

# The Ecological Impact of Biofuels

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biodiesel, biodiversity, ethanol, life cycle, water, indirect land-use change

## Abstract

The ecological impact of biofuels is mediated through their effects on land, air, and water. In 2008, about 33.3 million ha were used to produce food-based biofuels and their coproducts. Biofuel production from food crops is expected to increase 170% by 2020. Economic model estimates for land-use change (LUC) associated with food-based biofuels are 67–365 ha 10<sup>−6</sup> 1<sup>−1</sup>, leading to increased greenhouse gas emissions for decades compared to business as usual. Biodiversity is reduced by about 60% in U.S. corn and soybean fields and by about 85% in Southeast Asian oil palm plantations compared to unconverted habitat. Consequently, the largest ecological impact of biofuel production may well come from market-mediated LUC. Mitigating this impact requires targeting biofuel production to degraded and abandoned cropland and rangeland; increasing crop yields and livestock production efficiency; use of wastes, residues, and wildlife-friendly crops; and compensatory offsite mitigation for residual direct and indirect impacts.

## 1. INTRODUCTION

Agricultural production, whether for food, fiber, or fuel, affects ecosystems and the species within them, including humans. Many impacts of biofuels result from associated land-use change (LUC). Land for new biofuel production may come directly from clearing new land for agriculture, but often is the result of switching from one crop to another or diverting food crops to biofuel production. In these cases, the largest effects of biofuels on LUC occur indirectly, primarily through changes in food commodity prices (Hertel et al. 2010, Searchinger et al. 2008). The ecological impacts of biofuels are therefore inextricably linked with the economics of global agricultural markets. Understanding these complex effects requires the use of tools and methods novel to most ecologists. Therefore, we start with an overview of the tools and terminology of life-cycle assessment (LCA) as it has been applied to biofuels. We then review the ecological impacts of biofuels as mediated through their effects on land use, air pollution, and water use and quality. There has been less research linking the impacts of biofuels on land, air, and water with ecological responses at the species and community levels, yet we provide an overview of recent advances. We conclude with a brief discussion of ways to mitigate the impact of biofuels on biodiversity.

## 2. LIFE-CYCLE ASSESSMENT OF BIOFUELS

Life-cycle assessment is a tool to help assess the total resource use and environmental effects associated with products throughout their entire life cycle, from raw materials extraction through production, transportation, use, and disposal (ISO 2006a). There are two frameworks for performing LCA that answer different questions. The first, more traditional form focuses on the product supply chain, uses average production data and treats production as occurring in a static system. This framework, called attributional LCA, operates under the implicit assumption that the use of the product and its required inputs has no effect on markets (Weidema & Ekvall 2009). The alternative approach, known as consequential LCA, is a change-oriented analysis that examines the environmental consequences of a product's use as mediated by changes in the marketplace. For example, biofuel production that uses or displaces food commodities decreases the supply of these commodities available to food markets, thereby increasing the price of these commodities and causing more land to be brought into production. Many first-generation biofuel production systems generate coproducts that partially mitigate this reduction of supply in food and feed markets and should be considered in both attributional and consequential LCAs. Simply put, whereas an attributional life-cycle inventory (LCI) traces material and energy flows, a consequential LCI follows causal chains (Ekvall & Weidema 2004).

Reducing greenhouse gas (GHG) emissions is one stated goal of biofuel policy and use. Greenhouse gas emissions are mixed in the atmosphere globally, such that reducing environmental harm from GHG emissions can occur only if an action results in a net global decrease in GHG. Thus, the appropriate boundary for analyses of GHG emissions is global, including market-mediated (indirect) effects, which require consequential LCA (Feng et al. 2010, Hertel et al. 2010).

Another important distinction to make is between the average effects of a product's use and the effects of a marginal increase in a product's use. Both average and marginal effects can be calculated using LCA. To understand the environmental impacts of increasing biofuels production, we must consider marginal effects. Estimating marginal effects is more complicated than estimating average effects because one needs to identify the likely source of the next hectare of cropland or gallon of oil. In some cases, marginal and average effects may not be significantly different. For example, the U.S. Environmental Protection Agency (USEPA) found that the marginal gallon of oil displaced by biofuels is essentially the same as the global average of petroleum (USEPA 2010, p. 511). In other cases, marginal effects may be noticeably different from average effects. For example,

marginal crop yields would be lower than average crop yields if less fertile cropland is brought into production.

When system boundaries expand to include complex systems and human behavior, a high degree of uncertainty is unavoidable. Recent policies require regulation based on the quantification of the carbon emissions from indirect land-use change (ILUC) (CARB 2009, USEPA 2010). Indirect land-use changes are those induced through land and commodity markets when expanding biofuels production displaces agricultural commodities with relatively inelastic demand, such as food and feed (Babcock 2009). Some biofuels researchers have suggested that regulation of ILUC should be postponed until estimates can be made with certainty (Simmons et al. 2008), but this is based on a misperception that certainty is possible when predicting global economic behavior and associated environmental effects. Policy decisions must frequently be made in the absence of certainty. Biofuels policy is no exception and must take into account the uncertain risks associated with mandating or subsidizing different kinds of biofuels. Fortunately, production practices that use qualitatively less land and water can be identified even in the face of high uncertainty about the magnitude of indirect effects.

### 3. BIOFUELS AND LAND USE

Land requirements for biofuels can be characterized by this simple formula:

$$\text{land demand} = \frac{\text{biofuel quantity}}{\text{conversion efficiency} \times \text{crop yield} \times \text{unharvested correction}}.$$

However, some biofuels produce coproducts, which can replace other products in the marketplace, reducing the net quantity of food or feed displaced. Consequently, the amount of additional or net land required to produce these biofuels is less than the total amount of land on which the biofuel crop is produced. To calculate the amount of net land required for production, we modify the formula as follows:

$$\text{net land demand} = \left[ \frac{\text{biofuel quantity}}{\text{conversion efficiency} \times \text{crop yield} \times \text{unharvested correction}} \right] \times \text{coproduct discount}.$$

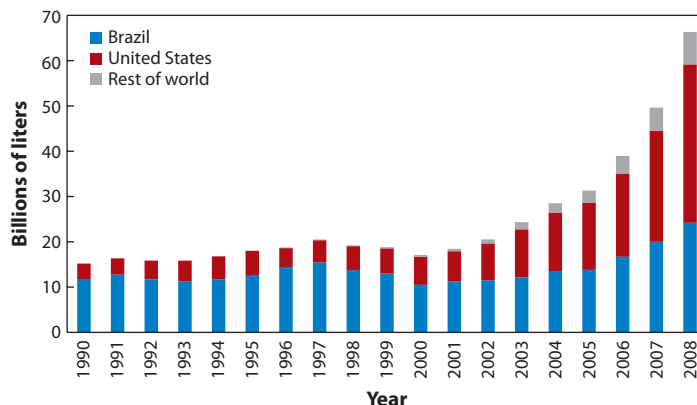
This calculation accurately represents the amount of additional land required for biofuel production, but it does not include indirect market effects that may influence other types of agricultural land use. These indirect effects could occur if biofuels increase food commodity prices, decreasing the ability of people to buy food and resulting in less land used for food production. Such changes are not socially desirable as they would increase hunger. Avoiding these effects should be a policy goal, and reductions in ILUC generated by food deficits should not be incentivized.

We address each of the factors affecting biofuel land use in turn below. Because each of these factors may change over time, we discuss both current values and likely future values.

#### 3.1. Biofuel Production

Recent trends in ethanol and biodiesel production by geography are presented in **Figures 1** and **2**. Global biofuel production in 2008 included 66 billion liters of ethanol (a 34% increase from 2007) and 15 billion liters of biodiesel (a 48% increase from 2007). This amounts to 1.67% of global liquid fuel production in 2008 on an energy basis (Worldwatch Institute 2009).

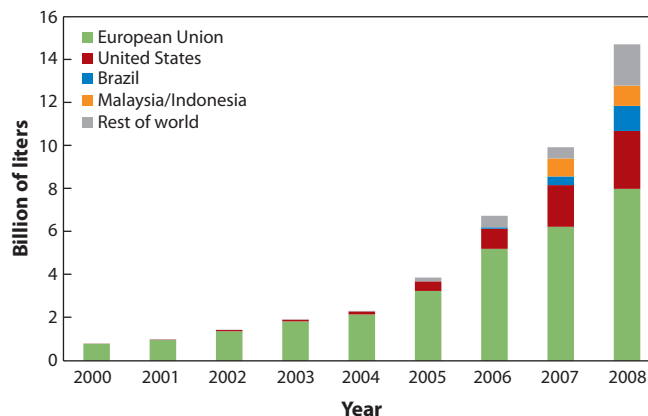
Biofuel production must increase to meet current mandates (**Table 1**). However, if the price of petroleum is sufficiently high, biofuel production will exceed mandated levels. For example,



**Figure 1**

Ethanol production by geographic area, 1990–2008. Data from Worldwatch Institute 2009.

during 2007/2008, U.S. ethanol production expanded faster than was mandated, driven by high petroleum prices, in addition to tax credits, tariffs on foreign ethanol, and mandates for replacing MTBE as an octane enhancer. The current U.S. renewable fuel standard (RFS2) mandates 136 billion liters (36 billion gallons) of biofuel by 2022. If met entirely with ethanol, this would quadruple U.S. ethanol production and cause a global increase of 150% in ethanol production over 2008 levels, even if production did not increase outside the United States. The RFS2 includes a mandate, starting in 2010, for cellulosic ethanol (ethanol made from plant biomass through either fermentation or thermochemical processes). The International Energy Agency (IEA) (2009) predicts that, compared to 2008, biofuel consumption will increase by 170–220% in 2020 and by 250–620% in 2030. The higher end of these ranges is predicted if a global atmospheric CO<sub>2</sub> target of 450 ppm is set. The IEA projects that second-generation biofuels (such as cellulosic ethanol) will not be commercially viable enough to be widely used until after 2020 (except to meet U.S. mandates), although the majority of increased production after 2020 is predicted to come from second-generation biofuels (International Energy Agency 2009). Consequently, we focus on the five major first-generation food crops used for biofuel (corn, sugarcane, soybean, oil



**Figure 2**

Biodiesel production by geographic area, 2000–2008. Data from Worldwatch Institute 2009.

**Table 1** Voluntary and mandatory biofuel targets for transport fuels in G8+5 countries<sup>a</sup>

Country/country grouping	Targets <sup>b</sup>
Brazil	Mandatory blend of 20–25% anhydrous ethanol with petrol; minimum blending of 3% biodiesel to diesel by July 2008 and 5% (B5) by end of 2010
Canada	5% renewable content in petrol by 2010 and 2% renewable content in diesel fuel by 2012
China	15% of transport energy needs through use of biofuels by 2020
France	5.75% by 2008, 7% by 2010, 10% by 2015 (V), 10% by 2020 (M = EU target)
Germany	6.75% by 2010, set to rise to 8% by 2015, 10% by 2020 (M = EU target)
India	Proposed blending mandates of 5–10% for ethanol and 20% for bio
Italy	5.75% by 2010 (M), 10% by 2020 (M = EU target)
Japan	500,000 kilolitres, as converted to crude oil, by 2010 (V)
Mexico	Targets under consideration
Russian Federation	No targets
South Africa	Up to 8% by 2006 (V) (10% target under consideration)
United Kingdom	5% biofuels by 2010 (M), 10% by 2020 (M = EU target)
United States	9 billion gallons by 2008, rising to 36 billion by 2022 (M). Of the 36 billion gallons, 21 billion are to be from advanced biofuels (of which 16 billion are to come from cellulosic biofuels)
European Union (EU)	10% by 2020 (M proposed by EU Commission in January 2008)

<sup>a</sup>Data taken from FAO 2008.

<sup>b</sup>Abbreviations: M, mandatory; V, voluntary.

palm, and rapeseed) and several cellulosic ethanol crops being considered in the U.S. (switchgrass, *Miscanthus*, corn stover, willow, and hybrid poplar).

### 3.2. Conversion Efficiency

Conversion efficiency varies among feedstocks. Conversion technology for oils, for the starch in grains, and for the sugar in sugarcane is relatively well established and approaches theoretical maximum yields, such that only small improvements can be expected unless additional parts of the grain (e.g., kernel fiber in addition to the endosperm) or the cane stalk (that is, the bagasse) are converted to ethanol. In contrast, conversion of biomass to cellulosic ethanol is still a relatively immature technology, so there is more uncertainty and room for technological improvement. In contrast with today's corn ethanol and biodiesel, sugarcane ethanol and cellulosic ethanol provide process heat and electricity, coproducts that displace fossil fuel use. **Tables 2** and **3** present efficiencies for ethanol and biodiesel, respectively.

**Table 2** Ethanol conversion efficiencies for different feedstocks

Feedstock	Location	Current average (l Mg <sup>-1</sup> )	Future potential (l Mg <sup>-1</sup> )	Reference
Corn	United States	416 (2.8 gal bu <sup>-1</sup> )	424 (2.85 gal bu <sup>-1</sup> )	USEPA 2010
Sugarcane	Brazil	86	92	Macedo et al. 2008
Biomass	— <sup>a</sup>	282	399	Lynd et al. 2008

<sup>a</sup>No commercial production; estimates based on laboratory research.

**Table 3 Biodiesel conversion efficiencies for different feedstocks**

Feedstock	Location	Biodiesel per crop <sup>a</sup> (l Mg <sup>-1</sup> )	Extractable oil (kg kg <sup>-1</sup> )	Reference
Soybean	United States	205	0.18	Hill et al. 2006
Palm	Indonesia/Malaysia	223–245	0.20–0.22	Reijnders & Huijbregts 2008
Rapeseed	Europe	434–479	0.39–0.43	FEDIOL 2010

<sup>a</sup>Assumes the density of biodiesel is 0.88 kg l<sup>-1</sup> and transesterification yields 98% biodiesel. Actual density of biodiesel can vary from 0.86–0.89, and transesterification yields can vary from 95–99%.

### 3.3. Crop Yields

Yields vary greatly between crops and between countries. Fortunately, there are relatively good country-specific data for most food crops (**Table 4**). However, there is more uncertainty about biomass crop yields, projecting future yields, yield decreases on marginal lands, and price-induced increases in yields. We address each of these in turn.

There is a lack of data on biomass crop yields necessary to predict land demand for biomass crops. In part, this is because the average yields obtained by farmers are often much lower than the yields achieved in research and demonstration plots. Based on experience with food crop yields, on-farm yields can be expected to be 20% to 70% lower than demonstration plot yields (Lobell et al. 2009) (see **Table 5**). Harvest losses may be higher for biomass crops than for food crops, exacerbating this effect. For example, switchgrass on-farm harvest yields were 35–45% lower than hand-harvested yields due to inefficiencies in the harvesting process (Monti et al. 2009). Additional on-farm trials are needed to improve predictions of potential biomass crop yields.

Because biomass crops have experienced relatively little breeding to date, there is certainly unexploited genetic variation in yield potential. However, the potential for future yield increases in biomass crops is too uncertain to quantify at this point.

Corn stover is the residue left over after grain harvest. Stover is potentially an abundant and cheap source of biomass for cellulosic ethanol. Leaving enough stover in fields to protect against wind and water erosion would still allow an estimated 54 million Mg to be harvested annually in

**Table 4 Current yields and growth trends for current biofuel crops**

Feedstock	Location	Current yield (3 years avg Mg ha <sup>-1</sup> )	Trend <sup>a</sup> (Mg ha <sup>-1</sup> year <sup>-1</sup> )
Corn <sup>b</sup>	United States	9.8	0.15
Sugarcane <sup>c</sup>	Brazil	87.1	0.87
Soy <sup>b</sup>	United States	2.8	0.02
Palm FFB <sup>d</sup>	Indonesia	17.9	0.09
Palm FFB <sup>d</sup>	Malaysia	21.3	0.20
Palm FFB <sup>d</sup>	Rest of World	6.5	0.08
Rapeseed <sup>d</sup>	Europe	2.7	0.02

Abbreviation: FFB, fresh fruit bunches.

<sup>a</sup>Based on yields since 1990.

<sup>b</sup>Data from ERS-USDA 2010.

<sup>c</sup>Data from Macedo et al. 2008 and FAO 2010.

<sup>d</sup>Data from FAO 2010.

**Table 5** Estimated yields for potential biofuel crops

Feedstock	Location	Plot size	Number of sites	Mean (se) (Mg ha <sup>-1</sup> )	range (Mg ha <sup>-1</sup> )	Expected average on-farm yields <sup>a</sup>
<i>Miscanthus</i> <sup>b</sup>	Europe	—	97	22.4 (4.1)	—	7–18
<i>Miscanthus</i> <sup>c</sup>	Illinois, U.S.A.	10 × 10 m	3	29.6 (1.8)	20.9–34.6	10–28
Switchgrass <sup>b</sup>	Illinois, U.S.A.	10 × 10 m	3	10.4 (1.0)	6.7–13.0	4–10
Switchgrass <sup>c</sup>	Europe	—	77	10.3 (0.7)	—	3–8
Switchgrass <sup>d</sup>	Northern Great Plains, U.S.A.	3–9 ha	10	7.1 (0.5)	5.2–11.1	5–11
Willow <sup>e</sup>	United Kingdom	9 × 11.5 m	49	9.0 (1.6)	4.6–11.3	3–9
Hybrid poplar <sup>e</sup>	United Kingdom	9 × 11.5 m	49	6.3 (1.7)	2.5–9.3	2–5
Corn stover <sup>f</sup>	United States	modeled	modeled	4.9	3.1–5.2	3–5

<sup>a</sup>Assumes a yield gap of 20–70% compared to potential yields (Lobell et al. 2009).

<sup>b</sup>Data from Heaton et al. 2004.

<sup>c</sup>Data from Heaton et al. 2008.

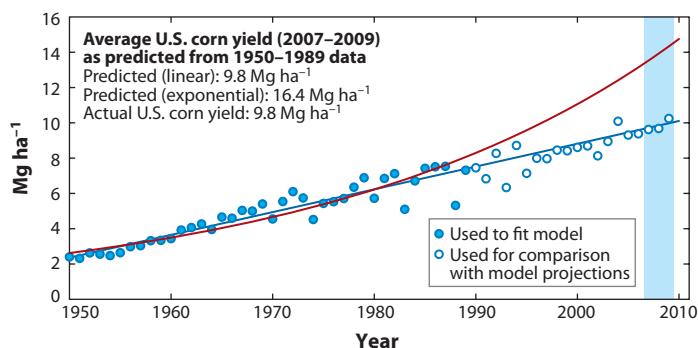
<sup>d</sup>Data from Schmer et al. 2008. As this study examined commercial-scale switchgrass production, no yield discount was applied.

<sup>e</sup>Yields are reported as average annual yields, from coppice harvests every three years (Aylott et al. 2008).

<sup>f</sup>Economically recoverable yields modeled based on corn grain yields (Graham et al. 1997).

the United States (Graham et al. 2007). However, this estimate is likely to be much lower once soil fertility and organic carbon depletion are also considered (Wilhelm et al. 2007).

Food crop yields have generally increased over time owing to ongoing technological improvement. Future crop yields are often projected by extrapolating from historical trends. Although yield increases are commonly reported as percent increases over time, this language is misleading because it suggests that historical yield growth has been exponential. However, this is not generally the case, so exponential extrapolations result in poor predictions, as illustrated in **Figure 3**.



**Figure 3**

U.S. corn yields have increased linearly over time, not exponentially. A linear (dark blue) trendline and an exponential trendline (red) were fit to U.S. corn yields from 1950–1989 and projected through 2009. Circles indicate observed annual average U.S. corn yields. Filled circles indicate data points used to fit models (1950–1989). Open circles indicate data points shown for comparison with fitted model projections, but not used to fit models (1990–2009). To evaluate the accuracy of the model projections, average yields for 2007–2009 (shaded area) were calculated based on linear model predictions (9.8 Mg ha<sup>-1</sup>) and exponential model predictions (16.4 Mg ha<sup>-1</sup>) and compared to average observed yields (9.8 Mg ha<sup>-1</sup>).



By fitting trends to data from 1950–1989, we predicted average U.S. corn yields in 2007–2009. A linear extrapolation was accurate to within  $0.01 \text{ Mg ha}^{-1}$ , but the exponential extrapolation overestimated yields by 67%. Projections of future yields that deviate significantly from historic linear trends should be treated with skepticism.

Other factors affect marginal crop yields. For example, the new area under production may not have the same yields as land currently under production. Specifically, new cropland may have lower yields than the average existing cropland. This is almost certainly the case in the United States, where the most fertile cropland is in use, and as recently as 2007, 14.9 million ha of less fertile land was enrolled in land retirement programs (that is, the Conservation Reserve Program, CRP). Much of the expansion of cropland in the United States is expected to come from bringing a portion of these lands back into production. Unfortunately, there are few estimates on how much lower the yields are on these marginal lands compared to the average yields on existing farmland. One study in Iowa predicted which lands would be brought into crop production as corn prices increased (Secchi et al. 2009). These data indicate that corn prices of \$4/bushel would result in the cropping of CRP land with average yields 9% lower than the average 2006 Iowa corn yields. Additional research is needed to determine how applicable this estimate is beyond Iowa and to price shocks expected with future biofuel production.

In developing countries, agricultural expansion is occurring on newly cleared land, rather than on marginal land brought back into production. In these situations, new cropland may have been previously unfarmed due to factors other than low soil fertility, such as lack of markets, infrastructure, or human capital. Babcock & Carriquiry (2010) found that regions in Brazil experiencing faster expansion of soybeans did not have lower soybean yields or yield growth. Thus, although additional research is needed in this area, the available data do not support lower yields on newly converted lands on the agricultural frontier.

If corn prices increase more rapidly than soybean prices, farmers may switch from corn-soybean rotations to continuous corn, resulting in lower corn yields. Compared to corn planted after soybean, corn planted after corn has yields that are 5–17% lower, depending on the tillage system (Vyn et al. 2000).

Finally, yields could increase in response to biofuel production if biofuel demand increases crop prices and farmers invest more in inputs (e.g., high-yielding crop varieties or fertilizer), or, in the longer term, if higher prices lead to increased technological improvements. This type of induced innovation has long been thought to play a role in agricultural development (Hayami & Ruttan 1971). The importance of such yield responses for estimating the magnitude of new land demand in response to increased biofuel production has recently been reviewed by Keeney & Hertel (2009). They estimate that about 30% of the increase in corn production for ethanol will come from yield increases rather than from cropland expansion. Because yields vary with weather from year to year, it is too soon to determine whether this hypothesized yield response to recent corn price increases in the United States is supported by empirical data. Further, Keeney & Hertel's estimate is based on average of values from four studies, only one of which uses data extending past 1973. Given the importance of yield increases for predicting land demand associated with increased biofuels, econometric analyses using more recent data are warranted.

### 3.4. Unharvested Correction

Not all land that is planted to a crop is harvested every year. In annual crops, this is largely due to crop failure. In perennial crops, plantations are often unharvested during establishment and must be periodically replanted. Because yields are calculated only on lands that are actually harvested, ignoring this factor underestimates actual land required to produce a given quantity of biofuel.



**Table 6** Percentage of feedstock harvested each year

Feedstock	Location	Percentage	Reference
Corn <sup>a</sup>	United States	98	Foreman 2006
Sugarcane	Brazil	83	Macedo et al. 2008
Soybean	United States	98	ERS-USDA 2010
Palm	Indonesia/Malaysia	90	Hensen 2003
Rapeseed	United States <sup>b</sup>	97	ERS-USDA 2010

<sup>a</sup>Indicates percentage of corn planted for grain that is harvested for grain.

<sup>b</sup>European percentage harvested data are not available.

For example, palm plantations are not harvested for the first 2.5–3 years after planting and the lifetime of a plantation is 23–30 years (Hensen 2003). Thus, on average, an oil palm plantation is harvested only ~90% of the time. Similarly, sugarcane is typically harvested five times over six years before replanting (Macedo et al. 2008), and short-rotation woody crops must be replanted after several cuttings and typically have lower yields in the first harvest. **Table 6** presents the relevant correction factor for a variety of biofuel feedstocks. Trends for this are flat, so predictions of future land demands should include the correction factors listed here.

### 3.5. Coproduct Discount

The allocation of environmental consequences among coproducts is an area of long-standing controversy in LCA (e.g., Ekvall & Finnveden 2001, Kim & Dale 2002). The approach favored in the International Standards Organization (ISO) standards for LCA (ISO 2006b)—and the only approach that is clearly related to actual environmental outcomes—is to expand the analytical boundaries to encompass the products displaced by the coproducts and to subtract the avoided consequences from those of the system under study (Weidema 2000).

Most biofuels produce coproducts. Biodiesel production also produces a small amount of glycerin (~2% of the oil, by mass). Glycerin is also a byproduct of soap making and vegetable cooking oil production, and its displacement by biodiesel production is unlikely to measurably reduce the amount of land required to make soap or cooking oil. The production of soybean oil yields soybean meal (~82% of the soybean, by mass), and the production of rapeseed oil yields rapeseed meal (~59% of the rapeseed, by mass). The production of crude palm oil also yields palm kernel oil (~19% of total oil in the full fruiting body), along with empty fruit bunches, fiber, and shell (Yusoff & Hansen 2007).

Sugarcane ethanol (and sugar) production also produces bagasse, the biomass left over after cane juice is extracted. Bagasse is typically burned to supply heat and electricity for the ethanol production process (Macedo et al. 2008). When one calculates the climate benefits of sugarcane ethanol, any GHG emissions avoided through the use of bagasse are credited to the ethanol.

Fermentation of grains (e.g., corn, sorghum, wheat, and barley) results in distillers grains and solubles (DGS) that can be fed to livestock, reducing the use of feed grains and oilseed meal. Synthesis of recent analyses of U.S. corn ethanol production suggests that one kilogram of DGS displaces about 0.81–0.95 kg of corn grain and 0.23–0.29 kg of soybean meal in animal feed markets (Arora et al. 2008, Bremer et al. 2010). We calculate that avoiding production of these quantities of corn and soybean—and, thus, the land these crops would require—offsets about 60% of the land required to produce corn for ethanol. However, this does not take into account potential yield decreases from expansion onto marginal lands or from planting corn continuously instead

**Table 7** 2008 global biofuel production and land required for production by country<sup>a</sup>

	Biodiesel		Ethanol		Total	
	Million liters	Million ha	Million liters	Million ha	Million liters	Million ha
United States	2,691	4.34	34,968	8.73	37,659	13.07
European Union	7,998	7.80	2,777	1.59	10,775	9.40
Brazil	1,165	2.16	24,200	3.88	25,365	6.04
Argentina	1,018	1.79	–	–	1,018	1.79
China	–	–	2,521	1.04	2,521	1.04
Canada	–	–	900	0.49	900	0.49
Malaysia	545	0.12	–	–	545	0.12
Indonesia	397	0.11	–	–	397	0.11
Thailand	–	–	340	0.09	340	0.09
India	–	–	250	0.05	250	0.05
Columbia	–	–	300	0.04	300	0.04
Rest of World	902	1.07	702	0.17	1,604	1.23
Total	14,718	17.38	66,336	15.91	81,054	33.30

<sup>a</sup>Production volume data taken from FAPRI 2009, Renewable Fuels Association 2009, and Worldwatch Institute 2009.

of in rotation with soybeans. Because soybean yields are lower than those of corn, the land-use benefit of displacing soybeans is relatively high. Therefore, the land-use displacement of DGS is highly dependent on whether DGS displaces soybeans or corn and varies across livestock type (e.g., beef cattle, dairy cattle, poultry, and swine).

In general, the GHG and land displacement credit to assign to biofuel coproducts is difficult to estimate because these quantities vary according to the specific use of each coproduct and the specific uses depend on market prices. In addition, new uses for coproducts are being developed as larger quantities of inexpensive coproducts are produced.

### 3.6. Global Biofuel Land Use

By synthesizing the above data we find that, globally, 15.9 million ha were required to produce ethanol and 17.4 million ha were required to produce biodiesel in 2008. In total, biofuels required 33.3 million ha in 2008 (**Table 7**). This is about 2.2% of global cropland. Assuming biofuels expand by 170% in 2020, as under a business as usual scenario (International Energy Agency 2009), cropland required for biofuel production would be 72–82 million ha if biofuel production efficiencies (that is, crop yields and conversion efficiencies) increased by 10–25%. Our estimates of the land required to produce biofuels do not include coproduct effects due to lack of data. Research on coproduct effects could help guide biofuel producers toward processes and coproducts that reduce the amount of new land required for biofuel production.

### 3.7. Modeled Land-Use Change

The effects of biofuels on biodiversity and climate change depend not only on how much new land is required for biofuels but also on where that land comes from. Globally, the area used for cropland has been expanding at a rate of about 3.4 million ha year<sup>-1</sup> (FAO 2010). This trend has been apparent over decades, prior to the large-scale production of biofuels, indicating that increasing demand for food is sufficient to drive cropland expansion, independent of biofuel

demand. Therefore, global land demand for crop-based biofuels must be viewed as requiring additional cropland beyond the expansion required to meet food demand.

We can classify the direct source of lands used for biofuel production into one of three categories: crop switching (that is, growing biofuel feedstocks on existing cropland), bringing previously cropped land back into production, or conversion of other land uses to cropland. One notable opportunity is for biofuel production to be targeted on lands that are degraded and abandoned from grazing or cropping. Conversion of current pasture to biofuel production is likely to result in the indirect expansion of grazing lands into forests and other natural ecosystems unless it is paired with increases in the efficiency of livestock production. However, if land availability and biofuel demand are located in different countries, targeting degraded lands for biofuel production may conflict with priorities in consuming nations to use domestically produced biofuels. For example, the use of degraded lands outside the United States to meet ethanol demand in the United States is limited by an import tariff on ethanol.

Most biofuels are not produced directly on land newly cleared from natural ecosystems. Rather, the majority of LUC associated with biofuels is indirect, illustrating that a complete understanding of the LUCs caused by biofuels must go beyond direct impacts.

Land-use change associated with biofuel production has been modeled for two purposes. One purpose has been to examine potential unintended consequences of capping carbon emissions when biofuels are assumed to be carbon neutral. For example, Wise et al. (2009) predict LUC under two carbon tax frameworks for limiting atmospheric concentration of CO<sub>2</sub>. In the first case, fossil fuel and industrial emissions, as well as biogenic emissions, are taxed; in the second case, only fossil fuel and industrial carbon emissions are taxed. When taxing only fossil carbon to limit the atmospheric concentration of CO<sub>2</sub> to 450 ppm, their model predicts massive deforestation, with virtually all natural ecosystems (that is, lands not used for food or forest products) being converted to either food or energy production. In contrast, when biogenic carbon is also taxed, forested land increases (Wise et al. 2009a). Similarly, Melillo et al. (2009a,b) predict that taxing only fossil carbon to limit the CO<sub>2</sub> concentration to 550 ppm would lead to deforestation of more than half of remaining natural forests, with biofuel production covering 16% of the Earth's terrestrial surface. For comparison, cropland currently covers 12% of the Earth's surface. The ILUC as a result of this policy would be nearly twice as large as the area under biofuel feedstock cultivation, and the global warming effect of increased nitrous oxide emissions would be larger than that of increased CO<sub>2</sub> emissions. Although the specific numbers are uncertain, the studies highlight the importance of including ILUC and other indirect effects in regulatory accounting protocols.

A second research purpose has been to support regulatory efforts to mitigate GHG emissions from the transportation sector. These analyses model market-mediated effects using global economic or agricultural-economic models that account for the supply, demand, and substitutability of biofuels, food, and feed (Hayes et al. 2009, Keeney & Hertel 2009, Melillo et al. 2009a, Searchinger et al. 2008, USEPA 2009). These models capture important aspects of economic systems but, as they are not spatially explicit, cannot on their own project where new agricultural production is likely to occur.

Current models use different approaches to predict spatial patterns of LUC. The Food and Agricultural Policy Research Institute (FAPRI) Agricultural Outlook Model (see <http://www.fapri.iastate.edu/>) projects LUC at the national level. The Global Trade Analysis Project (GTAP) model (see <http://www.gtap.agecon.purdue.edu/>) projects LUC for up to 18 agro-ecological zones (AEZs) within each of 18 different geographic regions. AEZs are defined by rainfall and temperature, indicating crop suitability. Predicted LUC within these nations or zones must then be mapped to specific ecosystem types to predict GHG consequences.

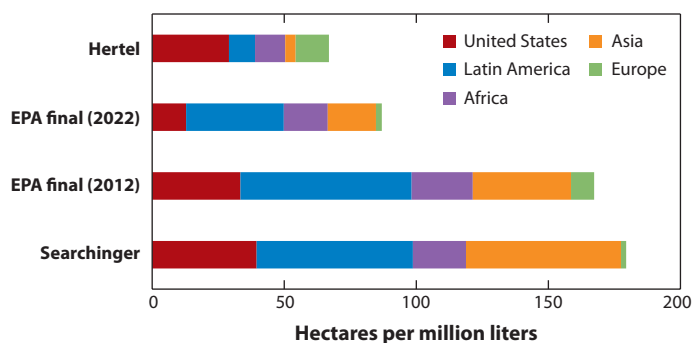
Changes in land use are commonly assumed to occur at the observed agricultural frontier in each region, based on historical records (Hertel et al. 2010, Searchinger et al. 2008) or satellite image analysis (Harris et al. 2009, USEPA 2010). Specifically, some analyses used data compiled by the Woods Hole Research Institute representing carbon stocks and land conversion in the 1990s (Hertel et al. 2010, Searchinger et al. 2008). In contrast, the USEPA, in its regulatory impact analysis for the RFS2, identified the agricultural frontier using a comparison of 2001 and 2007 MODIS satellite data (Harris et al. 2009). In both cases, further changes in land use in each country were assumed to occur at the agricultural frontier in proportion with the historical pattern.

The assumptions used to map economic model results to specific land cover classes are one of several sources of uncertainty in ILUC predictions. The drivers of LUC are diverse (Geist & Lambin 2002, Pfaff et al. 2007), and commodity price pressure may result in a pattern of LUC that differs from the gross pattern of conversion observed in the historical record. Actual patterns of LUC will also be influenced by demand for biomass from the electricity sector and from increasing demand for food from population growth and increasing per capita meat consumption. Understanding biofuels-induced LUC patterns is an important subject for further research.

**Figure 4** shows the distribution of LUC projected by three studies. The Hertel et al. (2010) study is based on GTAP, which, unlike FAPRI, assumes that price effects are transmitted primarily to existing trading partners. This assumption results in a different distribution of LUC, with GTAP predicting relatively more change in the U.S. and less in Latin America and China.

In its rulemaking for RFS2, the USEPA projected yield and efficiency increases in feedstock and biofuel production. As a result, their predicted environmental impact per liter of biofuel decreases over time. For example, USEPA estimates that corn ethanol in 2012 causes LUC emissions per liter that are 152% greater than those projected for 2022. The USEPA final ruling uses the lower projected 2022 estimates to determine compliance with the regulation's GHG performance standards in all years. For example, the USEPA considers corn ethanol production in 2012 to meet the required 20% GHG reduction, even though their own modeling predicts a 12–33% increase in GHG emissions for corn ethanol produced from natural gas-powered dry mills in that year.

The estimates of ILUC for U.S. corn ethanol range from  $67\text{--}179\text{ ha } 10^{-6}\text{ l}^{-1}$  (**Figure 4**). Estimates of ILUC in 2022 for U.S. switchgrass cellulosic ethanol are  $49\text{ ha } 10^{-6}\text{ l}^{-1}$ , for U.S. soybean biodiesel are  $365\text{ ha } 10^{-6}\text{ l}^{-1}$  ( $250\text{ ha } 10^{-6}\text{ l}^{-1}$  on an ethanol energy equivalent basis), and



**Figure 4**

Hectares of land-use change projected to occur in major regions for Searchinger et al. (2008), Hertel et al. (2010), and the U.S. Environmental Protection Agency (USEPA) studies. The USEPA figures are based on the Food and Agricultural Policy Research Institute model for all regions. Although the USEPA used Forest and Agricultural Sector Optimization Model (FASOM) results for the United States, it was not possible to identify the values produced from the FASOM analysis.

for Brazilian sugarcane ethanol are  $88 \text{ ha } 10^{-6} \text{ t}^{-1}$  (USEPA 2010). Land-use change can also be expressed as a net displacement factor, calculated as a ratio of (*a*) the net hectares of new cropland including ILUC to (*b*) the hectares dedicated directly to additional biofuel feedstocks. In general, this ratio is likely to be less than one because of coproduct displacement and price elasticities, although the ratio could be greater than one if biofuel feedstock production occurs on high-yielding cropland that is replaced by low-yielding food production. Projected net displacement factors for 2022 are 0.68 for sugarcane ethanol, 0.43 for corn ethanol, 0.29 for switchgrass cellulosic ethanol, and 0.14 for soybean biodiesel (USEPA 2010).

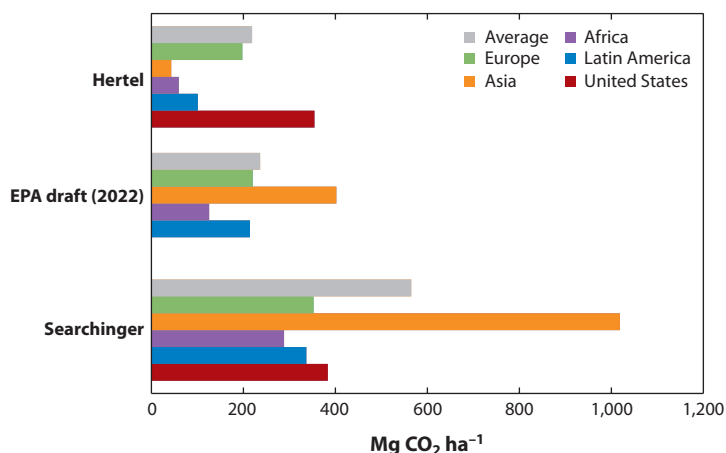
## 4. BIOFUELS AND AIR POLLUTION

Biofuels can decrease or increase both GHG emissions and other air pollutants compared to fossil fuels. Emissions can come from tailpipes, the biofuel production process, or LUC.

### 4.1. Greenhouse Gas Emissions

Until recently, LCA studies of biofuel GHG emissions did not consider LUC and generally concluded that biofuels led to a reduction of GHG emissions compared to fossil fuels (e.g., Farrell et al. 2006). Recent studies have found that the LUC component of GHG emissions from current food-based biofuels is often large enough to offset this benefit.

**4.1.1. Greenhouse gas emissions from land-use change.** Estimates of total LUC GHG emissions attributed to biofuel expansion depend on the number of hectares affected, the type of ecosystem affected, and the per hectare GHG emissions associated with LUC for each affected ecosystem. **Figure 5** shows the areal emission factor ( $\text{Mg CO}_2 \text{ ha}^{-1}$ ) for five key regions based on each model's chosen emission factors and prediction of the type of ecosystem affected. Using GTAP, Hertel et al. (2010) predict a net gain in forestry in Asia, which offsets the emissions



**Figure 5**

Regional emission factors ( $\text{Mg CO}_2 \text{ ha}^{-1}$ ) used by Searchinger et al. (2008), Hertel et al. (2010), and the U.S. Environmental Protection Agency's (USEPA 2009) draft ruling. Note that an emission factor from the USEPA for the United States is not shown, as it was not possible to disaggregate a factor from the published results. Regional definitions are approximate, owing to different aggregation strategies in each model.

for projected pasture conversion in that region, resulting in a low emission factor for Asia. Because GTAP considers only forests in economic use (that is, those managed for timber), it does not model the conversion of unmanaged forests to other uses, potentially underestimating Asian ILUC emissions.

The differences in emission factors for a given region could result from different assumptions about the type of ecosystem being converted (e.g., grassland, shrubland, forest, wetland, or peatland) or from different assumptions about the emission factor for a given ecosystem. Unfortunately, it is not possible to disaggregate ecosystem-specific emission factors from all the published models shown in **Figure 5**.

Emission factors are a function of [see supplementary material of Fargione et al. (2008)]: (a) aboveground biomass, (b) belowground biomass (20–56% and 64–480% of aboveground biomass for forests and grassland, respectively; Mokany et al. 2006), (c) dead biomass (estimated as a fraction of aboveground biomass), (d) soil carbon, (e) the fraction of soil carbon lost (or gained) upon conversion, (f) the fraction of woody biomass retained in long-lived forest products (lumber has a ~30-year half life), (g) the fraction of aboveground biomass remaining as charcoal carbon after fire to clear land (1.9–4.5%), and (h) the amount of carbon retained in converted lands (e.g., 36–48 Mg C in biomass ha<sup>-1</sup> in oil palm). In forests, the largest source of variation is associated with estimates of aboveground biomass (e.g., 161–270 Mg C ha<sup>-1</sup> in Southeast Asia lowland tropical rainforest; Fargione et al. 2008), but new methods can reduce this uncertainty (Gibbs et al. 2007). In grasslands, the largest source of variation is the fraction of soil carbon lost upon conversion. The largest emission factors are associated with the conversion of peatlands in Southeast Asia. These lands not only have the highest aboveground biomass, but also release 37–73 Mg CO<sub>2</sub> ha<sup>-1</sup> year<sup>-1</sup> due to decomposition of drained organic soils (Fargione et al. 2008).

Black carbon (commonly referred to as soot) is an underappreciated contributor to global warming (Bond & Sun 2005, Ramanathan & Carmichael 2008). Open fires associated with land clearing in tropical forests are a major source of black carbon in the atmosphere. Black carbon in the atmosphere efficiently absorbs light in the visible spectrum, raising atmospheric temperatures and decreasing surface temperatures (Ramanathan & Carmichael 2008). The global warming forcing of black carbon emissions attributable to LUC associated with transportation biofuels (or burning sugarcane fields used to produce sugarcane ethanol) has not yet been quantified.

**4.1.2. Nonland-use life-cycle emissions.** Attributional LCA studies suggest that the GHG emissions associated with corn ethanol production in a recent vintage natural gas-fired dry mill are ~61 g CO<sub>2</sub>e MJ<sup>-1</sup>, compared with ~94 g CO<sub>2</sub>e MJ<sup>-1</sup> for gasoline (Plevin 2009). However, results vary significantly across studies owing to different choices of system boundaries, coproduct allocation methods, data vintage, and the specific technologies modeled (Farrell et al. 2006, Plevin 2009). **Table 8** shows select values for life-cycle GHG emissions adopted by the California Air Resources Board for a range of fuels (available at [http://www.arb.ca.gov/fuels/lcfs/121409lcfs\\_lutables.pdf](http://www.arb.ca.gov/fuels/lcfs/121409lcfs_lutables.pdf)).

Indirect or market-mediated effects extend beyond land-use effects. Of the models discussed here, only the USEPA has attempted to estimate indirect non-LUC GHG emissions. The USEPA's analysis includes emissions of methane (CH<sub>4</sub>) from enteric fermentation and rice production; CO<sub>2</sub>, CH<sub>4</sub>, and N<sub>2</sub>O for on-farm energy use; N<sub>2</sub>O emissions from nitrogen fertilizer application; and CO<sub>2</sub> fluxes associated with changes in tillage practices.

Another indirect economic effect of biofuels production is the global petroleum rebound effect, in which petroleum consumption rebounds to make up for some of the fuel displaced by biofuels. This occurs because the production and use of biofuels reduces petroleum demand, thereby decreasing the price of petroleum globally and leading to increased consumption relative to the case

**Table 8 Carbon intensity values assigned to various fuels under the California Low-Carbon Fuel Standard**

Fuel	Pathway description	Carbon Intensity Values (g CO <sub>2</sub> e MJ <sup>-1</sup> )		
		Direct Emissions	Indirect Emissions	Total
Gasoline	CARBOB (avg. crude)	96	0	96
Diesel	Ultra-low sulfur diesel; average crude	95	0	95
Corn ethanol	Midwest average corn ethanol	>69	>30	>99
	California average corn ethanol	66	30	96
	California dry mill; wet DGS	>51	>30	>81
	Midwest dry mill; wet DGS	60	30	90
Sugarcane ethanol	Brazilian average	27	46	73
	Brazilian average; mechanized harvesting and electricity credit	12	46	58
Biodiesel	Midwest soybean FAME	21	62	83

Abbreviations: CARBOB, California reformulated gasoline blendstock for oxygenate blending; DGS, distillers grains and solubles; FAME, fatty acid methyl ester.

without biofuels (USEPA 2010, p. 512). Stoft (2010) estimates this rebound effect based on the results of prior modeling studies of global petroleum markets, showing a range from 29% to 70%. If our best estimate of the rebound effect is, say, 32%, then to properly estimate the net GHG benefits of biofuels, we need to add back in 0.32 units of petroleum for each energy-equivalent unit of biofuel produced. Of course, the magnitude of the rebound effect is uncertain, as it depends on the response of OPEC and other producers to changes in demand. Nonetheless, the potential magnitude of this effect demonstrates the need to consider the GHG consequences of biofuel effects in the context of the global petroleum market.

## 4.2. Emissions Affecting Air Quality

Research on the air pollution impacts of biofuels has focused on GHG, but air quality is also affected by a multitude of other compounds emitted during fuel production and use (Hess et al. 2009). These pollutants affect human health (e.g., cardiovascular disease, cancer, and birth defects) and the environment (e.g., acid rain, decreased crop production, and reduced visibility).

Both fuel production and end use (tailpipe emissions) are considered when estimating the full life-cycle emissions of pollutants affecting air quality. Tailpipe emissions affecting air quality have been well characterized for end-use combustion of biofuels, which depend upon vehicle type, vehicle emission standards, and biofuel blend levels. In recent reviews, Jacobson (2007) and Yanowitz & McCormick (2009) found that burning ethanol in an E85 blend (85% fuel grade ethanol blended with 15% gasoline) results in reduced emissions of nitrogen oxides, benzene, and 1,3-butadiene, but also in increased emissions of ethanol, formaldehyde, and acetaldehyde. Similarly, McCormick (2007) found biodiesel blends tend to reduce particulate matter, carbon monoxide, total hydrocarbons, and many toxic compounds, but can increase nitrogen oxides at higher blending levels.

Including fuel production in a well-to-wheel emissions assessment indicates that volatile organic compounds, carbon monoxide, nitrogen oxides, particulate matter, and sulfur oxides are higher for ethanol from corn burned as E85 compared to gasoline (Brinkman et al. 2005). The net damage to human health and the environment caused by these and other pollutants is not solely determined by the quantities emitted, however, but also on how, where, and when fuels are produced and



used. Hill et al. (2009), employing a spatially explicit LCA, estimated that the economic damages to human health due to life-cycle emissions of fine particulate matter ( $PM_{2.5}$ ) from ethanol are higher than those of gasoline when the ethanol is produced from corn, but lower when the ethanol is produced from cellulosic biomass.

As with GHG, LUC may have important consequences for air quality. Oil palm plantations in Malaysia emit more nitrogen oxides and volatile organic compounds than rainforest, which under increased biofuel production may lead to higher levels of ground-level ozone (Hewitt et al. 2009). In Brazil, conversion of land to sugarcane ethanol production may cause increased ground-level ozone (Goldemberg et al. 2008) and particulate matter levels (Cançado et al. 2006) from the burning of fields in preparation for harvest, in part leading to a federal decree to eliminate burning by 2017 (Macedo et al. 2008).

## 5. BIOFUELS AND WATER

Biofuel crops require water for growth and for processing and can lead to increased demand for water. Water used in processing is a potential point source of pollution, but is relatively easily regulated and treated. The volume of water used in the growth of crops is much greater than that used in processing, and this non-point source of water pollution is less easily regulated, so we focus on the impacts of water pollution from growing biofuel crops.

### 5.1. Water Use

The amount of water applied in biofuel production is highly dependent on whether the feedstock is irrigated (**Table 9**). For example, excluding agricultural water use, ~3 gallons of water are required at ethanol plants for every gallon of corn ethanol produced (Wu et al. 2009). However, irrigated corn requires ~643 gallons of irrigation water for every gallon of ethanol. Since 18% of corn comes from land that is irrigated (larger than the percent of corn area irrigated because irrigated acres have above average yields), each gallon of corn ethanol uses, on average, 115 gallons of irrigation water. However, because much of the land suitable for rainfed agriculture in the United States is already in production, corn expansion in the United States is occurring disproportionately on land that requires irrigation. This means that the marginal water use by corn is higher than the average water use. Between 2003 and 2008, area harvested for corn grain increased by 3.1 million ha and irrigated area increased by 0.9 million ha, suggesting that approximately 29% of the new corn area is irrigated and that 34% of new corn production comes from irrigated land (because of the higher yields on irrigated land). Assuming corn ethanol production uses average new corn, 216 gallons of irrigation water are used per gallon of ethanol. Only 60–80% of this water is consumed by evapotranspiration, leaving the rest to enter surface waters (Mubako & Lant 2008). However, because about 90% of irrigated corn acres are irrigated with ground water (USDA 2009), which may be withdrawn at unsustainable rates (Roberts et al. 2007), it is useful to include all water withdrawals in our measure of impact on water resources.

At present, oil palm is rarely irrigated, but research suggests that it could be profitable to do so, indicating that intensified production practices could lead to greater water use for oil palm. Similarly, although only 1% of Brazilian sugarcane is irrigated, 30% of global sugarcane area is irrigated. In general, expansion of biofuel crops (including potential crops such as *Miscanthus* or *Jatropha*) onto marginally suitable lands is likely to require more irrigation than they currently receive.

A full accounting of the consequences of biofuel production for water use must also take into account the opportunity cost that occurs when biofuels use land suitable for rainfed crops (Fingerman et al. 2010). In the absence of biofuels, precipitation on these lands could be used for

**Table 9** Water use by biofuel crops

Feedstock	Country	Process water (liter/liter biofuel)	Irrigated crops: water (liter/liter biofuel)	Percent irrigated	Total Water (liter/liter biofuel)	Total Water (liter/liter gasoline equivalent)	References
Corn	United States	3	643	18%	118	177	USDA 2009
Sugarcane	Brazil	10	1,211	1%	22	33	Evans & Cohen 2009, Laclau & Laclau 2009
Cellulosic biochemical	Current–Advanced	5.9–9.5	–	–	5.9–9.5	8.9–14.3	Wu et al. 2009
Cellulosic thermochemical	— <sup>a</sup>	1.9	–	–	1.9	2.9	Wu et al. 2009
Soybean	United States	3	23,222	12%	2709	2,574	USDA 2009
Rapeseed	Europe	3	–	0	3	2.9	van der Velde et al. 2009
Palm	Indonesia/Malaysia	3	–	–	3	2.9	National Research Council 2008
Gasoline	United States	1.5	2.1–5.4 (extraction)	–	3.6–6.9	3.6–6.9	Wu et al. 2009

<sup>a</sup>No commercial production; estimates based on laboratory research.

other crops, for environmental services, or for reservoir and groundwater recharge. Accounting for this embedded water is accomplished through modeling evapotranspiration in crop growth. However, the actual impact of the water consumed in biofuel production will depend on local water supplies and demands and the water uses being displaced by shifting land use to biofuel production.

## 5.2. Water Quality

Biofuels impact water quality through the application of fertilizers and pesticides as well as changes in tillage practices, which affect pollution and sediment loads. Impacts from fertilizer and pesticides tend to increase with the rate of application and with the amount of leaching and runoff. Biofuel feedstocks range widely in the amount of fertilizer applied (**Table 10**). Leaching of nutrients is much lower under perennial grasses compared to annual crops. For example, the amount of nitrate leaving tile-drained fields planted to grass was 98% lower than the amount leaving continuous corn (Randall et al. 1997).

Increased demand for corn may lead farmers to plant continuous corn rather than corn-soybean rotations, which leads to increased nutrient applications for several reasons. First, soybeans, as with other legumes, can acquire nitrogen from the atmosphere, and therefore require low rates of fertilization. Thus, replacing soybeans with corn increases fertilizer application. Second, corn planted after soybeans requires less fertilizer, because soybean residue acts as a green fertilizer. For example, in Iowa it is estimated that corn after soybeans receives 57 kg ha<sup>−1</sup> less fertilizer than continuous corn (Hennessy 2006). Consequently, Donner & Kucharik (2008) predict an increase of dissolved inorganic nitrogen export to the Gulf of Mexico of 10–34% due to the

**Table 10 Fertilizer use by biofuel crop**

Feedstock	Location	Nitrogen fertilizer applied (kg ha <sup>-1</sup> )	Phosphorus fertilizer applied (P <sub>2</sub> O <sub>5</sub> kg ha <sup>-1</sup> )	Reference
Corn	United States	146.1	53.1	Hill et al. 2006
Sugarcane	Brazil	70.5	41.7	Macedo et al. 2008
Soybean	United States	5.6	17.2	Hill et al. 2006
Palm fresh fruit bunches	Malaysia	48	28	Yusoff & Hansen 2007
Rapeseed	Europe	153	44	EUROSTAT 2008
Switchgrass	Northern Great Plains, U.S.A.	74	0	Schmer et al. 2008

ethanol production that will be required to meet the current U.S. renewable fuels standard. The increased fertilizer application rate on cropland once it is shifted to corn, along with increased crop acreage, will increase the size of the hypoxic zone in the Gulf of Mexico and very likely will preclude reaching nitrogen reduction goals set for the watershed (Donner & Kucharik 2008).

Increased biofuel production may also affect tillage practices. Perennial crops do not require tillage and thus cause less soil erosion and water quality impacts than do annual crops. No-till practices can also be applied with annual biofuel crops. However, compared to corn-soybean rotations, continuous corn is less likely to be managed with no-till because continuous corn without tillage typically results in yield decreases associated with residue build-up (e.g., Vyn et al. 2000). Therefore, to the extent that it increases the use of continuous corn, demand for corn ethanol may reduce the use of no-till. However, some research suggests that no-till farming on continuous corn may avoid yield decreases if practiced in combination with stover removal, which could provide biomass for cellulosic ethanol or bioenergy production (Moebius-Clune et al. 2008). Although no-till has other clear benefits, the hypothesized carbon benefits of no-till are questionable because shifting from conventional-till to no-till can increase N<sub>2</sub>O emissions (Six et al. 2004), and reported soil carbon increases with no-till may be an artifact of sampling soils only to shallow depths (Angers & Eriksen-Hamel 2008, Baker et al. 2006, Blanco-Canqui & Lal 2008, Yusoff & Hansen 2007).

## 6. ECOLOGICAL IMPACTS

The impacts of biofuels on biodiversity depend on which habitats are replaced and which crop and management practices they are replaced with, and are thus highly context specific. We review relevant literature on biofuel impacts in the United States, Brazil, and Southeast Asia.

### 6.1. United States

Between 2006 and 2007, the area planted with corn in the United States increased by 6.2 million ha. At the same time, the land planted with soybeans decreased by 4.4 million ha. Clearly, the majority of direct conversion to accommodate increased corn production happened through crop switching. However, previously planted cropland also came back into production. In the United States, the largest pool of previously cropped lands is the CRP, a cropland retirement program designed to support crop prices and provide environmental benefits by putting marginal cropland back into perennials. In the fall of 2007, 931,000 ha left the CRP. Recent legislation requires a further decrease of CRP area by at least 1.0 million ha. There is also newly cropped land: Between 2002 and 2007, at least 203,000 ha in North Dakota, South Dakota, and Montana that had never

been plowed before were converted to cropland (Fargione et al. 2009). This continues a pattern of conversion in which 10 million ha of private grassland was converted, primarily to cropland, between 1982 and 2003 (GAO 2007).

Land demand for biofuel crops could result in the conversion of grassland habitat to monocultures of corn or other dedicated energy crops. Water quality impacts will depend heavily on the crop type (for example, whether the crop is annual or perennial) and management practices such as fertilizer and pesticide use. However, habitat loss will occur with practically all crop types (Fargione et al. 2009). In a meta-analysis, Fletcher et al. (2010) report that animal diversity (as measured with, e.g., species richness or Shannon's Index) in row crops was reduced by about 60% compared to reference habitat. Further, Fletcher et al. found that the response of bird abundance to row crops was negatively correlated with an index of conservation concern, indicating that birds of greater conservation concern are more threatened by conversion to row crops. Vertebrate diversity and abundance was higher in CRP than in row crops (Fletcher et al. 2010), consistent with other work demonstrating the wildlife benefits of CRP (reviewed in Fargione et al. 2009). Further, beneficial insect taxa are more abundant and diverse in switchgrass and native prairie than in corn (Gardiner et al. 2010). These results suggest that native prairie plants such as switchgrass or diverse native prairie could provide habitat benefits compared to row crops, especially if managed to maintain habitat value (Fargione et al. 2009, Tilman et al. 2006).

## 6.2. Brazil

In Brazil, both sugarcane and soybeans are planted primarily in the tropical savanna (Cerrado) located south of the Amazon rainforest. Lapola et al. (2010) used an LUC model to predict the effects of the expansion of ethanol and biodiesel in Brazil by 2020. They predict that 97% of new sugarcane and soybeans will be grown on existing rangeland and cropland, and that indirect effects will cause the conversion of 12.2 million ha of forest and 4.6 million ha of other natural habitats. Their results also illustrate the importance of changes in the efficiency of livestock production. Their model assumes an increase in livestock density of 0.09 head per ha, but an increase of 0.13 head per ha would avoid the need for ILUC by fulfilling food and biofuel demand on existing cropland and rangeland (Lapola et al. 2010).

Direct habitat impacts of biofuel expansion are likely to occur primarily in the Cerrado but may also impinge on remaining Atlantic forest (Lapola et al. 2010). Most of the indirect effects are likely to occur in the Amazon (Lapola et al. 2010). All three of these areas support remarkable biodiversity that requires intact native habitat and secondary forest for survival.

The Cerrado contains over 7,000 plant species, with high levels of endemism (Klink & Machado 2005). The Cerrado is also rich in species of birds, fish, reptiles, amphibians, and insects. Numerous species are threatened with extinction, with notable risks to the estimated 20% of species that do not occur in protected areas. Only 2.2% of its area is under legal protection; in the past 35 years, over 50% of its approximately 200 million ha have been converted to pasture and cropland. Conversion in the Cerrado is ongoing, with a greater area of conversion per year than in the Amazon. Pasture lands are planted with exotic grasses that create a more frequent and intense fire regime and are incompatible with most native species (Klink & Machado 2005).

The Brazilian Atlantic forest is one of the most diverse biomes on Earth and may contain 1–8% of the Earth's total species (Ribeiro et al. 2009). This includes species of more than 20,000 plants, 261 mammals, 688 birds, 200 reptiles, and 280 amphibians, in addition to the species not yet described by science (Ribeiro et al. 2009). The Brazilian Atlantic Forest is now at least 84% deforested, and what remains is highly fragmented and poorly protected. More than 80% of remaining forest fragments are smaller than 50 ha, and almost half the remaining forest is less

than 100 m from its edges (Ribeiro et al. 2009). Nature reserves protect only 9% of remaining forest (Ribeiro et al. 2009). Protection of remaining Atlantic Forest and forest restoration may conflict with increasing sugarcane production (Lapola et al. 2010).

Historic conversion of the Amazon has been associated primarily with grazing by smallholders (Fearnside 2005). Rates of deforestation have historically been tied to rural population growth, but globalization has increased the role of intensive commodity production (crops, livestock, and lumber) in deforestation over the past two decades (Barona et al. 2010, Rudel et al. 2009). This increased role of commodity production as a driver of deforestation helps explain why Brazil and Indonesia comprise an increasing share of global tropical deforestation, up from about 20% in the 1980s to 40–60% between 2000–2005 (Rudel et al. 2009). If deforested lands are not cultivated or grazed, secondary forest can, within decades, support a surprisingly large number of forest species (Chazdon et al. 2009). However, LUC also threatens secondary forests. For example, in areas of eastern Amazon already dominated by human settlement, grazing area expanded between 1994 and 2003, even though it had remained constant during the preceding decade (de Almeida et al. 2010).

In total, increased demand for agricultural commodities will continue to be a driver behind the conversion of primary and secondary forests in Brazilian tropical forests and savannas. Increased protection of natural areas in these species-rich areas is necessary to preserve biodiversity in the face of these pressures (Brooks et al. 2009).

### 6.3. Southeast Asia

Koh & Wilcove (2008) estimate that the majority of new palm plantations between 1990–2009 in Indonesia and Malaysia were established on former primary and secondary forest land. This forest conversion threatens the remarkable biodiversity found in these rainforests: Across all animal taxa, in paired comparisons, 85% of species found in primary forest were absent in palm plantations (Fitzherbert et al. 2008). Trees, lianas, epiphytic orchids, and native palms are completely absent from palm plantations (Danielsen et al. 2009). Logged forests, when not converted to oil palm plantations, will recover much of their biodiversity value (Danielsen et al. 2009, Fitzherbert et al. 2008, Koh & Wilcove 2008). Oil palm plantations support less biodiversity than most alternative agricultural crops, including rubber, cocoa, coffee, and *Acacia* plantations (Fitzherbert et al. 2008, Koh & Wilcove 2008). Oil palm plantations are also likely to affect biodiversity in adjacent habitat through fragmentation and edge effects, such as changes in the microclimate, sunlight penetration, and fire regime (Fitzherbert et al. 2008).

In general, palm plantations tend to be dominated by one or a few generalist species that are not of conservation concern, such as exotic and invasive species (Danielsen et al. 2009). For example, 40% of ant species found in oil palm plantations were exotic, including the highly invasive crazy ant *Anoplolepis gracilipes* (Fitzherbert et al. 2008). Oil palm production also involves pollutants including rodenticides, insecticides, herbicides, fertilizers, and POME (palm oil mill effluent). POME is usually purified so it can be discharged with minimal environmental impact, and integrated pest management and leguminous cover crops can reduce the need for insecticides, herbicides, and fertilizers. However, even under ideal management practices, oil palm plantations can support only a small fraction of the biodiversity of primary forests. For example, the abundance of birds and butterflies in oil palm plantations was primarily influenced by the amount of adjacent forest cover and was relatively insensitive to variations in weed and epiphyte abundance within the plantation (Koh 2008). In total, the biggest opportunity for oil palm plantations to improve biodiversity is through avoiding conversion of forests, as compared to improving the habitat value of plantations themselves. Avoiding forest conversion can come, at least in part, from yield

**Table 11** Traits of biomass energy crops and invasiveness

Plant traits desirable for biomass energy crops	Traits that increase probability of invasiveness?
Perennial	N
Sterile seeds	N
Rapid growth in spring	Y
Long canopy duration	Y
Grows at high densities	Y
High water-use efficiency	Y
Tolerates saline soils	Y
Re-allocates nutrients belowground in fall	Y
No major pests/diseases	Y

increases and from targeting expansion onto degraded lands. For example, the invasive grass *Imperata cylindrica* covers at least 8.5 million ha in Indonesia (Fitzherbert et al. 2008) and could be converted to oil palm plantations with minimal adverse effects on biodiversity.

Expansion of oil palm in other areas around the world raises similar threats to biodiversity. There are 746 million ha of tropical forests that are suitable for palm oil production (Stickler et al. 2007). In the second half of 2008 alone, at least half a million hectares of new oil palm concessions were announced in the Brazilian Amazon, Papua New Guinea, and Madagascar, all of which are biodiversity hotspots (Koh et al. 2009).

## 6.4. Energy Crops and Invasion

Current biofuel crops are dependent on human cultivation and do not pose a risk of becoming invasive. However, proposed biomass energy crops pose significant risk of becoming invasive and causing unintended effects on species (Wilcove et al. 1998) and ecosystem services (Crooks 2002), with significant costs associated with damage and invasive species control.

Many of the traits that make plants desirable for biomass energy production, such as rapid growth, also increase the risk of their becoming invasive (Table 11; Barney & Ditomaso 2008, Council for Agricultural Science and Technology 2007, Raghu et al. 2006). Consequently, biomass energy crops are more likely to pose a risk of invasion than are other introduced plants (Buddenhagen et al. 2009, Mack 2008). Weed risk assessment applied to potential biomass energy crops could help focus biomass energy crop development efforts on species that pose less risk of invasion (Cousens 2008, Davis et al. 2010).

## 7. CONCLUSION: MITIGATING THE ECOLOGICAL IMPACT OF BIOFUELS

Biofuels are the most land-intensive form of energy production (McDonald et al. 2009). The land requirements for biofuels have potential negative consequences for biodiversity and GHG emissions by causing, either directly or indirectly, the conversion of natural ecosystems to cropland. Although the magnitudes of these effects are poorly constrained, we can identify strategies to mitigate these effects. Development that follows the mitigation hierarchy can dramatically improve outcomes for biodiversity (Kiesecker et al. 2010). The mitigation hierarchy follows these steps for development: (a) avoid sensitive areas, (b) minimize impacts through best practices, (c) restore areas



after use, and (d) fund compensatory offsite mitigation. In the case of conversion to biofuel crops, avoiding sensitive areas is perhaps the most important and complex of these steps. There are five steps that must be taken to help biofuels avoid impacts on sensitive areas. First, the use of biofuel feedstocks that do not compete with food for land, such as wastes, residues, cover crops, and forest thinnings, should be favored (Tilman et al. 2009). Second, abandoned and degraded cropland and rangeland should be identified and targeted for biofuel production (Campbell et al. 2008). Third, increasing yields, coproduct offsets, and livestock production efficiencies will better allow existing agricultural land to meet increasing demand, reducing the amount of new demand that is met with agricultural expansion and therefore reducing indirect effects. Fourth, certification standards must include prohibitions on direct conversion of natural ecosystems. Fifth, mechanisms protecting natural areas must be bolstered. Specifically, the societal goal of reducing GHG emissions by replacing petroleum with renewable energy sources can be met only if there is substantially increased investment in protecting high-carbon natural areas (Wise et al. 2009). Payments for the ecosystem service of carbon storage could be made through carbon offset mechanisms such as REDD (reduced emissions from deforestation and degradation). Payments could also be made as a part of compensatory offset mitigation associated with biofuel production (Melillo et al. 2009b). This last step in the mitigation hierarchy has been largely ignored in the context of biofuels but could play an important role in mitigating the impacts of biofuels on biodiversity.

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## LITERATURE CITED

- Angers DA, Eriksen-Hamel NS. 2008. Full-inversion tillage and organic carbon distribution in soil profiles: a meta-analysis. *Soil Sci. Soc. Am. J.* 72:1370–74
- Arora S, Wu M, Wang M. 2008. Update of distillers grains displacement ratios for corn ethanol life-cycle analysis, Cent. Transp. Res., Energy Syst. Div., Argonne Natl. Lab.
- Aylott MJ, Casella E, Tubby I, Street NR, Smith P, Taylor G. 2008. Yield and spatial supply of bioenergy poplar and willow short-rotation coppice in the UK. *New Phytol.* 178:358–70
- Babcock BA. 2009. Measuring unmeasurable land-use changes from biofuels. *Iowa Ag Rev.* 15:No. 3
- Babcock BA, Carriquiry M. 2010. An exploration of certain aspects of CARB's approach to modeling indirect land use from expanded biodiesel production. Staff report 10-SR 105. Center for Agriculture and Rural Development, Ames, Iowa
- Baker JM, Ochsner TE, Venterea RT, Griffiths TJ. 2006. Tillage and soil carbon sequestration—what do we really know? *Agric. Ecosyst. Environ.* 118:1–5
- Barney JN, Ditomasso JM. 2008. Nonnative species and bioenergy: Are we cultivating the next invader? *BioScience* 58:64–70
- Barona E, Ramankutty N, Hyman G, Coomes OT. 2010. The role of pasture and soybean in deforestation of the Brazilian Amazon. *Environ. Res. Lett.* 5:024002
- Blanco-Canqui H, Lal R. 2008. No-tillage and soil-profile carbon sequestration: an on-farm assessment. *Soil Water Manag. Conserv.* 72:693–701



- Bond TC, Sun H. 2005. Can reducing black carbon emissions counteract global warming? *Environ. Sci. Technol.* 39:5921–26
- Bremer VR, Liska AJ, Klopfenstein TJ, Erickson GE, Yang H, et al. 2010. Emission savings in the corn-ethanol life cycle from feeding coproducts to livestock. *J. Environ. Qual.* 39:472–82
- Brinkman N, Wang M, Weber T, Darlington T. 2005. Well-to-wheels analysis of advanced fuel/vehicle systems—a North American study of energy use, greenhouse gas emissions, and criteria pollutant emissions. Argonne, IL: GM Corp./Argonne Natl. Lab., May
- Brooks TM, Wright SJ, Sheil D. 2009. Evaluating the success of conservation actions in safeguarding tropical forest biodiversity. *Conserv. Biol.* 23:1448–57
- Buddenhagen CE, Chimera C, Clifford P. 2009. Assessing biofuel crop invasiveness: a case study. *PLoS ONE* 4(4):e5261
- Campbell JE, Lobell DB, Genova R, Field CB. 2008. The global potential of bioenergy on abandoned agricultural lands. *Environ. Sci. Technol.* 42:5791–94
- Cançado JED, Saldiva PHN, Pereira LAA, Lara LBLS, Artaxo P, et al. 2006. The impact of sugar cane-burning emissions on the respiratory system of children and the elderly. *Environ. Health Perspect.* 114:725–29
- CARB. 2009. *Proposed Regulation to Implement the Low Carbon Fuel Standard, Vol. I, Staff Report: Initial Statement of Reasons*. Sacramento, CA: Calif. Air Resourc. Board
- Chazdon RL, Peres CA, Dent D, Sheil D, Lugo AE, et al. 2009. The potential for species conservation in tropical secondary forests. *Conserv. Biol.* 23:1406–17
- Counc. Agric. Sci. Technol. 2007. *Biofuel Feedstocks: The Risk of Future Invasions*. Ames, IA: CAST
- Cousens R. 2008. Risk assessment of potential biofuel species: an application for trait-based models for predicting weediness? *Weed Sci.* 56:873–82
- Crooks JA. 2002. Characterizing ecosystem-level consequences of biological invasions: the role of ecosystem engineers. *Oikos* 97:153–66
- Danielsen F, Beukema H, Burgess ND, Parish F, Bruhl CA, et al. 2009. Biofuel plantations on forested lands: double jeopardy for biodiversity and climate. *Conserv. Biol.* 23:348–58
- Davis AS, Cousens RD, Hill J, Mack RN, Simberloff D, Raghu S. 2010. Screening bioenergy feedstock crops to mitigate invasion risk. *Front. Ecol. Environ.* DOI: 10.1890/090030
- de Almeida AS, Stone TA, Vieira ICG, Davidson EA. 2010. Nonfrontier deforestation in the Eastern Amazon. *Earth Interact.* 14:1–15
- Donner SD, Kucharik CJ. 2008. Corn-based ethanol production compromises goal of reducing nitrogen export by the Mississippi River. *Proc. Natl. Acad. Sci. USA* 105:4513–18
- Ekvall T, Finnveden G. 2001. Allocation in ISO 14041—a critical review. *J. Clean. Prod.* 9:197–208
- Ekvall T, Weidema BP. 2004. System boundaries and input data in consequential life cycle inventory analysis. *Int. J. Life Cycle Assess.* 9:161–71
- ERS-USDA. 2010. *ERS-USDA Data—Feed Grains Database Custom Queries*. <http://www.ers.usda.gov/Data/Feedgrains/CustomQuery>.
- EUROSTAT. 2008. *Mineral fertiliser consumption*. [http://epp.eurostat.ec.europa.eu/portal/page/portal/agri\\_environmental\\_indicators/indicators\\_overview/farm\\_management\\_practices](http://epp.eurostat.ec.europa.eu/portal/page/portal/agri_environmental_indicators/indicators_overview/farm_management_practices)
- Evans JM, Cohen MJ. 2009. Regional water resource implications of bioethanol production in the Southeastern United States. *Glob. Change Biol.* 15:2261–73
- FAO. 2008. *The State of Food and Agriculture—Biofuels: Prospects, Risks and Opportunities*. Rome: Food Agric. Organ. UN
- FAO. 2010. *FAOSTAT*. <http://faostat.fao.org/site/377/DesktopDefault.aspx?PageID=377#ancor>
- FAPRI. 2009. *U.S. and World Agricultural Outlook*. Columbia, MO: Food Agric. Policy Res. Inst.
- Fargione J, Hill J, Tilman D, Polasky S, Hawthorne P. 2008. Land clearing and the biofuel carbon debt. *Science* 319:1235–38
- Fargione JE, Cooper TR, Flaspohler DJ, Hill J, Lehman C, et al. 2009. Bioenergy and wildlife: threats and opportunities for grassland conservation. *BioScience* 59:767–77
- Farrell AE, Plevin RJ, Turner BT, O'Hare M, Jones AD, Kammen DM. 2006. Ethanol can contribute to energy and environmental goals. *Science* 311:506–8
- Fearnside PM. 2005. Deforestation in Brazilian Amazonia: history, rates, and consequences. *Conserv. Biol.* 19:680–88

- FEDIOL. 2010. *Oil and meal yields*. <http://www.fediol.be/2/index.php>.
- Feng H, Rubin OD, Babcock BA. 2010. Greenhouse gas impacts of ethanol from Iowa corn: life cycle assessment versus system wide approach. *Biomass Bioenergy* 34:912–21
- Fingerman KR, Torn MS, O'Hare MH, Kammen DM. 2010. Accounting for the water impacts of ethanol production. *Environ. Res. Lett.* 5:014020
- Fitzherbert EB, Struebig MJ, Morel A, Danielsen F, Brulh CA, et al. 2008. How will oil palm expansion affect biodiversity? *Trends Ecol. Evol.* 23:538–45
- Fletcher RJ Jr, Robertson BA, Evans J, Doran PJ, Alavalapati JRR, Schemske DW. 2010. Biodiversity conservation in the era of biofuels: risks and opportunities. *Front. Ecol. Environ.* DOI: 10.1890/090091
- Foreman LF. 2006. Characteristics and production costs of U.S. corn farms, 2001. *Econ. Inf. Bull. No. 7*, U.S. Dept. Agric., Washington, DC
- GAO. 2007. Agricultural conservation: Farm program payments are an important factor in landowners' decisions to convert grassland to cropland. *Rep. GAO-07-1054*. Washington, DC: U.S. Gov. Acc. Off.
- Gardiner MA, Tuell JK, Isaacs R, Gibbs J, Ascher JS, Landis DA. 2010. Implications of three biofuel crops for beneficial arthropods in agricultural landscapes. *Bioenergy Res.* 3:6–19
- Geist HJ, Lambin EF. 2002. Proximate causes and underlying driving forces of tropical deforestation. *BioScience* 52:143–50
- Gibbs HK, Brown S, Niles JO, Foley JA. 2007. Monitoring and estimating tropical forest carbon stocks: making REDD a reality. *Environ. Res. Lett.* 2:045023
- Goldemberg J, Coelho ST, Guardabassi P. 2008. The sustainability of ethanol production from sugarcane. *Energy Policy* 36:2086–97
- Graham JH, Duncan LW, Eissenstat DM. 1997. Carbohydrate allocation patterns in citrus genotypes as affected by phosphorus nutrition, mycorrhizal colonization and mycorrhizal dependency. *New Phytol.* 135:335–43
- Graham RL, Nelson R, Sheehan J, Perlack RD, Wright LL. 2007. Current and potential US corn stover supplies. *Agron. J.* 99:1–11
- Harris N, Grimland S, Brown S. 2009. Land use change and emission factors: updates since the RFS proposed rule. Winrock Int., Arlington, VA
- Hayami Y, Ruttan VW. 1971. Induced innovation and agricultural development, Staff Pap. Ser., Staff Pap. No. P71-1. Univ. Minn., Dept. Agric. Appl. Econ., St. Paul
- Hayes D, Babcock B, Fabiosa J, Tokgoz S, Elobeid A, et al. 2009. Biofuels: potential production capacity, effects on grain and livestock sectors, and implications for food prices and consumers. *J. Agric. Appl. Econ.* 41:1–27
- Heaton E, Voigt T, Long SP. 2004. A quantitative review comparing the yields of two candidate C-4 perennial biomass crops in relation to nitrogen, temperature and water. *Biomass Bioenergy* 27:21–30
- Heaton EA, Dohleman FG, Long SP. 2008. Meeting US biofuel goals with less land: the potential of *Miscanthus*. *Glob. Change Biol.* 14:2000–14
- Hennessy DA. 2006. On monoculture and the structure of crop rotations. *Am. J. Agric. Econ.* 88:900–14
- Hensen IE. 2003. The Malaysian national average oil palm: concept and evaluation. *Oil Palm Bull.* 46:15–27
- Hertel TW, Golub A, Jones AD, O'Hare M, Plevin RJ, Kammen DM. 2010. Global land use and greenhouse gas emissions impacts of U.S. maize ethanol: estimating market-mediated responses. *BioScience* 60:223–31
- Hess P, Johnston M, Brown-Steiner B, Holloway T, de Andrade JB, Artaxo P. 2009. Air quality issues associated with biofuel production and use. In *Biofuels: Environmental Consequences and Interactions with Changing Land Use. Proc. Sci. Comm. Probl. Environ. (SCOPE), Int. Biofuels Proj. Rapid Assess.*, 22–25 Sept. 2008, Gummersbach, Ger., ed. RW Howarth, S Bringezu, pp. 169–94. Ithaca, NY: Cornell Univ.
- Hewitt CN, MacKenzie AR, Di Carlo P, Di Marco CF, Dorsey JR, et al. 2009. Nitrogen management is essential to prevent tropical oil palm plantations from causing ground-level ozone pollution. *Proc. Natl. Acad. Sci. USA* 106:18447–51
- Hill J, Nelson E, Tilman D, Polasky S, Tiffany D. 2006. Environmental, economic, and energetic costs and benefits of biodiesel and ethanol biofuels. *Proc. Natl. Acad. Sci. USA* 103:11206–10
- Hill J, Polasky S, Nelson E, Tilman D, Huo H, et al. 2009. Climate change and health costs of air emissions from biofuels and gasoline. *Proc. Natl. Acad. Sci. USA* 106:2077–82

- Int. Energy Agency. 2009. *World Energy Outlook*. Paris, Fr.: IEA
- ISO. 2006a. *ISO 14040: Environmental management—Life cycle assessment—Principles and framework*. Rep. ISO 14040:2006(E). Geneva: Int. Stand. Organ.
- ISO. 2006b. *ISO 14044: Environmental management—Life cycle assessment—Requirements and guidelines*. Rep. ISO 14044:2006(E). Geneva: Int. Stand. Organ.
- Jacobson MZ. 2007. Effects of ethanol (E85) versus gasoline vehicles on cancer and mortality in the United States. *Environ. Sci. Technol.* 41:4150–57
- Keeney R, Hertel TW. 2009. The indirect land use impacts of United States biofuel policies: the importance of acreage, yield, and bilateral trade responses. *Am. J. Agric. Econ.* 91:895–909
- Kiesecker JM, Copeland H, Pocewicz A, McKenney B. 2010. Development by design: blending landscape-level planning with the mitigation hierarchy. *Front. Ecol. Environ.* 8(5):261–66
- Kim S, Dale B. 2002. Allocation procedure in ethanol production system from corn grain. *Int. J. Life Cycle Assess.* 7:237–43
- Klink CA, Machado RB. 2005. Conservation of the Brazilian Cerrado. *Conserv. Biol.* 19:707–13
- Koh LP. 2008. Can oil palm plantations be made more hospitable for forest butterflies and birds? *J. Appl. Ecol.* 45:1002–9
- Koh LP, Levang P, Ghazoul J. 2009. Designer landscapes for sustainable biofuels. *Trends Ecol. Evol.* 24:431–38
- Koh LP, Wilcove DS. 2008. Is oil palm agriculture really destroying tropical biodiversity? *Conserv. Lett.* 1:60–64
- Laclau PB, Laclau JP. 2009. Growth of the whole root system for a plant crop of sugarcane under rainfed and irrigated environments in Brazil. *Field Crops Res.* 114:351–60
- Lapola DM, Schaldach R, Alcamo J, Bondeau A, Koch J, et al. 2010. Indirect land-use changes can overcome carbon savings from biofuels in Brazil. *Proc. Natl. Acad. Sci. USA* 107:3388–93
- Lobell DB, Cassman KG, Field CB. 2009. Crop yield gaps: their importance, magnitudes, and causes. *Annu. Rev. Environ. Resour.* 34:179–204
- Lynd LR, Laser MS, Brandsby D, Dale BE, Davison B, et al. 2008. How biotech can transform biofuels. *Nat. Biotechnol.* 26:169–72
- Macedo IC, Seabra EA, Silva JEAR. 2008. Green house gases emissions in the production and use of ethanol from sugarcane in Brazil: the 2005/2006 averages and a prediction for 2020. *Biomass Bioenergy* 32:582–95
- Mack RN. 2008. Evaluating the credits and debits of a proposed biofuel species: giant reed (*Arundo donax*). *Weed Sci.* 56:883–88
- McCormick RL. 2007. The impact of biodiesel on pollutant emissions and public health. *Inhal. Toxicol.* 19:1033–39
- McDonald RI, Fargione J, Kiesecker J, Miller WM, Powell J. 2009. Energy sprawl or energy efficiency: climate policy impacts on natural habitat for the United States of America. *PLoS ONE* 4:e6802
- Melillo JM, Gurgel AC, Kicklighter DW, Reilly JM, Cronin TW, et al. 2009a. *Unintended Environmental Consequences of a Global Biofuels Program*. Rep. No. 168. Cambridge, MA: MIT
- Melillo JM, Reilly JM, Kicklighter DW, Gurgel AC, Cronin TW, et al. 2009b. Indirect emissions from biofuels: how important? *Science* 326:1397–99
- Moebius-Clune BN, van Es HM, Idowu OJ, Schindlebeck RR. 2008. Long-term effects of harvesting maize stover and tillage on soil quality. *Soil Sci. Soc. Am. J.* 72:960–69
- Mokany K, Raison RJ, Prokushkin AS. 2006. Critical analysis of root: shoot ratios in terrestrial biomes. *Glob. Change Biol.* 12:84–96
- Monti A, Fazio S, Venturi G. 2009. The discrepancy between plot and field yields: harvest and storage losses of switchgrass. *Biomass Bioenergy* 33:841–47
- Mubako S, Lant C. 2008. Water resource requirements for corn-based ethanol. *Water Resour. Res.* 44:W00A02
- Natl. Res. Counc. 2008. *Water Implications of Biofuels Production in the United States*. Washington, DC: Natl. Acad. Press
- Pfaff A, Robalino J, Walker R, Aldrich S, Caldas M, et al. 2007. Road investments, spatial spillovers, and deforestation in the Brazilian Amazon. *J. Reg. Sci.* 47:109–23
- Plevin RJ. 2009. Modeling corn ethanol and climate—a critical comparison of the BESS and GREET models. *J. Ind. Ecol.* 13:495–507

- Raghu S, Anderson RC, Daehler CC, Davis AS, Wiedenmann RN, et al. 2006. Adding biofuels to the invasive species fire? *Science* 313:1742
- Ramanathan V, Carmichael G. 2008. Global and regional climate changes due to black carbon. *Nat. Geosci.* 1:221-27
- Randall GW, Huggins DR, Russelle MP, Fuchs DJ, Nelson WW, Anderson JL. 1997. Nitrate losses through subsurface tile drainage in Conservation Reserve Program, alfalfa, and row crop systems. *J. Environ. Qual.* 26:1240-47
- Reijnders L, Huijbregts MAJ. 2008. Palm oil and the emission of carbon-based greenhouse gases. *J. Clean. Prod.* 16:477-82
- Ribeiro MC, Metzger JP, Martensen AC, Ponzoni FJ, Hirota MM. 2009. The Brazilian Atlantic Forest: How much is left, and how is the remaining forest distributed? Implications for conservation. *Biol. Conserv.* 142:1141-53
- Roberts MG, Male TD, Toombs TP. 2007. *Potential Impacts of Biofuels Expansion on Natural Resources; a Case Study of the Ogallala Aquifer Region*. New York: Environ. Defense
- Rudel TK, Defries R, Asner GP, Laurance WF. 2009. Changing drivers of deforestation and new opportunities for conservation. *Conserv. Biol.* 23:1396-405
- Schmer MR, Vogel KP, Mitchell RB, Perrin RK. 2008. Net energy of cellulosic ethanol from switchgrass. *Proc. Natl. Acad. Sci. USA* 105:464-69
- Searchinger T, Heimlich R, Houghton RA, Dong F, Elobeid A, et al. 2008. Use of U.S. croplands for biofuels increases greenhouse gases through emissions from land-use change. *Science* 319:1238-40
- Secchi S, Gassman PW, Williams JR, Babcock BA. 2009. Corn-based ethanol production and environmental quality: a case of Iowa and the conservation reserve program. *Environ. Manag.* 44:732-44
- Simmons BA, Keasling JD, Blanch H, Adams PD, Lane TW, Shaddix C. 2008. Letter to CARB Chairwoman Mary Nichols regarding the LCFS. Retrieved Sept. 20, 2009 from [http://www.bioenergywiki.net/images/7/7b/DOE\\_letter.pdf](http://www.bioenergywiki.net/images/7/7b/DOE_letter.pdf)
- Six J, Ogle SM, Breidt FJ, Conant RT, Mosier AR, Paustian K. 2004. The potential to mitigate global warming with no-tillage management is only realized when practised in the long-term. *Glob. Change Biol.* 10:155-60
- Stickler C, Coe M, Nepstad D, Fiske G, Lefebvre P. 2007. *Readiness for REDD: A Preliminary Global Assessment of Tropical Forested Land Suitability for Agriculture*. Rep. UN Framework Convent. Clim. Change (UNFCCC) Conf. Parties (COP), 13th Sess., 3-14 Dec., Bali, Indonesia, Woods Hole Res. Cent., Falmouth, MA
- Stoft SE. 2009. Renewable fuel and the global rebound effect. *Glob. Energy Policy Cent. Res. Pap. No. 10-06*. Available at [http://papers.ssrn.com/sol3/papers.cfm?abstract\\_id=1636911](http://papers.ssrn.com/sol3/papers.cfm?abstract_id=1636911)
- Tilman D, Hill J, Lehman C. 2006. Carbon-negative biofuels from low-input high-diversity grassland biomass. *Science* 314:1598-600
- Tilman D, Socolow R, Foley JA, Hill J, Larson E, et al. 2009. Beneficial biofuels—the food, energy, and environment trilemma. *Science* 325:270-71
- USDA. 2009. *Farm and Ranch Irrigation Survey (2008) AC-07-SS-1*. Washington, DC: USDA
- USEPA. 2009. *Draft Regulatory Impact Analysis: Changes to Renewable Fuel Standard Program*. Washington, DC: U.S. Environ. Prot. Agency
- USEPA. 2010. *Renewable Fuel Standard Program (RFS2) Regulatory Impact Analysis*. Washington, DC: U.S. Environ. Prot. Agency
- van der Velde M, Bouraoui F, Aloe A. 2009. Pan-European regional-scale modelling of water and N efficiencies of rapeseed cultivation for biodiesel production. *Glob. Change Biol.* 15:24-37
- Vyn TJ, West TD, Steinhardt GC. 2000. *Corn and soybean response to tillage and rotation systems on a dark prairie soil: 25 year review*. Proc. 15th Conf. Int. Soil Tillage Res. Organ., July 2-6, Forth Worth, Tex.
- Weidema B. 2000. Avoiding co-product allocation in life-cycle assessment. *J. Ind. Ecol.* 4:11-33
- Weidema B, Ekvall T. 2009. *Consequential LCA: Chapter for CALCAS deliverable D18, 2009 "Guidelines for applications of deepened and broadened LCA,"* (CALCAS) Co-ordination Action innovation in Life-Cycle Analy. Sustain. Available at [http://www.lca-net.com/files/consequential\\_LCA\\_CALCAS\\_final.pdf](http://www.lca-net.com/files/consequential_LCA_CALCAS_final.pdf)
- Wilcove DS, Rothstein D, Dubow J, Phillips A, Losos E. 1998. Quantifying threats to imperiled species in the United States. *BioScience* 48:607-15
- Wilhelm WW, Johnson JME, Karlen DL, Lightle DT. 2007. Corn stover to sustain soil organic carbon further constrains biomass supply. *Agron. J.* 99:1665-67

- Wise M, Calvin K, Thomson A, Clarke L, Bond-Lamberty B, et al. 2009. Implications of limiting CO<sub>2</sub> concentrations for land use and energy. *Science* 324:1183–86
- Worldwatch Inst. 2009. *Biofuel production up despite economic downturn*. <http://vitalsigns.worldwatch.org/vs-trend/biofuel-production-despite-economic-downturn>
- Wu M, Mintz M, Wang M, Arora S. 2009. Water consumption in the production of ethanol and petroleum gasoline. *Environ. Manag.* 44:981–97
- Yanowitz J, McCormick RL. 2009. Effect of E85 on tailpipe emissions from light-duty vehicles. *J. Air Waste Manag. Assoc.* 59:172–82
- Yusoff S, Hansen SB. 2007. Feasibility study of performing a life cycle assessment on crude palm oil production in Malaysia. *Int. J. Life Cycle Anal.* 1:50–58



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## Errata

An online log of corrections to *Annual Review of Ecology, Evolution, and Systematics*  
articles may be found at <http://ecolsys.annualreviews.org/errata.shtml>