Quantifying the Environmental Impacts of Biofuel Production: Knowns and Unknowns

Stephen Porder[a], Antonio Bento[b], Adrian Leip[c], Luiz A. Martinelli[d], Jon Samseth[e], Thomas Simpson[f]

[a]Department of Ecology and Evolutionary Biology, Brown University, Providence, RI, USA,[b]Department of Applied Economics and Management, Cornell Univeristy, Ithaca, NY, USA, [c]European Commission – DG Joint Research Centre, Institute for Environment and Sustainability, Ispra (VA), Italy, [d]Department of Isotope Ecology, CENA, Piracicaba, SP, Brazil, [e]Norwegian University of Science and Technology, Trondheim, Norway, SINTEF materials and technology Trondheim, Norway, [f]Water Stewardship Inc., Annapolis, MD, USA

In response to calls for energy security, climate change mitigation, and rural development, several governments (U.S., European Union, Brazil) have or will shortly establish mandatory targets for the incorporation of biofuels (defined here as fuels derived from plants or biological waste) into their liquid fuel portfolio (Chapter 2, Searchinger 2009). A number of recent papers, however, have pointed out that there are hard biophysical constraints on production - the amount of carbon fixed by all crops globally is already exceeded by the carbon released by fossil fuel combustion - and producing biofuels on all currently abandoned land would meet only ~7% of current energy demand (Campbell et al. 2008; Field et al. 2008). Additionally, legitimate concerns exist about the relative climate benefit of various biofuels (Crutzen et al. 2008) and competition for arable land between food, fiber, fuel and other ecosystem services (Zah et al. 2007; Searchinger et al. 2008, Melillo et al. 2009).

Available technology for generation of electricity and heat from biomaterial is considerably more efficient than using that material for liquid fuel (Edwards et al. 2008; Chapter 5, Menichetti and Otto 2009). Nevertheless, a combination of tariffs, government mandates and complex tax structures suggest that liquid biofuel use will continue to grow over the coming decade in both the developed (Chapter 2, Searchinger 2009) and developing world (Chapter 15, Bekunda et al. 2009).

We set two goals for this chapter: to review the available information on the environmental impacts of a few important biofuel feedstocks (corn and sugarcane ethanol, rapeseed biodiesel) and then to highlight major gaps in our knowledge that need to be addressed before a truly quantitative assessment of these feedstocks can be made. The chapter focuses mostly on global and regional energy supplies and the environmental costs of biofuel production

with case studies provided for Brazil, the European Union, and the U.S. While we recognize that the adoption of biofuels may have real economic benefits, particularly in developing countries (discussed in Chapter 15, Bekunda et al. 2009), environmental concerns apply to all countries, and the sustainable production of biofuels will require a full accounting of the environmental and social ramifications of biofuel production. We begin by briefly reviewing the limits of life cycle analysis (LCA), as well as literature results from traditional and expanded LCA for a few of the current major biofuel crops. We then present preliminary results from recent research which aims for a fuller accounting of impacts from large increases in biofuel production by accounting for effects such as indirect land use change (e.g. Fargione et al. 2008; Searchinger et al. 2008) and transportation adjustments due price shifts in fuel (Bento and Landry 2008).

Traditional and expanded lifecycle analysis.

LCA is a typical starting point in the discussion of whether or not to promote a particular biofuel, but such analyses may be insufficient for several reasons including:

- Data are typically not available to support several of the environmental impacts that should be included in LCA

- LCA is typically focused on greenhouse gas (GHG) emissions, but less frequently involves assessment of other ecological costs such as biodiversity loss, other emissions to air, water quality and quantity shifts, soil maintenance, particularly for feedstocks that are likely to become important in the coming decade but which do not yet produce large quantities of biofuels

- LCA is often based on the costs (both environmental and economic) of producing the first litre of fuel, and the costs relative to fossil fuels may change dramatically, in both magnitude and direction, with changes in the scale of production.

In light of these uncertainties, we suggest caution in moving forward with aggressive biofuel production targets, especially given the fundamental production constraints which suggest biofuels will never supply more than a modest percentage of global energy needs (Field et al. 2008).

Traditional lifecycle analysis. Traditional LCA is explored in greater detail elsewhere in this volume (Chapter 5, Menichettiand Otto 2009), so we will review it only briefly here. Because biofuels are often promoted in the context of GHG and climate mitigation, a major focus of LCA of biofuel production have been the related GHG emissions (CO_2, N_2O, CH_4). In order to complete an analysis quantitatively, the following three aspects of the field-to-wheel system need to be explored:

- cultivation (or collection) of the biomass feedstock – seed production, planting, agrochemicals (especially fertilizer), fossil fuels for equipment

- processing and conversion of the feedstocks - this includes both electricity and process heat, the chemicals needed for synthesis and waste processing, and details on the nature and fate of co-products[1]

- Distribution, i.e. transport needs in terms of freight haulage using ships (barge or tanker), rail, and trucks (with most of the transport fuel being diesel, marine bunker in the case of ocean-

going vessels). These are then com-pared with the emissions of an energy equivalent (or kilometer driven equivalent) of the fuel being replaced (gasoline or diesel)

In general, it is believed that agricultural production is responsible for a substantial share of GHG emissions and a majority of water quality degradation. Impacts of water quantity vary widely by feedstock (Chapter 8, de Fraiture and Berndes 2009). The impacts of energy use are significant in the conversion phase, particularly in the case of ethanol production. The quantity and type of process energy used (e.g. heat and power from coal, natural gas or bagasse) can change overall assessments dramatically. Furthermore, the allocation of impacts on co-products can also be very significant in this phase of the life cycle.

Traditional LCA has utility in that it presents a number than can be readily used by policy makers in decision making; however, we wish to highlight several concerns about LCA that need to be considered when interpreting these data. First, as measured by the number of peer-reviewed articles on LCA of biofuels in recent years and the disparity of results presented in the liter-ature, it is fair to say that there is a great deal of uncertainty associated with this methodology (Pimental and Patzek 2005; Farrell et al 2006; Chapter 5, Menechetti and Otto 2009). Part of this lies in the choice of system boundaries. Depending on the definition of those boundaries, different studies find that the net effect of biofuel production can range from positive to nega-tive. In fairness, system boundary problems are not unique to LCA; any regional analysis must treat the rest of the world as exogenous to some degree. A second

concern is the treatment of co-product credits. To under-stand the importance of this issue, note that some of the co-products of ethanol can displace feed products for livestock and, in turn, such displacements saves energy that otherwise would have been consumed. The point of contention lies in the details –how this displacement effect is measured. Currently, different authors use different ways of accounting for co-products. Studies can be broadly grouped based on the accounting method employed, i.e. process-based credit, market-based credit, or displacement-based credit accounting. Further, it is important to stress that even at the plot level, it is rare that we have sufficient data for a calculation of impacts across a broad suite of biophysical variables, especially when production systems are moved into novel landscapes. Even for GHG emissions the data are regrettably sparse. Based on a meta-analysis of Guo and Gifford (2002), Searchinger et al. (2008) assumes that conversion of forest to biofuel crop results in a 25% loss of soil carbon in the top meter of soil. This is a reasonable average, but one that may vary dramatically among sites, biofuel crops, and production practices, and does not account for turnover of deeper carbon that can augment or offset differences in the upper soil (Veldkamp et al. 2003)

Expanded lifecycle assessment. If there is uncertainty associated with GHG account-ing, the uncertainty grows when considering changes in water quality, quantity, and soil maintenance. We synthesize a relatively broad suite of environmental impacts of biofuel production for a few feedstocks that are commonly used today or may become common in the near future. We assign a positive value where use of a liter of fossil fuel equivalent biofuel has a net benefit (at the plot scale) over a liter of gasoline (or

Table 14.1 A qualitative assessment of some of the major biofuel feedstocks, based on comparison to an energy equivalent amount of gasoline. ++ indicates a strong improvement, + and improvement, 0 neutral, - deleterious effect, and – strongly deleterious effect. These values are assigned based on the authors review of the literature and expertise with particular feedstocks, and thus represent our view but not necessarily a quantifiable one. More quantitative estimates are derived from the reference below.

	Corn U.S.A.	Corn stover U.S.A.	Switchgrass U.S.A.	Sugarcane Brazil
Fuel		ethanol		
Energy(GJ ha^{-1})*	55-77[1]	17-23[2]	n/a	140[4]
CO_2	0/ +	+	++	++
N_2O	--	unk	unk	unk
Air quality	- / +	unk	unk	--
Water quantity (L GJ^{-1})	160-230[8]	360-520[8]	~4:1 #	340[8]
Water quality	--	--	++	-
Biodiversity	--	unk	++	-
Soil qual/ conservation	-	--	++	--
Indirect LUC	--	--	-	0/-
	Rapeseed Europe	Soy U.S.A.	Palm Oil Indonesia	Jatropha** India
Fuel		biodiesel		
Energy(GJ ha^{-1})*	35[3]-46[5]	18[1]	112-160[3]	41[6]
CO_2	+	0/+	--	+
N_2O	--	-	unk	unk
Air quality	unk	unk	unk	unk
Water quantity (L GJ^{-1})†	70-90[8]	180-230[8]	20-37[8]	100-240[8]
Water quality	-	--	--	unk
Biodiversity	unk	--	--	unk
Soil qual/ conservation	+	-	--	unk
Indirect LUC	--	--	-	unk

*   Assuming 21 MJ litre^{-1} for ethanol and 33 MJ litre^{-1} for biodiesel (Hill et al. 2006).

**   yields on degraded land are likely to be substantially smaller (Jongschaap et al. 2007).

†   does not include irrigation, only consumption at the plant. Water that is not consumed is not counted in this row, because it affects quality but not quantity.

\#   ratio of water: ethanol

unk indicates that we could find 0 or 1 study addressing the effect in question.

[1]   Patzek 2004, and references therein; Hill et al. 2006

[2]   assumed to be 30% of the energy produced by harvesting the corn rather than the stover.

[3]   Conner and Hernández, this volume.

[4]   Macedo et al. 2008b, assuming 21MJ litre^{-1} ethanol.

[5]   Thamsiriroj and Murphy (in press) estimates for rapeseed production in Ireland.

[6]   Prueksakorn and Gheewala 2006 for Jatropha in irrigated test plots in Thailand

[7]   Keeney and Muller 2006

[8]   Based on current small-scale production, 9:1 water to ethanol, (Pate et al. 2007)

diesel), and a negative value in the opposite case. It is important to point out, however, that the analyses presented below are subject to several of the major criticism of LCA. First, this approach explores only the costs of direct effects, and not of indirect changes in land use or economic behavior. This criticism pertains mostly to the GHG calculations, and is explored later in this chapter. Second, our assumptions about water quality changes do not take into account water pollution issues associated with transportation or storage of either the biofuel or the gasoline (or diesel) it replaces. Thus, while we can document changes in water quality and quantity associated with biofuel production, it is more difficult to quantify the net impact on water quality attained by shifting to biofuels from petroleum based liquid fuels. Petroleum production must have an impact on both water quality and quantity, but, to our knowledge, all discussion of water pollution that results from biofuel production gives no offset for pollution reduction from avoidance of petroleum-based fuels (assuming the biofuel is energy positive, i.e. produces more useable energy than is required in fossil fuel inputs to drive production processes).

The results of our analysis are summarized in Table 14.1. However, a major point of this chapter is to urge caution in relying on summary tables such as these since the uncertainties are almost always large and difficult to track down. Furthermore, it is not clear how relevant results from LCA are to large-scale biofuel production, where many of the major effects may result from changes in behavior by economic agents. To illustrate this point, we walk through a more detailed assessment of a few biofuel feed-stocks for which we have particularly good data.

Moving beyond LCA – a fuller accounting of biofuel impacts

The case studies provided within this chapter (boxes 14.1, 14,2, 14.3) illustrate the difficulty of condensing multiple environmental costs into a single measure, and highlight the complex set of environmental feedbacks that are associated with each form of production. Beyond these difficulties, it is worth repeating that a major criticism of LCA-type approaches is that they do not include indirect effects associated with the scaling up of production of particular product (Farrell et al. 2006). Despite an increasing recognition that in-direct effects may be as or more important from a GHG perspective than on site factors, LCA remains a common tool used by policy-makers and researchers to evaluate the potential environmental savings from biofuels.

In this section, we present preliminary results based on new methods that blend general equilibrium economic models with LCA to evaluate the effects of large increases in biofuel production. We suggest that such approaches need to be refined, but do represent a substantial improvement over LCA by providing a more realistic assessment of net effects at the global scale and highlighting the potential for unintended positive and negative feedbacks that may affect both the magnitude and direction of GHG responses relative to expectations based on traditional LCA.

LCA sets out to determine the percent energy savings (or other environmental indicator) of replacing fossil gasoline or diesel with a particular biofuel alternative. A key problem arises, however, because the relationship between energy savings and the level of fuel displacement is not linear. In fact, while it is reasonable to assume that

box 14.1

Case Study - An Environmental Assessment of Ethanol Derived from Brazilian Sugar Cane

Air quality and CO$_2$ effects. Brazilian ethanol production has the highest energy return to fossil fuel energy invested of any of the current biofuel crops (~9:1, Macedo et al. 2008). Sugarcane to ethanol has a relatively high energy/land use, roughly 140 GJ of biofuel energy can be produced per hectare. Furthermore, Brazilian ethanol from sugarcane emits roughly 70% less CO$_2$ than the gasoline it replaces (Menichetti and Otto this volume). According to Macedo et al. (2008) the net avoided CO$_2$ emissions is approximately 1,800 kg CO$_2$eq m^{-3} ethanol. A simple extrapolation of Brazil's 2008 production (27 million m^3) suggests avoided emissions of 0.05 Pg CO$_2$ for the country (slightly less than 0.5% of total anthropogenic emissions in 2007; IPCC 2007). This number does not include any land use change and carbon loss associated with displacement of farming and/or grazing into carbon rich forested regions. The case for the other GHGs is less well established. There is only study that we are aware of that directly measure N-N$_2$O emission from sugarcane soils (Campos 2003). This study found emissions of approximately 1 to 2 kg N-N$_2$O ha^{-1}yr^{-1}, which is equivalent to 1.5 to 2.5% of the fertilizer that it is averaged annually used in sugarcane (60 to 80 kg N ha^{-1}yr^{-1}). (Macedo et al. 2008) used an emission factor of 1.4% of the fertilizer N loss as N$_2$O and estimated an emission of 6 kg CO$_2$ eq kg^{-1} of N fertilizer used. A crude extrapolation to 9 million hectares (the entire production area for Brazilian sugarcane in 2008) would yield emissions ~5x10^9 mols N$_2$O. Given a global warming potential of ~300 x CO$_2$ (Prather et al. 2001) this could significantly offsets the CO$_2$ savings of ethanol, especially if a higher (overall) N$_2$O emission rate rate of 3-5% is used (Crutzen et al. 2008). However it is unclear how these emissions compare with background emissions, since intact Brazilian ecosystems can also emit large, though poorly constrained, amounts of N$_2$O (Melillo et al. 2001).

Sugarcane production also negatively impacts local air quality, mainly because of the annual burning that occurs during the harvest (March - December). The concentration of aerosol particles peaks during the harvesting season, reaches values of more than 200 mg m^{-3}, which is higher than the 24 hours-standard established by the Brazilian law (Martinelli and Filoso 2008). The same is true for the annual average, that in 2004 in the region of the Piracicaba municipality in the State of São Paulo was equal to 70 mg m^{-3}, approximately 20 mg m^{-3} above the maximum annual average value (Martinelli and Filoso 2008). Additionally, as a significant share (30%) of the applied nitrogen is lost via volatilization and 10% more is lost via pyro-volatilization during the sugarcane burning, the wet nitrogen deposition measured in the same municipality was significantly higher than the deposition observed in more pristine areas of the State of São Paulo and Amazônia (Lara et al. 2005). Part of the nitrogen in the atmosphere is transformed in nitric acid which is scavenging by the precipitation, as a consequence the annual weight average pH of the rainfall measured in four municipalities of the Piracicaba River basin were consistently lower than 5.6 (Lara et al. 2001).

Water use and soil erosion. sugarcane is a water intensive crop. According to Elia Neto (2008), in 1990, the average sugarcane mill used 5.6 m^3 per ton of sugarcane; this improved to 1.8 m^3 of water per ton cane in 2004, of which 0.6 m^3 is consumed (Table 14.1). Perhaps more problematic than water consumption, sugarcane production as currently practiced has several detrimental effects on water quality. The main effluent of the sugarcane industry is vinasse. Vinasse is a nutrient and organic-matter rich effluent (17 kg BOD m^{-3}; BOD - biological oxygen demand) and its production is 10-13 times that of ethanol by volume (Bertoncini 2008). Most vinasse is re-applied to the field as a fertilizer. Approximately 300 m^3 of vinasse is applied per hectare (Luz 2008). Using the average composition of vinasse provided by Elia Neto, we estimated an input of 90 kg N, 15kg P and 500 kg K ha^{-1} yr^{-1}. These high K applications have generated concerns of K contamination of groundwater. Additional concerns include vinasse spills from storage tanks and pipes, which have been subject to penalties from the State of São Paulo Environmental Agency (Cetesb; survey made in regional offices of Cetesb in the following municipalities Piracicaba, Americana, Santa

Barbara d'Oeste, Pirassununga, Barretos, and Ribeirão Preto covering a period between 2003 and 2008). It is worth noting, however, that the transport of gasoline and diesel products is not without spillage concerns, and to the extent that ethanol is replacing these products in the landscape, it is unclear how to assess the net threat or benefit of vinasse vs. petroleum production.

One of the greatest hopes for biofuel production is that it can provide a sustainable source of energy for many decades. At the heart of crop sustain-ability is the preservation of soil. While the loss of nutrients can be supplemented with fertilizer, the erosion of organic matter rich, base cation rich topsoil is often almost irreversible on human time scales (Sanchez 2002). Intensive cane cultivation requires tillage, planting, application of fertilizer and agrochemicals, annual burning and harvesting. All these operations are carried out by heavy equipment, which compact the soil, decrease pore space and water infiltration, and lead to increased water runoff (Cerri et al. 1991; de Oliveira et al. 1995; Silva and Ribeiro 1997; Silva et al. 1998; Ceddia et al. 1999; Prado and Centurion 2001) This has resulted in significant soil erosion losses in some cases, especially during the periods that soils are not cover with sugarcane which includes the planting, harvesting and re-growth periods (Sparovek and Schnug 2001; Politano and Pissarra 2005). The annual sugarcane burning also in-creases the chances of soil erosion by decreasing soil water content that leads to compaction and higher surface water runoff (Dourado-Neto et al. 1999; de Oliveira et al. 2000; Tominaga et al. 2002). Soil pollution by several compounds like poly-cyclic aromatic hydro-carbons (PAH) caused by sugarcane burning (Pereira-Netto et al. 2004), and heavy metals (Carvalho et al. 1999; Azevedo et al. 2004; Corbi et al. 2006) are also a concern for the long term sustainability of the soil.

Land use impact. The three most important indirect land effects of sugarcane expansion in Brazil is the potential conversion of natural vegetation like Atlantic Forest, Cerrado, Pantanal and Amazon Forest into sugarcane fields. For the Amazon Forest and the Pantanal there is a law that prohibits large scale cultivation of sugarcane, though enforcement is problematic. There is a sense that it is not economically viable to convert Cerrado vegetation into sugarcane (Isaias Macedo, personal communication during the First Work-shop ESSP on Bioenergy and Earth Sustainability, Piracicaba 19-21 July 2008), and it is more profitable to convert pasture or soybean fields into sugarcane. In any case, it is important to note that a survey made by CONAB (2008) showed that between 2006 and 2007 almost 8,000 hectares of sugarcane were planted under the category of "new areas". The report did not explain what the "new areas" are, but since the same report showed the replacement of the main type of crops (corn, soybean, citrus, coffee and pastures) by sugarcane, and also had a category under the name of "others", it is reasonably to suppose that these are areas covered with primary vegetation. Our speculation is that this conversion occurred in the remnants of Atlantic Forest or Cerrado in the State of São Paulo.

Another potential indirect effect of sugarcane expansion results from the relocation of pasture and soy bean in the Cerrado located in the Central Brazil area. If those uses are pushed further into the forest by expanding sugarcane, the carbon gains of sugarcane-ethanol would be eliminated and the system would be a large net source of CO_2 to the atmosphere (Searchinger et al. 2008). However the displacement of pasture and soy by sugarcane, and the chain of causation to deforestation, is highly speculative and hotly debated at this time.

Finally, the third indirect effect of sugarcane expansion is the conversion of food crops into sugarcane for ethanol. According to CONAB (2008) almost 65% of the sugarcane expansion in Brazil occurred on pasture, 15% replaced soybeans, 5% replaced corn and 5% replaced citrus. It has been argued that because Brazilian pastures can sustain 1.2 – 1.4 head of cattle ha^{-1}, and are currently stocked at ~1 head ha^{-1}, pasture can be converted to sugarcane without forcing an in-crease in pastureland (Goldemberg et al. 2008). We agree that the interplay between sugarcane, soy, and pasture are likely to play a critical role in determining the net benefits and costs of the sugarcane-ethanol systems as it expands, but argue there is insufficient data at this time to make a more quantitative assessment of the indirect land use impacts of sugarcane expansion.

the 'overall' economy remains constant for the first few units of biofuels production, the same is not true as policies start to require rather large mandates on biofuels production. As soon as other prices in the economy start to adjust, LCA methods are no longer valid simply because LCA methods do not have the ability to account for the indirect effects that occur due to behavioral adjustments. These adjustments happen whenever prices change.

Very recent literature (e.g. Searchinger et al. 2008; Melillo et al. 2009) has been instrumental in bringing public recognition to the issue of indirect land use change and has begun to account for the potential global indirect effects of biofuels. Indeed, Searchinger et al. (2008) clearly illustrate that an increase in crop production in the U.S. could lead to deforestation of the Amazon or other extensive land use changes. These effects occur because the relative values of alternative land uses change with the increased demand for biofuels. These effects are indirect and independent of the spatial location of biofuels. They arise due to market forces and competition for land for different uses. In a sense, one could say that the major problem of previous biofuels analyses lies I n the underlying assumption that land doesn't have an opportunity cost, and therefore, it is treated as if it is free! A major strength of the recent papers on indirect land-use change lies in pointing out that, other than for the very first unit of biofuels allocated to a parcel of land, land is not free and indeed has an opportunity cost. The opportunity cost is the foregone production of the previous land use, which will be made up for somewhere else.

While Searchinger et al. (2008), Melillo et al (2009), and others have highlighted the

importance of indirect effects, we suggest that these effects need to be explored in greater detail in order to develop rational biofuels policy. It is not clear, a priori, which indirect effect (changes in land use, fuel prices, or technological innovation) is likely to have the greatest effect on the net CO_2 cost or savings of biofuel production. Thus, there is a need to develop more comprehensive general equilibrium economic models that incorporate the behavior of a variety of key agents affected by increased biofuels production and not simply land-owners. There is also a need to incorporate different aspects of dynamics, including induced and exogeneous technological progress (both in the ethanol industry, in transportation and in agriculture) as well as other relevant macroeconomic aspects.

In ongoing research, Bento and Landry (2008) outline a general equilibrium model that links the behavior of key agents affected by biofuels policies including agricultural producers, other land-owners, ethanol producers and fuel blenders, as well as the behavior of consumers – who allocate their income for food, transportation services (including alternative types of vehicles and miles driven) and other relevant goods. When simulated for the U.S. economy, preliminary results suggest that there are at least three important levels of indirect impacts resulting from increased biofuels production, as outlined below.

First, similar to Searchinger et al. (2008) and Melillo et al. (2009), Bento and Landry (2008) show an extensive land use effect that results from the increased value of cropland. How-ever, Bento and Landry (2008) document other relevant indirect land use effects that are associated with the adjustment of agricultural producers at the

intensive margin. For example, in response to the increase in the corn-based ethanol mandate, agricultural producers increase the acreage of continuous corn rotation and convert land previously in the USDA conversation reserve program (CRP). To the extent that they affect the environmental indicators of biofuel performance, these indirect land use effects of increased biofuel production need to be carefully measured and accounted for (Searchinger et al. 2008; Melillo et al. 2009). However, so do other factors that have yet to be addressed. For example, increased yields as projected by the USDA could result in production of 15 billion gallons of ethanol (1.2 EJ) without reducing corn exports (Bento and Landry 2008). It should be noted, however, this is a very small portion of even today's transportation energy use (145 EJ; EIA 2008)

Bento and Landry (2008) also consider the policy sources that lead to the expansion of the biofuels sector in first place. In their general equilibrium model, there is a pre-existing ethanol tax credit of $0.13 USD liter^{-1} and an annual increase in the federal mandates for corn-based ethanol. The incorporation of biofuels policies into the general equilibrium model is extremely important as it affects the overall price of miles. Consequently, the adjustment of transportation consumer decisions – so far not discussed in the literature - may present another important indirect effect of increased biofuels production. Preliminary simulations suggest that, if the biofuels expansion is primarily driven by the pre-existing ethanol tax credit, the overall price of miles after the biofuel policy is in place is reduced. Theref-ore, a potential perverse effect of biofuels policies is the increase in overall vehicle miles traveled, a delay in the adoption of cleaner and more fuel efficient vehicles, and, consequently, increased

overall GHG emissions. On the other hand, if mandates for ethanol production bind – that is if the quantity required by law is greater than that the market would supply, then it is likely that the price of miles will actually increase and, irrespective of the feedstock used in the production of biofuels, there is a potential for GHG emissions savings from reductions in vehicle miles traveled and increased fuel economy.

Bento and Landry (2008) offer just one example, there are clearly many more, but the important point is that there are multiple indirect effects of increased biofuels production – not just indirect land uses at the extensive margin- and research-ers are only starting to unfold those effects and measure their environmental impli-cations. An important venue for future research is precisely to compare the magnitudes of the different potential indirect effects for different environmental indicators. Given the roughly equal GHG emissions from transportation and deforest-ation (IPCC 2007) it is unclear how models that consider one source and not the other will arrive at defensible conclusions.

Conclusion

As the global demand for energy grows, and the deleterious effects of our current sources become increasingly apparent, the push for the use of liquid biofuels continues to increase. We believe this is a case in which policy shifts have outstripped the science. While we acknowledge and applaud the urgency with which various nations have begun to search for alternative energy sources, we suggest that commitments to large, long-term biofuel production would be premature if GHG mitigation, or other environmental concerns, are the motivating factor behind such a policy choice. If

box 14.2

Case Study - An Environmental Assessment of Corn Ethanol Produced in the United States

In 1990, the United States produced about four billion liters of ethanol annually. By 2002, ethanol production had doubled to approximately eight billion liters per year. Between 2002 and 2008, an exponential increase in U.S. ethanol production occurred resulting in the capacity to produce about 53 billion liters (~ 1 EJ) per year by 2009 when all plants under construction are completed (RFA, 2008). At present, more than 95% of the ethanol produced in the United States is made from corn, so we focus on the environ-mental impacts of corn (rather than cellulosic) ethanol here. The exponential growth in ethanol production increased the demand for corn as feed stock from less than five percent of US domestic corn production in 2002 to about 30% of a much higher production level in 2008 (Elobeid et al. 2007, IATP 2007).

The exponential growth of the ethanol industry in the US was fueled by several converging circumstances (Keeney 2009). The US government created a blenders credit of 0.13 USD per liter and an import tariff of 0.14 USD per liter. In addition, concerns about groundwater contamination from methyl tertiary butyl ether (MTBE), a gasoline oxygenating agent, led the petroleum industry to look to ethanol as a replacement oxygenate. These two factors in combination with a surplus supply of inexpensive corn led to the initial boom in the ethanol industry between 2002 and 2005. When California banned the use of MTBE as an oxygenating gasoline starting in 2005, it created an instant demand for a large quantity of ethanol. As a result, hundreds of ethanol plants were planned primarily in the mid-western corn belt of the US. Many of these plants came online during 2006 and 2007 creating a large increase in demand for corn for use as ethanol. (RFA 2008).

Indirect effects. The rapid increase in demand created a major shortage of domestic corn for use as animal feed in the US and forced a reduction in grain exports to meet feed demands - a point that supports the concern that indirect land-use change elsewhere will occur when U.S. food crops are diverted for use as an energy sources (Searchinger et al. 2008). As the price of corn rose to historically high levels, US farmers planted seven million more hectares in 2007 than in 2006. Most of this new corn land came from displacement of annual crops such as cotton or soybeans along with some displacement of hay pasture and conservation reserve program lands. In 2008, this pattern changed with continued reductions in cotton acreage but increases in soybeans acreage so that corn plus soybean acreage (a typical two year, two crop rotation) was nearly five million hectares in 2008 than it has been in 2006. While some of this came from the displacement of cotton in the southern US, much of it came from the con-version of perennial hay, pasture or idle lands into row crops (USDA-NASS 2008).

Atmospheric effects. The rapid growth in the ethanol industry in the US and its singular reliance on corn grain as a feed stock has had numerous environmental and social impacts. First, unlike the energy return of Brazilian sugar-based ethanol, corn ethanol barely reduces the use of fossil fuels because corn production is it-self energy intensive (Hill et al. 2006, Keeney 2009). Thus as a GHG mitigator, corn ethanol is not a particularly useful biofuel, especially because corn agriculture, at least as is currently practiced in the US, is also a substantial N_2O source (Smeets et al. 2009). Those studies that do ascribe a GHG benefit to corn ethanol do so without considering indirect land use change - when such changes are considered, corn ethanol seems even less useful as a GHG mitigator (Searchinger et al. 2008). For example, the current price spike in corn resulted in the conversion of substantial amounts of in the Conservation Reserve Program lands (CRP) or left as pastures, hay or fallow to row crops. This may have resulted in a substantial carbon debt (Searchinger et al.

2008), though soil carbon losses are widely variable and difficult to con-strain. Estimates are that currently one to two million hectares of CRP land was converted with the potential of four to ten million hectares in total. The heavy fertilization of corn also in-creases N2O emissions, which are often higher from corn than other crops (Parkin and Kaspar 2006).

Water Impacts. The other primary impact of the rapid expansion of biofuels in the US has been on water quality and quantity. It takes three to six liters of water for each liter of ethanol produced so the production of 53 billion liters during 2009 would require more than 200 billion liters of water. While water scarcity is impacting the siting and/or operation of some ethanol facilities, the process water is not the major source of water consumption. The high price for corn and the need for greater confidence in yield are putting pressure on US farmers to increase irrigation of corn, which has historically been primarily grown without irrigation. It typically takes about 3 million liters per hectare of water to irrigate corn in a humid to sub-humid climate where most corn is produced in the US. Thus it would take about 3 trillion liters of water to if irrigated corn increased by one million hectares which is viewed as likely if demand continues.

Water quality is discussed in detail in chapter 8 (Simpson et al 2009); we briefly summarize the major effects here. The primary water quality concerns from corn-based ethanol are increased N and P losses to water from corn production and the feeding of ethanol production by-products known as Dried Distiller's Grains and Solubles (DDGS). Corn production has been a major contributor to pollution of surface waters of the US for many decades. Major efforts are underway in many parts of the US to reduce nutrient pollution from agriculture to help reduce hypoxic zones in coastal waters, most notably the Gulf of Mexico and the Chesapeake Bay. The conversion of about seven million hectares of perennial grasses and other crops to corn in 2007 is estimated to have increased nitrogen losses to surface waters by about 120 million kilograms (Simpson et al. this volume). In addition it is thought, though not well documented, that nitrogen applications and thus losses from the pre-existing 35 million hectares of crop land increased as farmers sought to assure adequate nitrogen for maximum potential yield. Nitrogen discharges to the Gulf of Mexico from the Mississippi River were at historically high levels in 2007 and then increased in 2008 probably due to the severe flooding in the Mississippi River basin that occurred with the expanded corn acreage (personal communication, Rabalais, USGS 2008).

The feeding of DDGS primarily to beef and dairy animals, is increasing the P content, and to a lesser extent the N content, of their manures (Erickson et al. 2002). This comes while major efforts are underway to reduce manure N and P content through feed management. DDGS contain relatively high amounts of phosphorus and nitrogen and, when included at 15-35 percent of ration can substantially increase N and P in manure, which in turn can have deleterious downstream consequences.

The future of the renewable fuel industry in the United States is unclear at the beginning of 2009. The global economic crisis and low crude oil prices have halted the boom in the corn-based ethanol industry and cellulosic ethanol remains elusive. There is growing thought that the US should focus on biomass production for heat, energy, and electricity and reduce its emphasis on renewable transportation fuels. Interest remains in expansion of other renewable energy sources, such as wind and solar, to provide electricity, and to then use electrically assisted vehicles to reduce dependence on fossil fuels. The United States is committed to reduce its dependence on fossil fuels in transportation and other uses both to reduce its energy dependence in an increasing volatile world and to reduce its emissions of greenhouse gasses. While that commitment remains clear, the path to the future is yet to be determined.

box 14.3

Case Study - An Environmental Assessment of Biodiesel from European Rapeseed

Fuel-grade biodiesel is easiest to make from rapeseed (from *Brassica campestris*), which grows well in Europe. On a per hectare basis, rapeseed yields about 1/3 the liquid fuel energy of Brazilian sugarcane, and emits roughly 50% less CO_2 than the diesel it replaces (Chapter 5, Menichetti and Otto 2009). As has been discussed elsewhere in the volume, the calculated fossil energy and GHG savings of conventionally produced biofuels are critically dependent on manufacturing processes and the fate of co-products. Edwards et al. (2007) give a range of net fossil energy consumed per energy produced from 0.36 to 0.51 MJ . This compares well to the values used in the study by Zah et al. (2007) ranging from 0.4-0.6 MJ per MJ of energy output. Based on the data presented in (Edwards et al. 2007) and a coupled agro-economic/ biogeochemistry model (see Leip et al. 2008) we postulate 44 GJ ha^{-1} yr^{-1} of saved fossil fuel use if they included a (likely optimistic) credit for the use of co-products of 3 GJ ha^{-1} y^{-1}. Calculations show that the overall energy benefit is potentially more than offset by on-farm N2O emissions. The high emission rates for N_2O are in contrast to the values often used in LCA for rapeseed production, and highlight the large uncertainty that soil-N_2O emissions introduce to the overall GHG pressure of rape seed biodiesel.

Rapeseed cultivation is not suited for water stressed regions, as the water requirements are medium to high and success depends greatly on the water supply of a minimum of 600 mm yearly precipitation. Nevertheless, most rapeseed field are rainfed (AEA Technology 2008). Rapeseed is also sensitive to high temperatures and grows best between 15 and 20°C (AEA Technology 2008). Although rapeseed cultivation requires intensive nitrogen inputs (130 kg N/ha/y mineral fertilizer and 50 kg N ha^{-1} y^{-1} manure N; (Leip et al. 2008), the deep and dense rooting system of rape may help reduce N leaching losses relative to other biofuel crops. In addition, these deep roots reduce the risk of soil erosion and soil compaction, especially because rapeseed is a winter crop. Nevertheless, the cascade of problems associated with N losses (Galloway et al. 2003) are of concern if rapeseed production for biofuels expands dramatically. While technological progress could lead to future rapeseed cultivars with a broader C/N ratio, increasing the energy yield per hectare and reducing nitrogen requirements under current practices the environmental burden with nitrogen must be seen critically for their effects on water and habitat quality.

As EU rapeseed oil is being diverted from the food market, it is replaced by imported oilseeds and oils, particularly the cheaper palm oil from Indonesia (Edwards et al. 2008). On the other hand, rape meal and dried distillers grains and solubles (DDGS) have high protein content and are suitable for displacing animal fodder. Using DDGS in this way potentially allows for fewer crops to be grown specifically for animal feed, particularly protein rich sources such as soy (Gallagher 2008). The overall evaluation of the environmental performance will depend to a high degree on the trade-off between these indirect land-use effects.

Within Europe, rapeseed production expanded dramatically during the last decade, largely on subsidized set-aside land, on which production of non-food crops was allowed. Set-aside subsidies are abolished now and further expansion will mainly replace grassland areas (Edwards et al. 2007). A rough estimate by (Edwards et al. 2007) gives a carbon loss on these lands of 73 MgC ha^{-1}, which is close difference of equilibrium carbon stocks of grassland and cropland of the IPCC (IPCC 1997).

biofuels should play a role, it may well be wiser to use them for heat and electricity generation, which is a far more efficient process at present than conversion to liquid energy. Due to thermodynamics, internal combustion engines (ICE) are no more than 45% efficient. For a diesel engine running on fossil diesel, the well-to-wheel (WTW) energy efficiency is about 35%. WTW efficiencies for biodiesel and bioethanol are limited to 18% and 13% respectively (Klintborn 2008). When idling and driving patterns in cities are considered, WTW efficiencies can be less than 20% for fossil transport fuels and less than 10% for liquid biofuels, or comparable to cooking over an open fire. On the other hand, electric cars have grid-to-wheel efficiencies of 60-70 % (Bossel 2006). This suggests that even within the limited role that biofuels are likely to play in the global energy portfolio in the coming decades, liquid biofuels may not be the wisest use for transportation. In addition, indirect effects of biofuel policy, including land use changes, transport sector adjustments, and others, must be considered if a new cost or benefit to a particular policy is to be assessed.

Notes

1. in the case of residues and wastes

2. However, the availability of cheap nitrogen for animal feed could have further – negative – consequences for the environment

References

AEA Technology. 2008. Review of work on the environmental sustainability of international biofuels production and use. Department for Environment, Food and Rural Affairs, London U.K.

Azevedo, D.A., E. Gerchon, and E. O. Reis. 2004. Monitoring of pesticides and polycyclic aromatic hydrocarbons in water from Paraý´ba do Sul River, Brazil. *J Brazilian Chem Soc* 15: 292–299.

Bento, A., and J. Landry. 2008. Efficiency Costs of Increased U.S. Biofuels Mandates. Working paper.

Bekunda, M., C.A. Palm, C. de Fraiture, P. Leadley, Luc Maene, L.A. Martinelli, J. McNeely, M. Otto, N.H. Ravindranath, R.L. Victoria, H. Watson, J. Woods. 2009. Biofuels in developing countries. In R.W. Howarth and S. Bringezu (eds) 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 NY, USA.(http://cip.cornell.edu/biofuels/)

Bossel, U. 2006. Does a hydrogen economy make sense? *Proceedings of the IEEE* 94:1826-1837.

Campbell, J. E., D. B. Lobell, R. C. Genova, and C. B. Field. 2008. The global potential of bioenergy on abandoned agricultural lands. *Enviro Sci Tech* 42:5791-5794.

Carvalho, C. E. V., Ovalle, A. R. C. , C. E. Rezende, M. M. Molisani, M. B. Saloma˜ o, and L. D. Lacerda. 1999. Seasonal variation of particulate heavy metals in the Lower Paraiba do Sul River, RJ, Brazil. *Envir Geology* 37:297–302.

Ceddia, M. B., L. H. C. dos Anjos, E. E. Lima, A. R. Neto, and d. S. L.R. 1999. Sistemas de colheita da cana-deac¸ucar e alterac¸ ˜oes Nas propriedades fý´sicas de um solopodzo´ lico amarelo no Estado do Espýrito Santo. *Pesquisa Agropecuaria Brasileira* 34:1467-1473.

Cerri, C.C., C. Feller, and C.A. 1991. Evoluc¸a˜o das principais propriedades de um Latossolo Vermelho-Escuro apos desmatamento e cultivo por doze e cinquenta anos com cana-de-ac¸u´car. *Cahiers ORSTOM Se´ rie Pe´dologie* 26:37-50.

Corbi, J. J., S. T. Strixino, A. Santos, and M. Del Grande. 2006. Diagno´ stico ambiental de metais e organoclorados em co´ rregos adjacentes a a´ reas de cultivo de cana-de-ac¸u´ car (Estado de Sa˜o Paulo, Brasil). *Quý´mica Nova* 29:61–65.

Crutzen, P. J., A. R. Mosier, K. A. Smith, and W. Winiwarter. 2008. N2o release from agro-biofuel production negates global warming reduction by replacing fossil fuels. *Atmosph Chem Phys* 8:389-395.

de Fraiture, C. and G. Berndes. 2009. Biofuels and water. In R.W. Howarth and S. Bringezu (eds) 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 NY, USA.(http://cip.cornell.edu/biofuels/)

de Oliveira, J. C. M., C. M. P. Vaz, and K. Reichardt. 1995. Efeito do cultivo contý´nuo da cana-de-ac¸u´ car em propriedades fý´sicas de um latossolo vermelho escuro. *Scientia Agricola* 52:50-55.

De Oliveira, J.C.M., K. Reichardt, O.O.S. Bacchi, L.C. Timm, D. Dourado-Neto, P.C.O. Trivelin, T.T. Tominaga, R.C. Navarro, M.C. Piccolo, F.A.M. Ca´ssaro. 2000. Nitrogen dynamics in a soil-sugar cane system. *Scientia Agricola* 57:467–472.

Dourado-Neto, D., C. Timm, J. C. M. Oliveira, K. Reichardt, O. O. S. Bacchi, T. T. Tominaga, and F. A. M. Ca´ssaro. 1999. State-space approach for the analysis of soil water content and temperature in a sugarcane crop. . *Scientia Agricola* 56:1215–1221.

Edwards, R., J.-F. Larive, V. Mahieu, and P. Rouveirolles. 2007. Well-to-wheels analysis of future automotive fuels and powertrains in the European context. JRC/Eucar/Concawe .

Edwards, R., S. Szekeres, F. Neuwahl, and V. Mahieu. 2008. Biofuels in the European Context: Facts and Uncertainties. Joint Research Centre, European Communities, Petten, Netherlands.

EIA (Energy Information Administration). 2008. International Energy Outlook. *US*DOE/ EIA, Washington DC..

Elobeid, A., S. Tokgoz, D. J. Hayes, B. A. Babcock, and C. E. Hart. 2007. The long-run impact of corn-based ethanol on the grain, oilseed and livestock sectors: A Preliminary Assessment. Center for Agriculture and Rural Development, Ames, Iowa, USA.

Erickson, G., T. Klopfenstein, T. Milton, D. Brink, M. Orth, and K. Whittet. 2002. Beef Cattle Report. University of Nebraska-Lincoln.

Fargione, J., J. Hill, D. Tilman, S. Polasky, and P. Hawthorne. 2008. Land Clearing and the Biofuel Carbon Debt. *Science* 319:1235-1238.

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, D.B. Lobell. 2008. Biomass energy: the scale of the potential resource. *Trends Ecol Evol* 23:65-72.

Gallagher, E. 2008. The Gallagher Review of the indirect effects of biofuel production. Renewable Fuels Agency., London U.K.

Galloway, J.N., J.D. Aber, J.W. Erisman, S.P. Seitzinger, R.W. Howarth, E.B. Cowling, and B.J. Cosby. 2003. The Nitrogen Cascade. BioScience 53:341-356.

Goldemberg, J., S.T. Coelho, and P. Guardabassi. 2008. The sustainability of ethanol production from sugarcane. *Energy Policy* 36:2086-2097.

Hill, J., E. Nelson, D. Tilman, S. Polasky, and D. Tiffany. 2006. Environmental, economic, and energetic costs and benefits of biodiesel and ethanol biofuels. *PNAS* 103:11206-11210.

IATP. 2007. Staying Home: How ethanol will change U.S. corn exports. . Institute for Agriculture and Trade Policy

IPCC. 1997. Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories. IPCC/OECD/IEA, Paris, France.

Jongschaap, R.E.E., W.J. Corre, P.S. Bindraban, and W.A. Brandenburg. 2007. Claims and facts on Jatropha curcas L., Plant Research International, Wageningen, The Netherlands.

Keeney, D. 2009. Ethanol USA. *Environ Sci Tech* 43:8-11.

Keeney, D., and M. Muller. 2006. Water use by ethanol plants: potential challenges. The Institute for Agriculture and Trade Policy, Minneapolis, Minnesota USA.

Klintborn, P. 2008. Well-to-wheel studies. Topsoe catalysis forum.

Lara, L.L., P. Artaxo, L.A. Martinelli, R.L. Victoria, P.B. de Camargo, A. Krusche, G.P. Ayers, E.S.B. Ferraz, M.V. Ballester. 2001. Chemical composition of rain water and anthropogenic influences in the Piracicaba River basin, southeast Brazil. *Atmosph Environ* 35:

Lara, L.L., P. Artaxo, L.A. Martinelli, P.B. de Camargo, R.L. Victoria, and E.S.B. Ferraz. 2005. Properties of aerosols from sugar-cane burning emissions in Southeastern Brazil. *Atmosph Environ* 39:4627–4637.

Leip, A., G. Marchi, R. Koeble, M. Kempen, W. Britz, and C. Li. 2008. Linking an economic model for European agriculture with a mechanistic model to estimate nitrogen and carbon losses from arable soils in Europe. *Biogeosciences* 5:73-94.

Macedo, I.C., J.E.A. Seabra, J.E.A.R. Silva. 2008. Green house gas emissions in the production and use of ethanol from sugarcane in Brazil: The 2005/2006 averages and a prediction for 2020. *Biomass and Bioenergy* 32:582-595.

Martinelli, L.A. and S. Filoso. 2008. Expansion of sugarcane ethanol production in Brazil: Environmental and social challenges. *Ecol App* 18:885-898.

Melillo, J.M., A.C. Gurgel, D.W. Kicklighter, J M. Reilly, T.W. Cronin, B.S. Felzer, S. Paltsev, C.A. Schlosser, A.P. Sokolov, X. Wang. 2009. Unintended environmental consequences of a global biofuels program. Report no. 168. MIT Joint Program on the Science and Policy of Global Change, Cambridge MA, USA

Melillo, J.M., P.A. Steudler, B.J. Feigl, C. Neill, D. Garcia, M.C. Piccolo, C.C. Cerri, H. Tian. 2001. Nitrous oxide emissions from forests and pastures of various ages in the Brazilian Amazon. *J Geophysicl Res - Atmospheres* 106 (D24):179-188.

Menichetti, E. and M. Otto. 2009. Energy balance and greenhouse gas emissions of biofuels from a lifecycle perspective. *In* R.W. Howarth and S. Bringezu (eds) 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, Gummers-bach Germany. Cornell University, Ithaca NY, USA. (http://cip.cornell.edu/biofuels/)

Parkin, T.B. and T.C. Kaspar. 2006. Nitrous oxide emissions from corn-soybean systems in the midwest. *J Environ Qual* 35:1496-1506.

Pate, R., M. Hightower, C. Cameron, W. Einfeld. 2007. Overview of Energy-Water Interdependencies and the Emerging Energy Demands on Water Resources, Sandia National Laboratories, Los Alamos, NM, USA.

Patzek, T.W. 2004. Thermodynamics of the corn-ethanol biofuel cycle. *Crit Rev Plant Sci* 23: 517-567.

Pereira-Netto, A.D., I.F. Cunha, T.M. Krauss. 2004. Persistence of polycyclic aromatic hydrocarbons in the soil of a burned area for agricultural purposes in Brazil. *B Environ Contam Tox* 73: 1072–1077.

Politano, W. and T.C.T. Pissarra. 2005. Avaliac¸a˜o por fotointerpretac¸a˜o das a´ reas de abrangeˆncia dos diferentes estados da erosa˜o acelerada do solo em canaviais e pomares de citros. *Engenharia Agrýcola* 25:242-252.

Prado, R.M., and J.F. Centurion. 2001. Alterac¸ ˜oes na cor e no grau de floculac¸a˜o de um Latossolo Vermelho-Escuro sob cultivo contý´nuo de cana-de-ac¸u´ car. Pesquisa *Agropecua´ria Brasileira* 36:197-203.

Prueksakorn, K. and S.H. Gheewala. 2006. Energy and greenhouse gas implications of biodiesel production from Jatropha curcas L. The 2nd Joint International Conference on "Sustainable Energy and Environment", Bangkok, Thailand.

RFA (Renewable Fuels Association). 2008. Ethanol Facts. (http://www.ethanolrfa.org).

Sanchez, P.A. 2002. Soil Fertility and Hunger in Africa. Science 295:2019-2020.

Searchinger, T., R. Heimlich, R.A. Houghton, F. Dong, A. Elobeid, J. Fabiosa, S. Tokgoz, D. Hayes, and T. 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. 2009. Government policies and drivers of world biofuels, sustainability criteria, certification proposals and their limitations. In R.W. Howarth and S. Bringezu (eds) 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 NY, USA.(http://cip.cornell.edu/biofuels/)

Silva, A.J.N. and M.R. Ribeiro. 1997. Caracterizac¸a˜o de Latossolo Amarelo sob cultivo contý´nuo de cana-de-ac¸u´car no Estado de Alagoas: atributos morfolo´ gicos e fý´sicos. *Revista Brasileira de Cieˆncia do Solo* 21:677-684.

Silva, A J.N., M.R. Ribeiro, A.R. Mermut, and M.B. Benke. 1998. Influeˆncia do cultivo contý´nuo da cana-de-ac¸u´ car em latossolo amarelos coesos do Estado de Alagoas: propriedade micromorfolo´ gicas. *Revista Brasileira de Cieˆncia do Solo* 22:515-525.

Simpson, T. W., A. Sharpley, R. W. Howarth, H. W. Paerl, and K. Mankin. 2008. The new gold rush: fueling ethanol production while protecting water quality. *J Environ Qual* 37:318-324.

Smeets, E.M.W., L.F. Bouwman, E. Stehfest, D.P. van Vuuren, and A. Posthuma. 2009. Contribution of N_2O to the greenhouse gas balance of first-generation biofuels. *Global Change Biology* 15:1-23.

Sparovek, G., and E. Schnug. 2001. Temporal erosion-induced soil degradation and yield loss. *Soil Sci Soc Amer J* 65:1479-1486.

Thamsiriroj, T.T., and J.D. Murphy. (in press). Is it better to import palm oil from Thailand to produce biodiesel in Ireland than to produce biodiesel from indigenous Irish rape seed? *Applied Energy*.

Tominaga, T.T., F.A.M. Cassaro, O.O.S. Bacchi, K. Reichardt, J.C. Oliveira, T.L.C. 2002. Variability of soil water content and bulk density in a sugarcane field. *Australian J Soil Res* 40::605–614.

USDA-NASS. 2008. Crop Plantings Report.

Veldkamp, E., A. Becker, L. Schwendenmann, D. A. Clark, H. Schulte-Bisping. 2003. Substantial labile carbon stocks and microbial activity in deeply weathered soils below a tropical wet forest. *Global Change Biology* 9:1171-1184.

Zah, R., H. Boni, M. Gauch, R. Hischier, M. Lehmann, and Wager. 2007a. Life cycle assessment of energy products: Environmental impact assessment of biofuels. Empa,