (Melillo et al. 2009), can readily turn even cellulosic biofuel systems from a net CO_2 sink to a net CO_2 source. To avoid these indirect effects requires planting biofuel crops on land that is not now used for food or timber production.

The different GHG footprints of annual and perennial cropping systems become most apparent in whole-system analyses. Table 2 presents one such analysis for a site in the northern portion of the U.S. corn belt. The threefold difference in net radiative forcing among these cropping systems, one grain-based and two cellulosic, from 102 to −211 g CO_2-equivalents$\cdot m^{-2} \cdot yr^{-1}$, suggests substantial latitude for managing the climate forcing of biofuel production systems.

Major differences between the grain and cellulosic systems are due principally to energy use (fuel and N fertilizer), soil carbon change, and N_2O emission. In cellulosic systems the recovery of soil organic carbon following the establishment of perennial vegetation (the converse of carbon debt associated with indirect land use change) is a substantial factor contributing to the negative GHG footprint of these systems. No-till cultivation can offset some of the climate forcing associated with grain-based systems but not to the same degree as can woody or mixed-species communities (Robertson et al. 2000, West and Marland 2003).

However, not all cellulosic systems will accumulate soil carbon. Annual grain crops for which all or most aboveground residue is removed for cellulosic production—corn stover for example—will not accumulate carbon even under no-till cultivation unless the soil is highly depleted of carbon due to many years of conventionally tilled low input rotations. And as noted earlier, the amount of residue necessary to preserve or enhance soil carbon stores (Wilhelm et al. 2007) is likely to be substantially greater than that necessary for erosion control (Graham et al. 2007). The capacity of a given soil to withstand removal of nearly all of the aboveground biomass production will depend on a variety of factors that are imperfectly understood. Tillage and crop characteristics such as perenniality, species and rotational diversity, and rooting depth are among factors likely to be most important.

Nitrogen Cycle Considerations

Agriculture's impact on the amount of excess reactive nitrogen circulating in the environment are well recognized and documented (Galloway et al. 2008). As noted above, biofuel cropping systems managed in a manner similar to food crops will only contribute to this excess. Greater nitrate loss to groundwater and coastal marine zones and more nitrous oxide emission to the atmosphere are two immediate consequences of intensive grain production. While with appropriate management these impacts can be lessened (Robertson and Vitousek 2009), they cannot be avoided with current technology.

Poor nitrogen retention in annual cropping systems is mainly related to the low nitrogen use efficiency of most grain crops, to their high nitrogen demand, and in temperate climates to the absence of plants and plant N uptake during most of the year (Robertson 1997). Nitrogen conservation can be improved by including in the rotation winter cover crops that can capture soil N that would otherwise be lost following crop senescence, and by advanced fertilizer management (Robertson and Vitousek 2009).

Perennial cellulosic crops achieve high nutrient conservation by providing year-round cover. Lower nitrogen demand also contributes to a more closed N cycle: vegetative tissue contains much less N than grain with its high protein content. Perenniality additionally provides the opportunity for nutrient retranslocation prior to harvest; in contrast to annual crops, nitrogen and other nutrients can be retranslocated to roots prior to leaf senescence and thus immobilized until used for new leaf and stem production at the start of the next growing season. And the presence of a mature root system at the beginning of the growing season helps to ensure that little if any fertilizer nitrogen that may be added escapes the rooting zone. Thus, if fertilized at or near nutrient-replacement rates, perennial cellulosic crops are likely to be very nutrient conservative.

Few comparative studies are available to confirm high N retention in perennial cellulosic systems. Patterns of N_2O flux noted in Table 2 for a site in the upper Midwest are consistent with this notion, as are recent N leaching results from the same system. Over an 11 year period Syswerda et al. (submitted) found nitrate losses from poplar tree and early successional systems that were negligible in contrast to rates of $\sim$65 kg $N \cdot ha^{-1} \cdot yr^{-1}$ in a conventionally managed annual grain system.

Agricultural Water Use and Effects on Water Quality

Most biofuel cropping systems in place today affect water cycling and downstream water quality in a manner analogous to when these kinds of crops are grown for food production. In general, where intensive methods are employed to cultivate annual crops in temperate climates, there usually are long periods with little or no plant uptake. In humid regions these periods can result in elevated surface runoff, soil erosion, nutrient loss, and groundwater contamination by agrichemicals (Nolan et al. 1997, Kort et al. 1998, Böhlke 2002, Alexander et al. 2008). Increased area and intensity of row-crop production to meet demands for both biofuels and food are projected to exacerbate existing water quality issues, including the problem of eutrophication and consequent oxygen depletion in marine coastal zones (Donner and Kucharik 2008, Simpson et al. 2009).

Long-term soil salinization can also be a problem related to crop water use in semi-arid and semi-humid landscapes, either related to irrigation or water table changes where crops replace more deeply rooted woodlands or forests. If cultivation of crops for biofuels directly or indirectly causes agricultural activity to expand into drylands that need irrigation or are otherwise at risk of salinization, then the attendant impacts of biofuel crop production on water resources become magnified over those associated with conventional food production (de Fraiture and Berndes 2009).

Biofuel crop production can also drive regional shifts in land allocation towards one conventional grain crop over another, as, for example, from soybeans to corn in the United States, with positive, neutral, or negative implications for water use and quality depending on the regional setting (National Research Council 2007, Donner and Kucharik 2008, Domínguez-Faus et al. 2009). Also, as noted in *Sources of cellulosic feedstock*, harvest of corn stover for cellulosic biofuel production could exacerbate the impact of corn on water quality unless careful precautions are in place to ensure that this practice does not increase overland runoff and soil erosion, leading to greater nutrient and sediment loading into waterways.

Perennial cropping systems currently considered as candidates for biofuel production likely would affect water resources differently than intensive row crops, generally with reduced potential for soil erosion and nutrient leaching, although this may not necessarily be the case if these perennial crops are established on marginal lands that are more environmentally sensitive. Compared to grain-based biofuel production by crops such as corn or canola, perennial cropping systems may require far lower inputs of fertilizers and pesticides, if any, per unit of net energy gain (European Environment Agency 2006, Tilman et al. 2006, National Research Council 2007). Soil and nutrient retention are encouraged by the continuous presence of belowground biomass with a more extended season of root activity, and in the case of long rotation systems the more protracted periods between harvests. The high nutrient retention capability of perennial biofuel crops could make them suitable for situations with high nutrient loading, such as riparian buffers or as disposal areas for nutrient-rich wastewater or sludge.

Water demands are likely to differ for cellulosic biofuel cropping systems compared to conventional grain crops because they tend to have longer growing seasons and deeper root systems, although this will be specific to the crops and regional climate. Water demand can be considered as overall water demand (e.g., evapotranspirative losses per hectare per year), or as water use efficiency, ideally considered as the ratio of harvestable biomass production to water lost to the atmosphere from the crop–soil system (often principally via transpiration). Estimates of water use efficiency exist in the literature but quantitative comparisons are difficult due to variable definitions of the concept and to the confounding effects of climatic variation. Furthermore, it is difficult to predict the water demands for future cellulosic biofuel crops in light of efforts to optimize crops for biomass production and water use efficiency by modern breeding and transgenic techniques.

In general, however, water use efficiencies of perennial biofuel crop systems, particularly woody and C_4 grass crops, are expected to exceed those of food crops (Jørgensen and Schelde 2001). C_4 crops can have twice the water use efficiency of C_3 crops in comparable settings and under optimal temperatures (Stanhill 1986). However, because of their higher productivity, the water demands for highly productive biofuel crops such as C_4 grasses can be higher on an areal basis despite their greater efficiency of water use on a biomass production basis.

Combined considerations of water demand, water use efficiency, and impacts on water quality are embodied in the concept of a water footprint. Gerbens-Leenes et al. (2009) presented a comprehensive analysis of the water footprint of a variety of bioenergy crops that are currently cultivated around the world, comparing their use for biofuel (ethanol or biodiesel derived from sugar, starch, or oil) and for bioelectricity (using all harvestable biomass), and that analysis showed how bioelectricity consumes (and contaminates) considerably less water per unit of energy generated. Domínguez-Faus et al. (2009) also evaluated water footprints, focusing on various current U.S. biofuel crops, plus switchgrass to represent a potential future cellulosic biofuel feedstock. These studies as well as compilations of estimates by de

Fraiture and Berndes (2009) and Porder et al. (2009) suggest that potential cellulosic biofuel crops may have smaller water footprints than grain-based biofuel crops, but the uncertainty in assumptions that needed to be made for cellulosic crops and the wide range in estimates among regions make this conclusion tentative and in need of further research.

The effects of potential cellulosic biofuel crops on landscape water balances are difficult to predict because most studies have examined conventional crop monocultures, and to a lesser extent relatively pristine forests or grasslands (National Research Council 2007). Cropping systems potentially affect landscape water balances through the evapotranspirative water demand of the plants, as well as via secondary impacts on soil water retention, surface runoff, and infiltration. Irrigation represents a major perturbation of landscape water balances; irrigation of cellulosic biofuel crops may seem improbable in the United States today but this could become economically viable with rising energy prices and the introduction of highly productive plant systems. Changes in landscape water balances are in turn coupled to landscape energy balances that could become important with large-scale biofuel crop production (Uhlenbrook 2007). Natural vegetation that existed on agricultural lands prior to conversion, or that would return by natural succession after cessation of agriculture, often has a distinct water balance with implications for groundwater recharge and stream runoff. Perennial cropping systems may mimic natural vegetation in this regard when native species mixtures are grown, or when crops are monocultures with growth forms resembling those of the native vegetation.

Important distinctions regarding water balances involve more deeply rooted woody vegetation vs. herbaceous plant cover, and the intensity of management and frequency of harvest of quasi-natural vegetation used for biofuel feedstocks. Conversion of forest to agricultural crops or pasture generally reduces precipitation interception, infiltration into soils, and groundwater recharge, and increases surface runoff and the potential for soil erosion (Uhlenbrook 2007). Establishing native forest on deforested land would be expected to have the converse effects. However, fast-growing forest plantations such as those envisaged for biofuel production tend to have higher water demand and are thus more likely to reduce stream flow and groundwater recharge compared to mature forests (Calder 1993, Farley et al. 2005), and poorly managed plantations can show high rates of soil erosion (Calder 1993, Kort et al. 1998). If such cropping systems are to be situated in riparian lands, their effects on water quality and quantity could be positive or negative and must be evaluated for a given setting.

The hydrology and nutrient balance of tropical landscapes where biofuel crop production is likely to accelerate in the future has been much less studied (Uhlenbrook 2007). Sugar cane as currently grown in tropical regions provides an instructive example (Martinelli and Filoso 2008, Simpson et al. 2009). Intensive sugar cane production whether for sugar or ethanol entails the use of fertilizers, pesticides, and fire. Soil erosion and water pollution are well documented problems in cane growing regions (Sparovek et al. 2001, Rayment 2003). In certain countries including India and China, large-scale irrigation of biofuel crops is practiced in regions where rainfall is inadequate, and in the case of Brazilian sugar cane, as a means of organic waste disposal from the refining process (Varghese 2007, de Fraiture et al. 2008, de Fraiture and Berndes 2009, Simpson et al. 2009).

Biorefineries also require water and their current consumptive water use exceeds that of petroleum refineries per volume of fuel produced, although their overall water use is modest compared to consumptive water use by the crops from which their feedstocks are derived (de Fraiture and Berndes 2009). Nonetheless, biorefineries can exert significant pressure on local water resources, competing with other uses, and often their locations are dictated more by the availability of feedstock than water (National Research Council 2007). Technological developments, including the possibility for new technologies such as thermochemical conversion, are expected to increase the efficiency of water use in these facilities.

The waste effluent produced by biorefineries will also require management of nutrients, salt, and in the case of cellulosic feedstocks, biological oxygen demand, all of which can be addressed with current technologies. An emergent threat to water quality may be posed by the expanding use of dry distillers' grains and solubles, byproducts of grain-based ethanol production, as an affordable source of livestock feed; this material tends to contain much more phosphorus than the animals require and therefore its widespread use may increase phosphorus loading to the environment (Simpson et al. 2009).

Modeling Biofuel Systems

Ecosystem models have been used extensively to evaluate the impacts of changes in agricultural land management and climate on ecosystem dynamics (Cramer et al. 2001, Pepper et al. 2005, Li 2007). Such models can simulate the effect of environmental change on plant production, nutrient cycling, soil trace gas fluxes, and soil water and temperature dynamics. They are particularly valuable for evaluating the effects of long-term change involving complex interactions, and thus they are an important tool for examining the outcomes of alternative biofuel cropping systems at local and regional scales.

Adler et al. (2007), for example, used the DAYCENT biogeochemical model (Del Grosso et al. 2006) to examine greenhouse gas fluxes and biomass yields for a variety of biofuel crops in the eastern United States. Model results were combined with fossil fuel costs to estimate that net greenhouse gas reductions for poplar

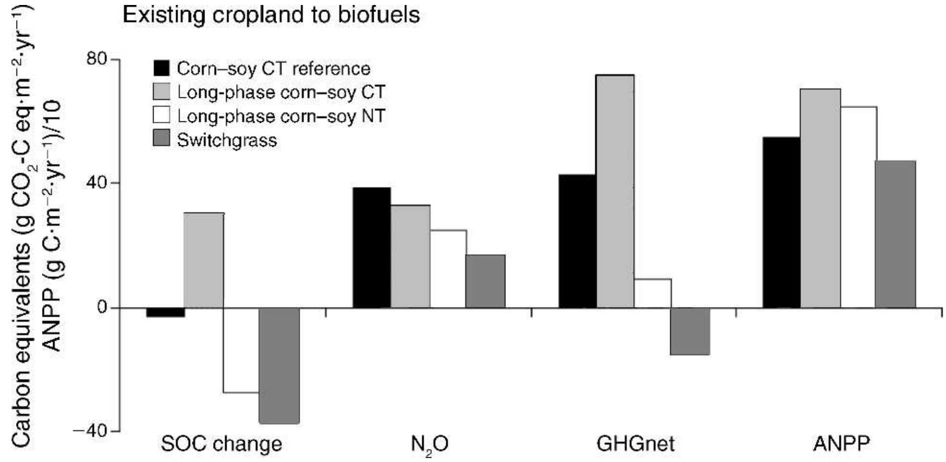

FIG. 2. Model predictions of soil carbon change, N_2O flux, net greenhouse gas (GHG) balance (GHGnet), and aboveground net primary production (ANPP) for existing corn–soybean conventionally tilled farmland in central Iowa converted to (left to right within each group of bars) long-phase corn–soybean (four years of corn followed by one year of soybean) conventionally tilled (CT), long-phase corn–soybean no till (NT), and switchgrass biofuel production systems. Net GHG change includes changes in soil organic carbon (SOC), direct and indirect N_2O emissions, and CO_2 emissions associated with production and application of N fertilizer. Negative values indicate net GHG mitigation.

and switchgrass exceeded those from corn by a factor of three. Consistent with experimental evidence from elsewhere, displaced fossil fuel use was the largest factor in the greenhouse gas reduction, and N_2O emissions were the largest single greenhouse gas source in this analysis.

To illustrate the value of models for projecting impacts in the absence of comprehensive empirical studies, we provide here the results of DAYCENT model simulations that compare the impacts of converting three types of agricultural land to biofuel production: (1) existing cropland, (2) Conservation Reserve Program (CRP) or set-aside fallow land, and (3) native grassland.

DAYCENT (Parton et al. 1998, Del Grosso et al. 2001, 2006) is a process-based model of intermediate complexity. It includes submodels for plant growth and senescence; microbial decomposition; water and nutrient flows through soil; soil temperature change; evaporation and transpiration of soil water; and nitrification, denitrification, and other nitrogen and carbon cycle processes. Soil C levels fluctuate according to inputs from senesced biomass, and emissions are a function of soil texture, inorganic N and labile C availability, water, temperature, and plant N demand. Plant growth is a function of soil nutrient and water availability, temperature, and plant specific parameters such as maximum growth rate, minimum and maximum biomass C:N ratio, and above vs. below ground C allocation.

We parameterized DAYCENT to simulate the conversion of three contrasting systems to biofuels: (1) conventionally tilled cropland with no residue removal; (2) 20 year-old CRP land in Webster County, Iowa; and (3) native prairie in Manhattan, Kansas. The conversion of each of these three systems to one of several biofuel systems was then modeled for 10 years. Modeled biofuel systems include (1) four years of corn fertilized at 150 kg N/ha followed by one year of soybean (not fertilized) with 70% of aboveground corn residue removed and different combinations of tillage (conventional vs. no-till); (2) switchgrass fertilized at 70 kg N/ha); (3) harvested prairie; and (4) harvested prairie fertilized at 70 kg N/ha.

To calculate net GHG emissions, we accounted for changes in soil organic carbon, direct and indirect N_2O emissions, and CO_2 emissions associated with production, transport, and application of N fertilizer. Indirect N_2O emissions were estimated by assuming that 1% of volatilized NO_x-N and NH_3-N are converted to N_2O-N offsite and that 0.75% of leached NO_3-N is converted to N_2O-N in aquatic systems as recommended by De Klein et al. (2006). Emissions associated with N fertilizer were calculated by assuming that 0.8 g of CO_2-C is released for every gram of N applied (Schlesinger 2000).

DAYCENT shows that conversion of a conventionally tilled (CT) corn–soybean system to a long-phase corn–soybean system (four years of corn and one year of soybean) leads to soil carbon loss under CT but storage under no till (NT), decreased N_2O emissions, particularly under NT, and overall increases in net greenhouse gas fluxes for CT but substantial decreases for NT cultivation (Fig. 2). Plant production increased about 27% under CT and 17% under NT for the long-phase corn–soybean rotation because of increased plant production by corn compared to soybean. Conversion of the CT corn–soybean system to switchgrass resulted in the highest soil carbon storage and substantially decreased N_2O emissions. N_2O emissions are lower for the NT conversion because harvesting residue removes some N from the system and soil organic matter storage

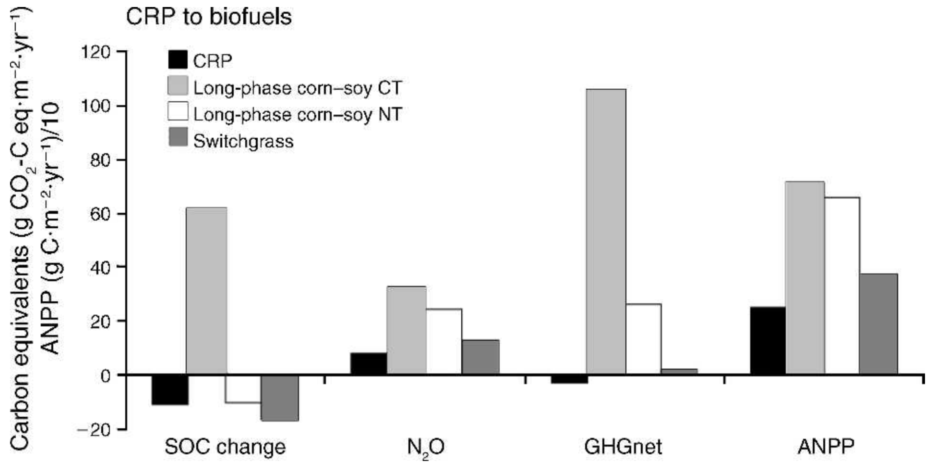

FIG. 3. Model predictions of soil carbon change, N_2O flux, net GHG balance (GHGnet), and aboveground net primary production for conservation reserve program (CRP) farmland in central Iowa converted to (left to right within each group of bars) long-phase corn–soybean (four years of corn followed by one year of soybean) conventionally tilled (CT), long-phase corn–soybean no till (NT), and switchgrass biofuel production systems. See Fig. 2 legend for further explanation.

results in less mineral N available for the processes that result in N_2O emissions. The switchgrass system had a net negative GHG flux (i.e., it consumed more GHGs than it emitted to the atmosphere), while the long-phase corn–soybean system was a small GHG source under NT but a substantial source under CT.

Model results suggest that conversion of CRP land into a CT long-phase corn–soybean rotation results in loss of soil carbon, greatly increased N_2O emissions, and a consequent large increase in the net GHG fluxes (Fig. 3). Plant production in the converted land increases by over 100% due to high corn yields. Converting CRP land into a NT long-phase corn–soybean system results in soil carbon storage similar to CRP, increased N_2O emissions, and a much smaller (but still positive) net GHG source than CT.

The modeled increase in soil carbon levels for the NT system compared to the CT system results from large increases in carbon inputs to the system without the increased decomposition of plant material associated with conventional tillage. Conversion of CRP land into switchgrass production resulted in increased soil carbon storage and a small increase in N_2O emission, leading to a slightly positive net greenhouse gas balance. ANPP for switchgrass is 50% higher than for the CRP land because of N-fertilizer additions.

Harvesting restored prairie for cellulosic biomass (Fig. 4) appears to result in net greenhouse gas production unless the prairie is fertilized. This is because harvested prairie (Tilman et al. 2006) is predicted to lose soil carbon: less aboveground residue is returned to the soil as compared to grazed prairie that is burned every four years. Plant productivity in harvested prairie slowly

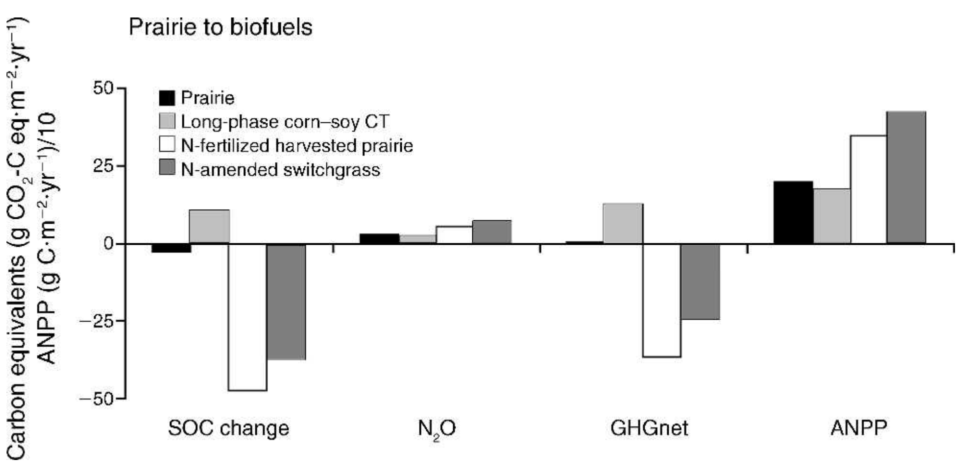

FIG. 4. Model predictions of soil carbon change, N_2O flux, net GHG balance (GHGnet), and aboveground net primary production for native prairie in eastern Kansas converted to (left to right within each group of bars) harvested prairie, fertilized harvested prairie, and fertilized switchgrass production systems. See Fig. 2 legend for further explanation.

PLATE 1. CO_2 eddy flux towers provide the resolution necessary to immediately assess the annual gains and losses of carbon from candidate biofuel crops, such as this tower in a newly established switchgrass (*Panicum virgatum*) field at the Kellogg Biological Station in southwest Michigan, USA. Photo credit: Tullio Zenone.

declines as more nitrogen is removed in the harvested biomass than would have been removed by grazing and burning. Fertilization improves the greenhouse gas balance because it stimulates carbon accumulation in roots as it stimulates by almost 75% aboveground plant growth and, although there is an N_2O cost to fertilization, it is minor because fertilizer requirements (70 kg $N \cdot ha^{-1} \cdot yr^{-1}$) are relatively low.

Converting the prairie system to annually fertilized (and harvested) switchgrass appears to result in increased soil C storage, a slight increase in N_2O emissions, and a negative net GHG flux. ANPP is higher for the switchgrass system compared to the fertilized prairie system, while soil carbon storage is lower due to fewer root inputs.

Overall model outputs suggests that converting CRP into CT long phase corn–soybean rotations will lead to substantial soil carbon losses and overall large increases in net GHG fluxes. In contrast, converting CRP into NT long phase corn–soybean rotations maintains soil carbon storage and leads to smaller increased net GHG fluxes mainly due to increased N_2O flux. The simulated increase in soil carbon with NT results from the increase in carbon inputs; although CT also had high carbon inputs, tillage led to its rapid oxidation. These results are consistent with field data from Iowa showing that conversion of CRP to CT cropland decreases soil C while conversion to NT cropping maintains soil C (Gilley et al. 1997, Follett et al. 2009), and with date from Nebraska showing higher soil C levels under NT compared to CT wheat cropping (Kessavalou et al. 1998).

Model results suggest that long phase corn–soybean rotations under NT will store carbon compared to typical corn–soy rotations under CT. This is largely because of higher C inputs (ANPP) plus the absence of tillage in these systems, and results are consistent with eddy covariance data for Nebraska (Grant et al. 2007) showing that net ecosystem productivity is negative during soybean years and positive during corn years.

Overall, the most negative greenhouse gas balances resulted from conversion of existing cropland to switchgrass, and conversion of prairie to either a fertilized and harvested prairie or fertilized switchgrass system.

An important caveat in this modeling exercise involves the 10-year time scale, which would represent the period of greatest soil C change after a change in cropping systems. In the cases of a change from CT to perennial crops or NT, soil C accumulation would begin to level off if the new crop management regime were maintained for ensuing decades, thereby substantially changing the overall net greenhouse gas fluxes.

Additionally, if the cropping system were to change again in the future in a way that fosters enhanced loss of soil C, the climate benefits of sequestered soil C would be proportionally attenuated. Lastly, the long phase corn–soy systems considered here assumed that 70% of residue would be harvested. Further analyses would be required to evaluate the benefits of biofuel production from these residues compared to the decrease in soil carbon for the CT long phase corn–soy system associated with residue harvest.

Conclusions

Biofuel production systems can confer biogeochemical benefits if managed with an appropriate mixture of crop choice, rotational complexity, and nitrogen and tillage management practices. Soil carbon change and nitrous oxide fluxes most affect the overall greenhouse gas balance of converted ecosystems, and other effects of conversion will include nitrate leaching and changes to the hydrologic cycle. Overall outcomes must be considered from a systems perspective. There is sufficient latitude in different management scenarios to provide a wide range of potential outcomes, ranging from sharp increases in biogeochemical liabilities to overall reductions. And effects will not necessarily be unidirectional, such that tradeoffs among alternative outcomes will need to be carefully assessed.

Acknowledgments

Support for this work was provided by the DOE Great Lakes Bioenergy Research Center (DOE BER Office of Science and DOE OBP EERE), the NSF Long-term Ecological Research Program, and the Michigan Agricultural Experiment Station.

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, R. B., R. A. Smith, G. E. Schwarz, E. W. Boyer, J. V. Nolan, and J. W. Brakebill. 2008. Differences in phosphorus and nitrogen delivery to the Gulf of Mexico from the Mississippi River Basin. Environmental Science and Technology 42:822–830.

Böhlke, J. K. 2002. Groundwater recharge and agricultural contamination. Hydrogeology Journal 10:153–179.

Calder, I. R. 1993. Hydrologic effects of land use change. Pages 13.1–13.50 *in* D. R. Maidment, editor. Handbook of hydrology. McGraw-Hill, New York, New York, USA. 13.1–13.50.

Costello, C., W. M. Griffin, A. E. Landis, and H. S. Matthews. 2009. Impact of biofuel crop production on the formation of hypoxia in the Gulf of Mexico. Environmental Science and Technology 43:7985–7991.

Cramer, W., et al. 2001. Global response of terrestrial ecosystem structure and function to CO_2 and climate change: results from six dynamic global vegetation models. Global Change Biology 7:357–373.

Crutzen, P. J., A. R. Mosier, K. A. Smith, and W. Winiwarter. 2008. N_2O release from agro-biofuel production negates global warming reduction by replacing fossil fuels. Atmospheric Chemistry and Physics 8:389–395.

de Fraiture, C., and G. Berndes. 2009. Biofuels and water. Pages 139–153 *in* R. W. Howarth and S. Bringezu, editors. Biofuels: environmental consequences and interactions with changing land use. Proceedings of the Scientific Committee on Problems of the Environment (SCOPE) International Biofuels Project Rapid Assessment, 22–25 September 2008, Gummersbach, Germany. Cornell University, Ithaca, New York, USA. ⟨http://cip.cornell.edu/biofuels/⟩

de Fraiture, C., M. Giordano, and Y. Liao. 2008. Biofuels and implications for agricultural water use: blue impacts of green energy. Water Policy 10:67–81.

De Klein, C., R. S. A. Nova, S. Ogle, K. A. Smith, P. Rochette, T. C. Wirth, B. G. McConkey, A. Mosier, and K. Rypdal. 2006. N_2O emissions from managed soils, and CO_2 emissions from lime and urea application. Pages 11.1–11.54 *in* H. S. Eggleston, L. Buendia, K. Miwa, T. Ngara, and K. Tanabe, editors. IPCC guidelines for national greenhouse gas inventories. Volume 4: agriculture, forestry and other land use. Institute for Global Environmental Strategies, Hanagawa, Japan.

Del Grosso, S. J., W. J. Parton, A. R. Mosier, M. D. Hartman, J. Brenner, D. S. Ojima, and D. S. Schimel. 2001. Simulated interaction of carbon dynamics and nitrogen trace gas fluxes using the DAYCENT model. Pages 303–332 *in* M. Schaffer et al., editors. Modeling carbon and nitrogen dynamics for soil management. CRC Press LLC, Boca Raton, Florida, USA.

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 N_2O emissions from cropped soils in the USA. Journal of Environmental Quality 35:1451–1460.

Domínguez-Faus, R., S. E. Powers, J. Burken, and P. J. Álvarez. 2009. The water footprint of biofuels: a drink or drive issue? Environmental Science and Technology 43:3005–3010.

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.

Economic Research Service. 2005. Land use, value, and management: agricultural land use change. Economic Research Service, U.S. Department of Agriculture, Washington, D.C., USA. ⟨http://www.ers.usda.gov/Briefing/LandUse/agchangechapter.htm#extensive⟩

Economic Research Service. 2008. Major uses of land in the US. Economic Research Service, U.S. Department of Agriculture, Washington, D.C., USA. ⟨http://www.ers.usda.gov/Data/MajorLandUses/⟩

European Environment Agency. 2006. How much bioenergy can Europe produce without harming the environment? Report No. 7. European Environment Agency, Copenhagen, Denmark.

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

Farley, K. A., E. G. Jobbagy, and R. B. Jackson. 2005. Effects of afforestation on water yield: a global synthesis with implications for policy. Global Change Biology 11:1565–1576.

Farrell, A. E., R. J. Plevin, B. T. Turner, A. D. Jones, M. O'Hare, and D. M. Kammen. 2006. Ethanol can contribute to energy and environmental goals. Science 311:506–508.

Field, C. B., J. E. Campbell, and D. B. Lobell. 2008. Biomass energy: the scale of the potential resource. Trends in Ecology and Evolution 23:65–72.

Fletcher, R. J., B. A. Robertson, J. Evans, D. J. Schemske, Alavalapati, and P. Doran. 2010. Biodiversity conservation in the era of biofuels: risks and opportunities. Frontiers in Ecology and the Environment. [doi: 10.1890/090091]

Follett, R. F., G. E. Varvel, J. M. Kimble, and K. P. Vogel. 2009. No-till corn after bromegrass: effect on soil carbon and soil aggregates. Agronomy Journal 101:261–268.

Galloway, J. N., A. R. Townsend, J. W. Erisman, M. Bekunda, Z. Cai, J. R. Freney, L. A. Martinelli, S. P. Seitzinger, and M. A. Sutton. 2008. Transformation of the nitrogen cycle:

Recent trends, questions, and potential solutions. Science 320:889–892.

Gerbens-Leenes, W., A. Y. Hoekstra, and T. H. van der Meer. 2009. The water footprint of bioenergy. Proceedings of the National Academy of Sciences USA 106:10219–10223.

Gibbs, H. K., M. Johnston, J. A. Foley, T. Holloway, C. Monfreda, N. Ramankutty, and D. Zak. 2008. Carbon payback times for tropical biofuel expansion: the effects of changing yield and technology. Environmental Research Letters 3:034001.

Gilley, J. E., J. W. Doran, D. L. Karlen, and T. C. Kaspar. 1997. Runoff, erosion, and soil quality characteristics of a former Conservation Reserve Program site. Journal of Soil and Water Conservation 52:189–193.

Graham, R. L., R. Nelson, J. Sheehan, R. D. Perlack, and L. L. Wright. 2007. Current and potential US corn stover supplies. Agronomy Journal 99:1–11.

Grandy, A. S., T. D. Loecke, S. Parr, and G. P. Robertson. 2006. Long-term trends in nitrous oxide emissions, soil nitrogen, and crop yields of till and no-till cropping systems. Journal of Environmental Quality 35:1487–1495.

Grant, R. F., T. J. Arkebauer, A. Dobermann, K. G. Hubbard, T. T. Schimelfenig, A. E. Suyker, S. B. Verma, and D. T. Walters. 2007. Net biome productivity of irrigated and rainfed maize–soybean rotations: modeling vs. measurements. Agronomy Journal 99:1404–1423.

Hamilton, S. K., A. L. Kurzman, C. Arango, L. Jin, and G. P. Robertson. 2007. Evidence for carbon sequestration by agricultural liming. Global Biogeochemical Cycles 21: GB2021.

Hammerschlag, R. 2006. Ethanol's energy return on investment: a survey of the literature 1990–present. Environmental Science and Technology 40:1744–1750.

Howarth, R. W., S. Bringezu, L. A. Martinelli, R. Santoro, D. Messem, and O. E. Sala. 2009. Introduction: biofuels and the environment in the 21st century. Pages 15–36 *in* R. W. Howarth and S. Bringezu, editors. Biofuels: environmental consequences and interactions with changing land use. Proceedings of the Scientific Committee on Problems of the Environment (SCOPE) International Biofuels Project Rapid Assessment, 22–25 September 2008, Gummersbach Germany. Cornell University, Ithaca, New York, USA. ⟨http://cip.cornell.edu/biofuels/⟩

Intergovernmental Panel on Climate Change. 2007. Fourth assessment report. Working group I report: the physical science basis. Cambridge University Press, Cambridge, UK.

Jørgensen, U., and K. Schelde. 2001. Energy crop water and nutrient use efficiency. The International Energy Agency, IEA Bioenergy Task 17, Short Rotation Crops, Danish Institute of Agricultural Sciences, Department of Crop Physiology and Soil Science, Research Centre Foulum. Tjele, Denmark.

Kessavalou, A., A. R. Mosier, J. W. Doran, R. A. Drijber, D. L. Lyon, and O. Heinemeyer. 1998. Fluxes of carbon dioxide, nitrous oxide, and methane in grass sod and winter wheat-fallow tillage management. Journal of Environmental Quality 27:1094–1104.

Kim, S., and B. E. Dale. 2005. Life cycle assessment of various cropping systems utilized for producing biofuels: bioethanol and biodiesel. Biomass and Bioenergy 29:426–439.

Kort, J., M. Collins, and D. Ditsch. 1998. A review of soil erosion potential associated with biomass crops. Biomass and Bioenergy 14:351–359.

Lal, R. 1998. Soil erosion impact on agronomic productivity and environment quality. Critical Reviews in Plant Science 17:319–464.

Lazear, E. P. 2008. Testimony before the Senate Foreign Relations Committee Hearing on Responding to the Global Food Crisis, May 14, 2008. Federal Register. ⟨http://georgewbush-whitehouse.archives.gov/cea/lazear20080514.html⟩

Li, C. 2007. Quantifying greenhouse gas emissions from soils: scientific basis and modeling approach. Soil Science and Plant Nutrition 53:344–352.

Martinelli, L. A., and S. Filoso. 2008. Expansion of sugarcane ethanol production in Brazil: environmental and social challenges. Ecological Applications 18:885–898.

Melillo, J. M., J. M. Reilly, D. W. Kicklighter, A. C. Gurge, T. W. Cronin, S. Paltsev, B. S. Felzer, X. Wang, A. P. Sokolov, and C. A. Schlosser. 2009. Indirect emissions from biofuels: how important? Science 326:1397–1399.

Mosier, A. R., A. D. Halvorson, G. A. Peterson, G. P. Robertson, and L. Sherrod. 2005. Measurement of net global warming potential in three agroecosystems. Nutrient Cycling in Agroecosystems 7:67–76.

National Research Council. 2007. Water implications of biofuels production in the United States. National Academies Press, Washington, D.C., USA.

National Research Council. 2009. Liquid transportation fuels from coal and biomass. National Academies Press, Washington, D.C., USA.

Natural Resources Defense Council. 2007. Move over, gasoline: here comes biofuels. Natural Resources Defense Council, Washington, D.C., USA. ⟨http://www.nrdc.org/air/transportation/biofuels.asp⟩

Nolan, B. T., B. C. Ruddy, K. J. Hitt, and D. R. Helsel. 1997. Risk of nitrate in groundwater of the United States: a national perspective. Environmental Science and Technology 31:2229–2236.

Ohlrogge, J., D. Allen, B. Berguson, D. DellaPenna, Y. Shachar-Hill, and S. Stymne. 2009. Driving on biomass. Science 324:1019–1020.

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.

Pepper, D. A., S. J. Del Grosso, R. E. McMurtrie, and W. J. Parton. 2005. Simulated carbon sink response of shortgrass steppe, tallgrass prairie and forest ecosystems to rising [CO_2], temperature and nitrogen input. Global Biogeochemical Cycles 19:GB1004.

Perlack, R. D., L. L. Wright, A. F. Turhollow, R. L. Graham, B. J. Stokes, and D. C. Erbach. 2005. Biomass as feedstock for a bioenergy and bioproducts industry: the technical feasibility of a billion-ton annual supply. Technical report DOE/GO-102005-2135. U.S. Department of Energy, Washington, D.C., USA.

Pimentel, D., and T. Patzek. 2005. Ethanol production using corn, switchgrass, and wood; biodiesel production using soybean and sunflower. Natural Resources Research 14:65–76.

Porder, S., A. Bento, A. Leip, L. A. Martinelli, J. Samseth, and T. Simpson. 2009. Toward an integrated assessment of biofuel technologies. Pages 233–248 *in* R. W. Howarth and S. Bringezu, editors. Biofuels: environmental consequences and interactions with changing land use. Proceedings of the Scientific Committee on Problems of the Environment (SCOPE) International Biofuels Project Rapid Assessment, 22–25 September 2008, Gummersbach Germany. Cornell University, Ithaca, New York, USA. ⟨http://cip.cornell.edu/biofuels/⟩

Rayment, G. E. 2003. Water quality in sugar catchments of Queensland. Water Science and Technology 48:35–47.

Renewable and Appropriate Energy Laboratory. 2007. Energy and resources group biofuel analysis meta-model (EBAMM). University of California, Berkeley, California, USA. ⟨http://rael.berkeley.edu/EBAMM⟩

Richter, D., L. R. McCreery, K. P. Nemestothy, D. H. Jenkins, J. T. Karakash, and J. Knight. 2009. Wood energy in America. Science 323:1432–1433.

Robertson, G. P. 1997. Nitrogen use efficiency in row-crop agriculture: crop nitrogen use and soil nitrogen loss. Pages

347–365 *in* L. Jackson, editor. Ecology in agriculture. Academic Press, New York City, New York, USA.

Robertson, G. P. 2004. Abatement of nitrous oxide, methane, and the other non-CO_2 greenhouse gases: the need for a systems approach. Pages 493–506 *in* C. B. Field and M. R. Raupach, editors. The global carbon cycle. Island Press, Washington, D.C., USA.

Robertson, G. P., et al. 2008. Sustainable biofuels redux. Science 322:49.

Robertson, G. P., E. A. Paul, and R. R. Harwood. 2000. Greenhouse gases in intensive agriculture: contributions of individual gases to the radiative forcing of the atmosphere. Science 289:1922–1925.

Robertson, G. P., and P. M. Vitousek. 2009. Nitrogen in agriculture: balancing an essential resource. Annual Review of Environment and Resources 34:97–125.

Samson, R., C. H. Lem, S. B. Stamler, and J. Dooper. 2008. Developing energy crops for thermal applications: optimizing fuel quality, energy security, and GHG mitigation. Pages 395–423 *in* D. Pimentel, editor. Biofuels, solar and wind as renewable energy systems. Springer, New York, New York, USA.

Schlesinger, W. H. 2000. Carbon sequestration in soils: some cautions amidst optimism. Agriculture Ecosystems and Environment 82:121–127.

Schmer, M. R., K. P. Vogel, R. B. Mitchell, and R. K. Perrin. 2008. Net energy of cellulosic ethanol from switchgrass. Proceedings of the National Academy of Sciences USA 105:464–469.

Searchinger, T., R. Heimlich, R. A. Houghton, F. Dong, A. Elobeid, J. Fabiosa, S. Tokgoz, D. Hayes, and T.-H. Yu. 2008. Use of U.S. croplands for biofuels increases greenhouse gases through emissions from land-use change. Science 319:1238–1240.

Searchinger, T. D., et al. 2009. Fixing a critical climate accounting error. Science 326:527–528.

Secchi, S., J. Tyndall, L. A. Schulte, and H. Asbjornsen. 2008. High crop prices and conservation: raising the stakes. Journal of Soil and Water Conservation 63:68A–73A.

Simpson, T. W., L. A. Martinelli, A. N. Sharpley, and R. W. Howarth. 2009. Impact of ethanol production on nutrient cycles and water quality: the United States and Brazil as case studies. Pages 153–167 *in* R. W. Howarth and S. Bringezu, editors. Biofuels: environmental consequences and interactions with changing land use. Proceedings of the Scientific Committee on Problems of the Environment (SCOPE) International Biofuels Project Rapid Assessment, 22–25 September 2008, Gummersbach, Germany. Cornell University, Ithaca, New York, USA. ⟨http://cip.cornell.edu/biofuels/⟩

Sparovek, G., M. A. Anisimova, M. Kolb, M. Bahadir, H. Wehage, and E. Schnug. 2001. Organochlorine compounds in a Brazilian watershed with sugarcane and intense sediment redistribution. Journal of Environmental Quality 30:2006–2010.

Stanhill, G. 1986. Water use efficiency. Advances in Agronomy 39:53–85.

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

Uhlenbrook, S. 2007. Biofuel and water cycle dynamics: what are the related challenges for hydrological processes research? Hydrological Processes 21:3647–3650.

U.S. Department of Agriculture. 2010. US corn production. National Agricultural Statistics Service, U.S. Department of Agriculture, Washington, D.C., USA. ⟨www.nass.usda.gov/Charts_and_Maps/Field_Crops/cornprod.asp⟩

U.S. Department of Energy. 2010. Annual energy review 2009. Report No. DOE/EIA-0384(2009). U.S. Department of Energy, Washington, D.C., USA.

U.S. Environmental Protection Agency. 2007. A wedge analysis of the US transportation sector. EPA Office of Transportation and Air Quality. Technical Report EPA420-R-07-007. U.S. Environmental Protection Agency, Washington, D.C., USA.

Varghese, S. 2007. Biofuels and global water challenges. Institute for Agriculture and Trade Policy, Minneapolis, Minnesota, USA.

West, T. O., and G. Marland. 2003. Net carbon flux from agriculture: carbon emissions, carbon sequestration, crop yield, and land-use change. Biogeochemistry 63:73–83.

West, T. O., and A. C. McBride. 2005. The contribution of agricultural lime to carbon dioxide emissions in the United States: dissolution, transport, and net emissions. Agricultural Ecosystems and Environment 108:145–154.

Wilhelm, W. W., J. M. F. Johnson, D. L. Karlen, and D. T. Lightle. 2007. Corn stover to sustain soil organic carbon further constrains biomass supply. Agronomy Journal 99:1665–1667.

Wise, M., K. Calvin, A. M. Thomson, L. Clarke, B. Bond-Lamberty, R. D. Sands, S. J. Smith, A. Janetos, and J. Edmonds. 2009. Implications of limiting CO_2 concentrations for land use and energy. Science 324:1183–1186.