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The Impact of Expanding Biofuel Production on GHG emissions

White paper #1: Accessing and interpreting existing data.

WINROCK INTERNATIONAL

CONTENTS

Winrock International aims to support the efficient development of sustainable biofuels standards by assisting in providing access to relevant data on the technical, social, economic and environmental characteristics of biofuels.

Winrock International will develop three technical White Papers on GHG emissions, the role of water and building capacity to monitor standards. Three country impact evaluations of applying standards in national settings will be undertaken for the US, Brazil and Indonesia.

This White Paper focuses on accessing and interpreting data on GHG emissions related to biofuels.

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SUMMARY

This white paper synthesizes existing scientific data on greenhouse gas (GHG) emissions related to the production and expansion of biofuels. It is specifically focused towards assisting organizations that are developing sustainability standards for biofuels with the collection and interpretation of data.

There are a number of organizations developing sustainability standards or guidelines for biofuels. The organizations vary in their composition, structure and objectives. Many are voluntary multi-stakeholder approaches and while some are not considering greenhouse gases within their guidelines or standards, others are developing methods to quantify these GHG emissions or they are interested in promoting practices that can reduce GHG emissions.

This paper is based on peer-reviewed data and published GHG calculation methodologies and is principally focused on currently commercial biofuel production from sugarcane, corn, soy, rapeseed, palm oil and on future feedstocks (lignocellulosic material); switchgrass, miscanthus, agricultural and woody residues and short rotation coppice.

This white paper illustrates that:

- Existing modeling approaches cannot yet effectively and robustly define the global GHG impact of expanding biofuel production.
- Studies with system boundaries that measure ‘well-to-wheel’ GHG emissions can identify key contributing parameters within the biofuel supply chain. This approach can be used to develop appropriate guidelines to reduce GHG emissions.
- The well-to-wheel system boundaries as currently defined in many tools could provide future risks of double counting emissions or reductions e.g. emissions associated with fertilizer production counted in the chemical industry are also counted in the biofuel calculation.
- Reported well-to-wheel GHG emissions can vary according to methodological decisions, the use of different emission factors and uncertainties in data e.g. N_2O emissions from soil.
- Well-to-wheel GHG emissions can also vary substantially on the basis of different cultivation practices and fuels used to process biofuel. It is not possible to classify biofuel as ‘good’ or ‘bad’ on the basis of the feedstock they are developed from alone.
- The uncertainty associated with N_2O emissions from soil is significant and yet is a key component of the GHG emission profile of biofuels. Many tools being developed for sustainability standards rely on default IPCC calculations for N_2O emissions. Detailed models for calculating emissions exist in the US and Europe.
- Emissions associated with fertilizer manufacture differ between different types and play a key role in the emissions associated with biofuel crop cultivation. Opportunities to substantially reduce these emissions for ammonium nitrate production through GHG pricing mechanisms exist and would positively impact the GHG balance for biofuel.
- Emissions associated with some types of land use change can negate GHG savings associated with biofuels and lead to long ‘carbon payback times’.
- Co-product treatment method has a large impact on the GHG savings reported. There is no internationally agreed and consistent approach.

- Cultivation management practices to increase soil carbon sequestration and effective utilization of co-products can play a role in improving the GHG balance of biofuels, providing they are maintained long-term. Some emerging co-product markets (food grade CO_2) and their GHG implications have not yet been addressed.
- The reported GHG savings for biofuels differ depending on the reference they are compared to. A fuel that demonstrated an 80% GHG saving against a high carbon intensity reference translates into greater savings calculated as gCO_2eq/MJ_{fuel} than if the 80% GHG saving is related to a lower carbon intensity reference. If GHG benefits were monetized, this would result in different incentives depending on regional differences in the reference fuel.
- Incentives for GHG reduction ($\$/tCO_2eq$) are unlikely to represent a large proportion of net returns ($/ha) at $\$10/tCO_2eq$. In some cases such as sugarcane, the incentives may not be necessary to establish economically competitive biofuel markets; however land allocation decisions for advanced biofuel crops could be influenced by GHG incentives that reduce the breakeven returns (used as a proxy for land allocation decision). High yields per hectare and soil carbon sequestration rates are key and incentives greater than $\$10/tCO_2eq$ are likely to be required for advanced biofuels.

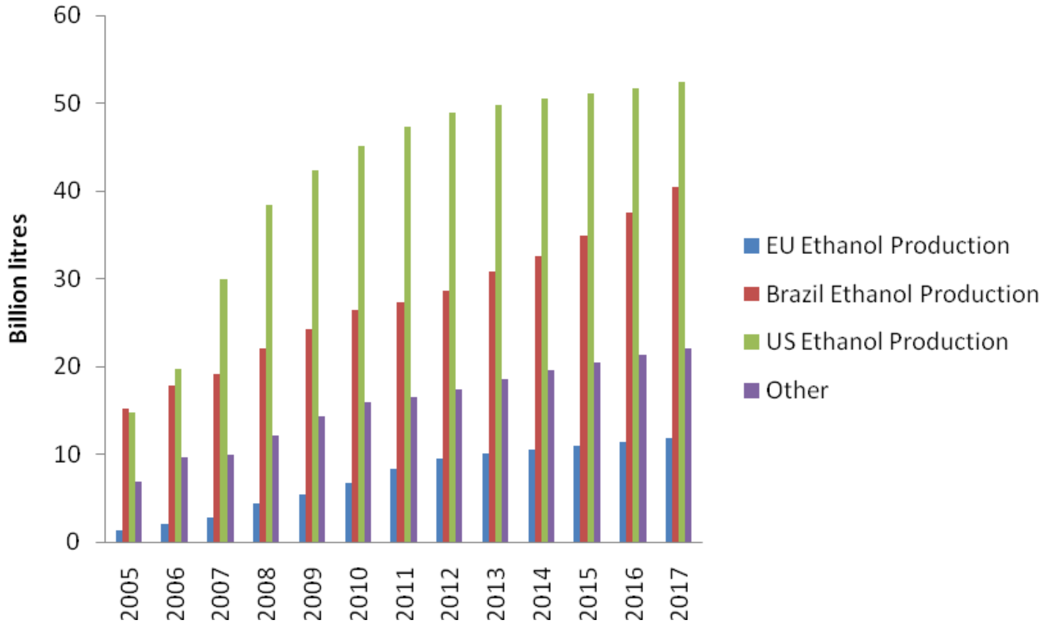
INTRODUCTION

Over recent years as energy security and environmental concerns have risen up various political agendas, there has been a substantial interest in biofuels and their potential contribution to energy security, mitigation of GHGs in the transport sector and also in delivering rural economic development benefits.

Many countries around the world have developed or are developing biofuel mandates that require specific and rising contributions within the transport sector over time. Brazil's Proalcool program, established in the 1970's and not without some setbacks along the way, has led to the current situation of an average blend of 25% of anhydrous ethanol in vehicles (BNDES, 2008). In addition, sugarcane now represents around 16% of the national energy mix (BNDES, 2008). Most recently the US and Europe have enacted mandates that will require substantially greater volumes of biofuels to be produced and consumed. The primary driver of biofuels policy differs in different parts of the world. In some regions that are substantial importers of fossil fuels with a declining agricultural sector, biofuels may be pursued in order to deliver a positive balance of payments and ensure a future for the agriculture sector.

Figure 1 and Figure 2 illustrate the substantial increases that are projected in world bioethanol and biodiesel production from 2005-2017. The EU is anticipated to be a small bioethanol producer in comparison to biodiesel, whereas Brazil and the US will dominate ethanol production.

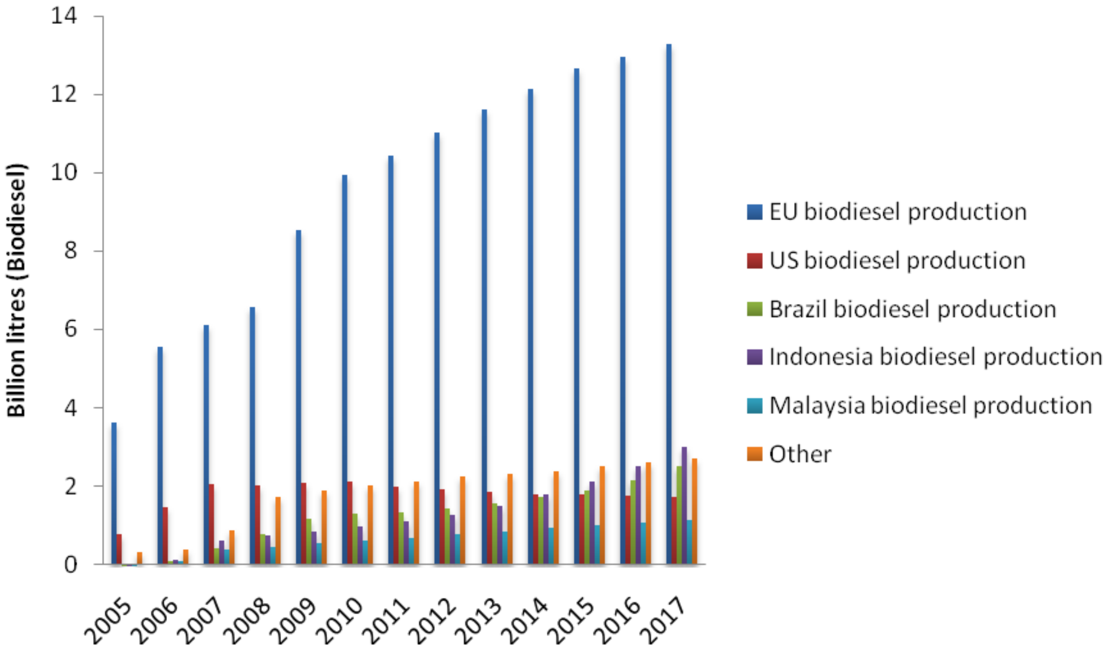
Figure 1: World ethanol production projection (OECD)



Source: OECD-FAO (2008)

Corn (in the US) and sugarcane (in Brazil) account for around 90% of all ethanol in use today. The agricultural area used for that purpose amounts to 10 million hectares, less than 1% of the arable land in use in the world (Zuurbier, 2008).

Figure 2: World biodiesel production projection (OECD)



Source: OECD-FAO (2008)

Not all feedstock for European biodiesel production will be grown in the EU. It is expected that, as is currently the case, imported soybeans and crude palm oil will be processed in the EU.

In the US the Energy Independence and Security Act of 2007 (EISA) will increase the original Renewable Fuels Standard (RFS) target[1] of 4 billion gallons renewable fuel production in 2006 to 36 billion gallons by 2022. The EISA categorizes fuels and caps the so-called ‘conventional’ renewable fuel (corn starch ethanol), so by 2022, 21 billion gallons of the 36 billion gallons required must come from cellulosic biofuel or advanced biofuels derived from feedstocks other than cornstarch.

The categorization of fuels within EISA contains specific life-cycle GHG emissions for biofuels relative to life-cycle emissions from fossil fuels as Figure 3 illustrates. The EISA states that these lifecycle emissions must include direct *and indirect* emissions.

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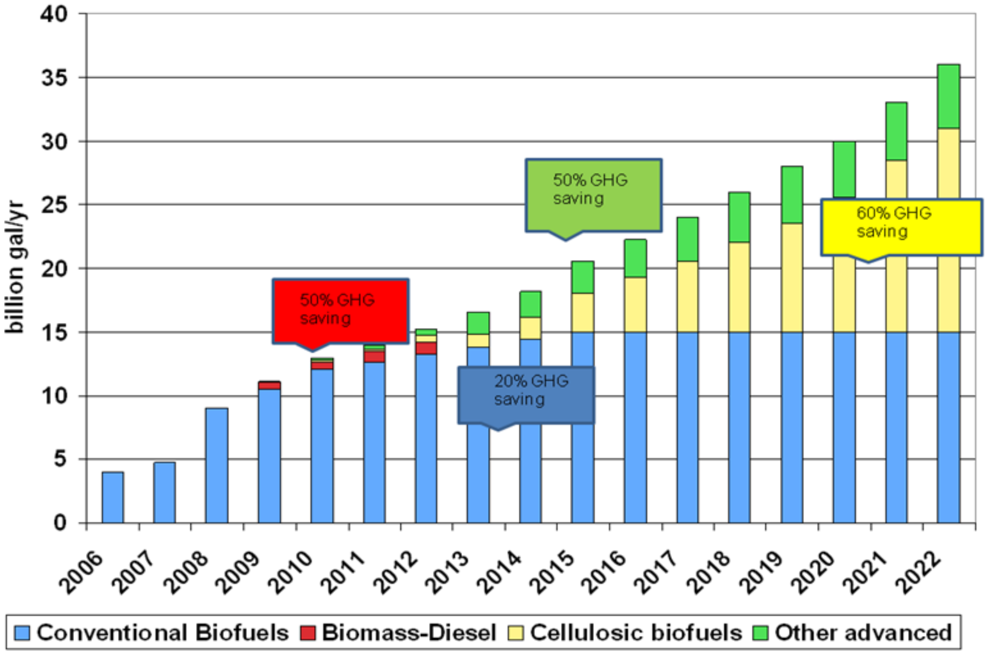
[1] Established in the Energy Policy Act of 2005

Figure 3: EISA volume and GHG requirements



In California the rules and technical data that will comprise the Low Carbon Fuel Standard (LCFS) are under development and intended to be complete in 2009. The LCFS is focused specifically on reducing GHG emissions in the transport sector and will require a ***reduction in carbon intensity of fuel chains by 10% by 2020*** in which biofuels may play a part.

The European Parliament passed the Renewable Energy Sources Directive at the end of 2008. It contains a specific mandate for Member States to include 10% (by energy content) of renewable fuel in the transport sector by 2020 and is expected to be met largely by biofuels. The mandate includes specific sustainability criteria including a requirement that the fuels meet a ***35% GHG saving initially, rising to 60% in 2017 and a requirement that biofuels used to meet the target are not produced from land with high carbon stock***

There is now a substantial focus on recognizing the potential for GHG emission savings from biofuels but with the publication of recent studies (Searchinger *et al* 2008, Fargione *et al* 2008) concern also exists about the potential emissions associated with consequential (or indirect) land use change of increasing biofuel production that could negate any GHG savings that displacing fossil fuel can deliver. A review into these indirect effects of biofuels on land use change was commissioned by the UK Government[2]. It concluded that there is a future for a sustainable biofuels industry but that significant risks of consequential (indirect) land use change exist (RFA, 2008b). The Government has since amended its target but recognized that a moratorium would make it significantly more difficult for the potential of biofuels to be achieved.

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[2] The ‘Gallagher Review’

SCOPE AND AIM OF THE WHITE PAPER

The aim of this white paper is to assist in the development of effective standards and guidelines for biofuels through the exchange and synthesis of existing information based on the best available science related to GHG emissions from biofuels.

The objectives of this paper are:

- Prepare a scientific knowledge-based synthesis about the key aspects of biofuels with respect to GHG emissions including land use change, cultivation practices and conversion technologies.
- Compare fuel chain GHG emissions of current and potential future biofuel systems.
- Identify key areas of uncertainty for standards organizations and identify actions that may improve GHG emission reductions.
- Explore the potential influence of GHG emissions on land allocation decisions with respect to economic valuations.

Where methodologies and tools have been developed and published these are compared within the paper and the data used within the calculations is illustrated and discussed. 'Best' practices that have been identified by organizations are also identified and, using peer-reviewed scientific studies, the value of potential incentives for emissions reductions carbon sequestration are quantified. This potential monetization of GHG emission reduction is explored in the context of providing incentives for allocating land to different feedstocks and/or undertaking specific management activities.

Box 1: Terminology and Metrics

- 1 gallon = 3.79 litres
- 1t on = 2000lb
- 1t (metric) = 2205lb
- 1t ethanol = 794 litres
- 1t biodiesel = 890 litres
- 1t corn = 39.4 bushels
- 1t soybeans = 36.8 bushels
- 1kg = 2.2lb
- 1hectare (ha) = 2.47 acres
- 1tC (carbon) = 3.667tCO_2eq
- 1MJ = 0.95mmBTU
- CO_2 = carbon dioxide
- N_2O = nitrous oxide
- CH_4 = methane
- CO2eq = greenhouse gas emissions expressed as carbon dioxide equivalent (based on IPCC global warming potentials)

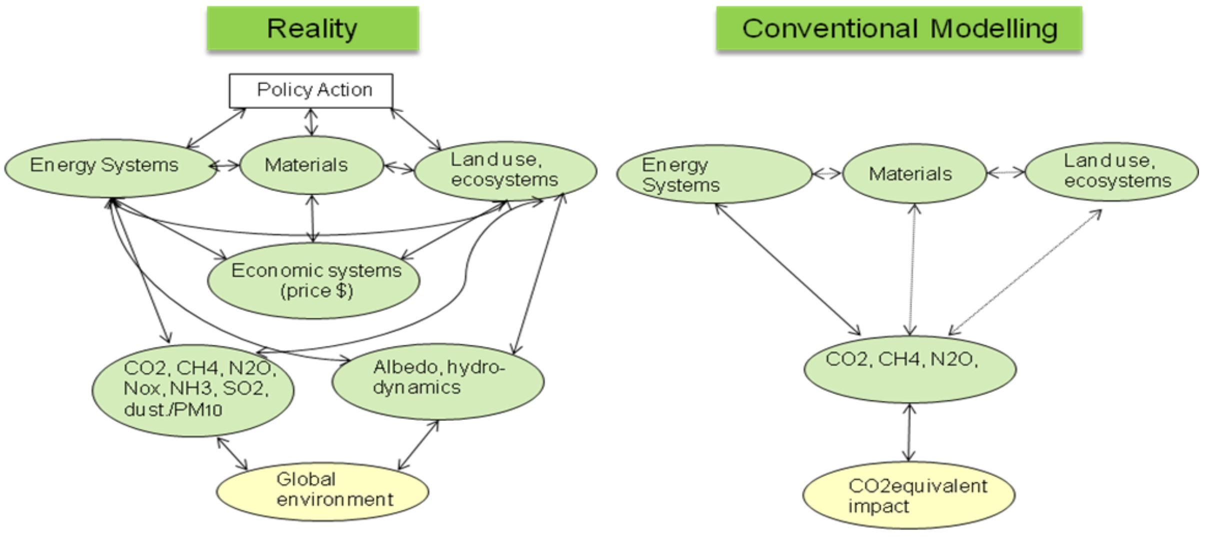
ASSESSING GHG IMPACTS OF BIOFUELS

The basis of assessment of GHG emissions from biofuels (or any product) is the identification of the global warming potential of the relevant gases. Over time, the global warming potentials of the gases has been reviewed, modified and published by the IPCC in their assessment reports. As illustrated below, nitrous oxide has a global warming potential nearly 300 times greater than CO_2 therefore even small amounts of this gas within the chain can contribute substantially to the net GHG impact. Methane has a global warming potential 25 times that of CO_2 according to the 4th IPCC assessment report.

	CH_4	N_20	CO_2
2nd IPCC Assessment report	21	310	1
3rd IPCC Assessment report	23	296	1
4th IPCC Assessment report	25	298	1

A robust quantification of the net global GHG impacts of expanding biofuels has not yet been agreed upon. Figure 4 illustrates the complexity of that this entails and shows the current modelling approaches have so far not captured the economic systems and wider climate forcing inter-relationships that would be required to effectively answer this question.

Figure 4: The reality of the impacts of GHG emissions versus current modelling approaches



Dotted lines indicate recent attempts to integrate these components.

Source: Redrawn and modified from Delucchi (2007)

Attributional vs consequential life cycle analysis (LCA)

The real interrelationships simplified in Figure 4 can be described as a consequential approach. A consequential approach to LCA integrates all aspects that could influence or be influenced by each other including economic and environmental feedback loops. A consequential LCA would describe and quantify the effects of oil or fertilizer price increases or investments in technology as well as climatological feedback loops and would result in a global & dynamic LCA.

> *There is no agreed method for conducting a consequential LCA. Current engineering-type approaches to estimating GHG impacts of biofuels identify key activities and can assist in the development of best practice guidelines for GHG mitigation*

The tools developed and applied to measuring biofuel GHG emissions to date are largely attributional approaches to LCA. These are based on process steps that add GHG emissions factors to calculate the GHG impacts. This attributional approach is likened to an engineering-type approach and is not usually integrated with price signals or feedback loops for example[3].

To date there is no agreed and accepted methodology applied to biofuels for a complete consequential LCA analysis. In addition, there will not be a fixed consequence or single conclusion from a consequential LCA as global economic circumstances change over time. The Lifecycle Emissions Model (LEM) from University of California Davis is likely the most comprehensive model to date for estimating transportation and energy lifecycles for energy use, criteria pollutant emissions, and CO_2-equivalent greenhouse-gas emissions. The lifecycle of fuels also includes the manufacture and assembly of materials for vehicles but does not fully include economic price impacts and price elasticities on GHG emissions. Regulators such as the US Environmental Protection Agency and California Air Resources Board are using LCA approaches combined with general equilibrium models in attempting to determine the overall impacts of biofuel policy. The European Commission is also required to undertake relevant research as part of a planned biofuel policy review in 2012.

Engineering-approaches to determining GHG impacts of biofuels identify key activities associated with GHG emissions in the fuels chain and should be used to assist in the development of better management guidelines within standards to reduce emissions[4].

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[3] Some attempt at consequential LCA is attempted within engineering approaches – treating co-product by system expansion (also called substitution or displacement) follows the logic of a consequential LCA. It follows the route to a change (or displacement) of the production of a co-product e.g. displacement of soymeal by corn DDGS and provides a credit to the biofuel based on the avoided emissions.

[4] This assumes that the underlying data on global warming potentials, N_2O emissions etc provide accurate results upon which to base best practices.

Box 2: Consequential LCA and the land use change debate

The current debate about indirect land use change (Searchinger *et al,* 2008) is a good example of a consequential approach to LCA. The overarching principle of the indirect land use change debate is that diverting existing crops to biofuel production to comply with policies induces a land-use change somewhere else in the world to 'fill the gap' in demand for the crop and the consequential GHG emissions from this land use change are attributed to the biofuel which are so large they negate any fossil displacement benefit.

In attempting to determine the consequential land use change a substantial number of interrelationships would have to be considered that are subject to debate. These include magnitude of yield responses to technological investments, increasing volumes of co-products that can substitute animal feedstuffs such as soymeal, changing land values and relationships to land allocation decisions, access to energy through rural electrification, potential increases in income for farming communities as well as potential impacts that fertilizer prices may have on land allocation and management decisions. Implications of valuing GHG emission reductions and the feedback loop this could establish would also have to be considered e.g. enabling cost-effective reductions in emissions in ammonium nitrate manufacture would influence future biofuel GHG savings as well as influencing decisions related to water management and irrigation. Further consideration of 'wastes' such as tallow would also need to be included in these analyses. A UK Government review into the implications of the use of tallow as a feedstock for biofuel observed that as a limited resource in the UK, the use of tallow for biodiesel *may* result in the use of heavy fuel oil to fire boilers where previously tallow had been used (AEA, 2008).

Biofuels and co-products

The production of feedstock and processing steps for liquid biofuels for transport generates co-products. The following table illustrates the productivity of 1 hectare with typical data. The data illustrate that evaluating productivity of biofuel based on gallons or liters of biofuel per hectare is misleading and the potential value from co-products should be acknowledged and considered in any evaluation.

Table 1 Typical productivity of 1 hectare for different crops processed to biofuel. Illustrative only.

Feedstock	Yield (t /ha)*	Primary co-product (per ha)	Intermediate processing (t/ha)	Biofuel production (per ha)	Secondary (processing) by-products (per ha)
Ethanol					
Sugarcane	78.8	21t bagasse & trash (dry)	-	8.6t ethanol	7250 MJ electricity$_{eq}$ Vinasse (fertilizer) CO_2
Corn (wet mill)	9.5	9.5t corn stover	-	4.7t ethanol	0.56t - corn oil 0.69t - gluten meal 2.99t - gluten feed CO_2
Corn (dry mill)	9.5	9.5t corn stover	-	4.6t ethanol	2.4t – DDGS CO_2
Miscanthus[1]	14[2]	-	-	4.4t ethanol	Electricity CO_2
Switchgrass[1]	13.5[2]	-	-	6.5t ethanol	Electricity CO_2
SRC e.g. willow[1]	8.8	-	-	3.7t ethanol	Electricity CO_2
Biodiesel					
Palm (fresh fruit bunches)	17.7	3.2t - Empty fruit bunches 7.2t - Old stems & fronds	3.5t – CPO 1.1t - palm kernel 2.8t – palm olein 0.6t – palm stearin	2.7t biodiesel	0.3t - glycerin 0.1t - potassium sulphate
Soybeans	2.8		0.48t - soy oil 2.06t – soymeal	0.45t biodiesel	0.05t - glycerin 0.01t - potassium sulphate
Rapeseed	3.1	3.1t - straw	1.26t rape oil 1.66t rapemeal	1.2t biodiesel	0.13t - glycerin 0.05t - potassium sulphate

[*] Yields vary substantially within different crop varieties but within different climates, soils etc. These are representative figures only.

[1] Lignocellulosic crops can also produce diesel through Fischer-Tropsch conversion technology

[2] Yields can be as low as 5.8t/ha for switchgrass (Khanna et al, 2008) and 9t/ha for miscanthus which would substantially alter results. Those reported here are based on relatively high yields reported for each crop.

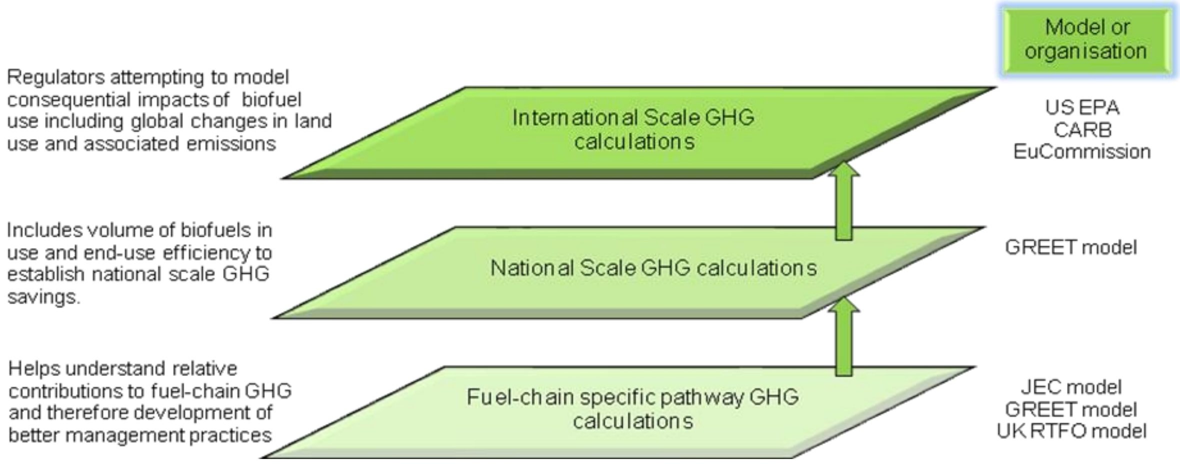
Sources: IGBE (2008), RFA (2008b), Macedo et al (2008), CA-GREET model, Woods et al (2006).

Scale of assessment

The scale at which the GHG assessment takes place influences the boundaries that are set for the assessment. System boundaries define the scope of the calculation i.e. what is to be included and excluded.

There are a host of detailed and comprehensive analyses (attributional approaches) of the fuel chain specific emissions from biofuels, some of which are detailed in Table 9, Annex A and illustrated in Figure 5. Whilst they do not provide the global picture they allow a closer exploration of the steps within the biofuel production pathway that can be measured and modeled and provide the foundations for establishing guidelines for reductions and fuel chain pathway GHG emission quantification. This paper focuses on the first level of assessment illustrated in Figure 5, the fuel chain specific pathways.

Figure 5: Scales of assessing GHG impacts of biofuels



Note: Acronyms are US Environmental Protection Agency (USEPA) and California Air Resources Board (CARB) using a variety of models to determine indirect impacts on land use change and the European Commission (EuCommission) required by legislation to do the same, UK Renewable Transport Fuel Obligation model (UK RTFO) from the Renewable Fuels Agency, Joint Research Centre, EUCar and Concawe model (JEC), Greenhouse Gases Regulated Emissions and Energy Use in Transportation model (GREET) from Argonne National Laboratory.

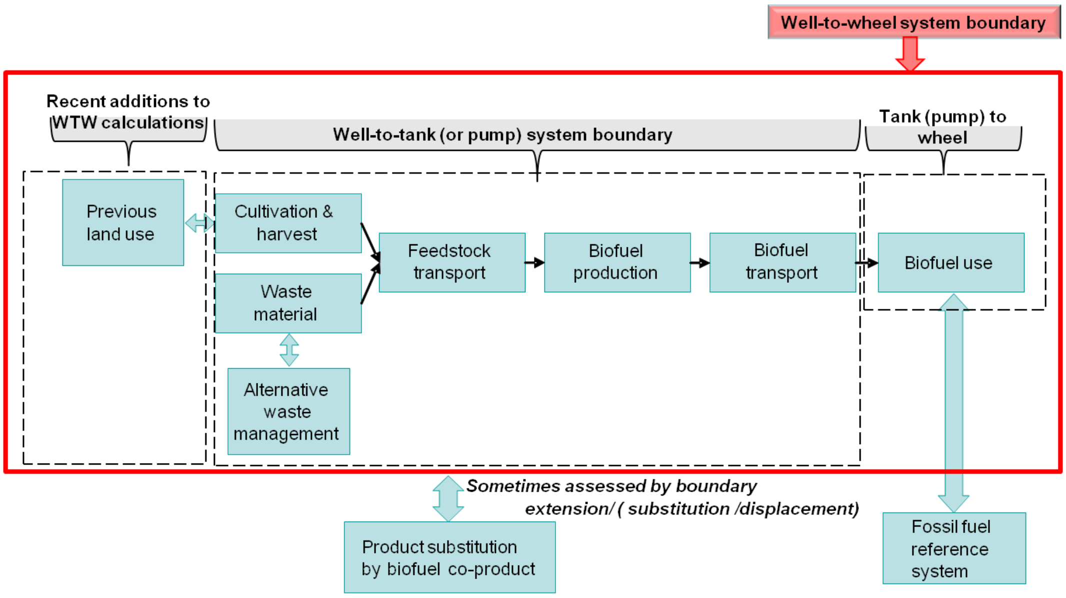
System Boundaries

The system boundary denotes all the units or steps that will be taken into account in calculating the GHG emissions for the biofuel. Figure 6 illustrates the system boundaries for a majority of well-to-wheel studies for biofuels. Previous land use was not considered in the past but is now becoming part of the analysis given the potentially significant contributions land-use change can make.

GHG emissions are often reported in two distinct phases, the *well-to-tank* (WTT) phase includes resource extraction, feedstock production, fuel production, refining, blending, transportation and distribution, and the *tank-to-wheels* (TTW) phase includes refuelling, consumption and evaporation. The complete fuel cycle analysis is also referred to as a *well-to-wheels* (WTW) analysis (Farrell & Sperling, 2007).

Well-to-wheel analyses are not full lifecycle assessments, even if land use change is added. The boundaries for full life cycle studies would be wider for example many of the well-to-wheel studies do not include small contributors (<1%) to emissions which may include seed manufacture. Emissions, associated with building and construction of machinery for biofuel production, are also not often included. The purpose of using a well-to-wheel study is to determine what activities can be undertaken to improve the emissions associated with cultivation, processing and transport.

Figure 6: An illustration of the units included in a well-to-wheel GHG analysis for biofuel.



Source: Redrawn from E4Tech (2008)

Fossil Fuel Reference

Fossil fuel reference systems are used to calculate the net GHG savings resulting from the displacement of fossil fuels by biofuels by comparing well-to-wheel emissions of the fossil fuel to the well-to-wheel emissions of the biofuel.

In some cases the fossil fuel that is displaced is assumed to be the *marginal* production from a refinery. The EU methodology (JEC, 2008) assumes Middle Eastern crude oil is the marginal fuel. In the draft Low Carbon Fuel Standard (LCFS) (CARB, 2009), reference fossil emissions are based on *average* crude recovery which takes into consideration crude extracted in California as well as crude recovered overseas.

The emissions associated with extraction, processing and transporting the fossil fuel are included in fossil references but manufacturing the equipment required to undertake the extraction are not included in many of the fossil references used to date given that they represent. These emissions have been included within the boundaries of some biofuel WTW calculations (Macedo et al, 2008) but not within others (JEC 2008 and CA-GREET (CARB, 2009) for example). Given the current debate on widening the system boundaries for biofuels to include indirect effects such as land use change there has also been discussion associated with appropriate boundaries for the reference fossil fuel to ensure comparisons can be made fairly.

Based on current calculations, fossil sources generally have a lower well-to-tank emission profile than biofuel but a higher tank-to-wheel emission profile. The biogenic carbon released on combustion of the biofuel released is often assumed to be negated by the CO_2 uptake in growing the crop; however methane and nitrous oxide emissions are also released at the tailpipe. In the draft version of the California Air Resources Board model (CA-GREET) for example a TTW emission of 0.78gCO_2eq/MJ combustion for diesel vehicles related to methane and nitrous oxide emissions is included and 0.82gCO2eq for gasoline vehicles.

Table 2 Comparison of WTT and TTW emissions

	JEC gasoline	JEC E5	CA-GREET (CARBOB)[+]	JEC diesel	JEC B5	CA-GREET diesel
Well-to-tank (gCO_2eq/MJ)	12.5	**	23.97	14.2	**	20.43
Tank-to-wheel (gCO_2eq/MJ)	73.38	73.31	72.91	73.25	73.39	74.9
Well-to-wheel (gCO_2eq/MJ)	***85.88***	*****	***96.88***	***87.45***	*****	***95.3***

[+] CARBOB is the blendstock not the end product

* depends upon well-to-tank emissions

** depends upon biofuel production emissions

Source: JEC (2008), CARB (2009)

The common metrics of measurement for biofuel are in units of mass per unit of fuel as energy (gCO_2e/MJ). This does not acknowledge the potential changes in efficiency owing to the use of biofuel in terms of distance travelled (gCO_2e/mile or km). The vehicle efficiency of biofuel blends should therefore be taken into account in calculating the overall GHG impact. JEC (2008) assumes that the energy efficiency of vehicles using low biofuel blends fuels would be the same as use of the base fuel. This is contradicted by the results of a 2007 study co-sponsored by the US Department of Energy and American Council for Ethanol (Shockey & Aulich, 2007) which suggests that using gasoline blended with ethanol in specific cars models can increase mileage per gallon compared to using unblended gasoline. There is no widespread agreement on these issues. Adoption of a different assumption to using a comparable energy basis has not been seen to-date within methodologies.

EXISTING STUDIES AND TOOLS FOR QUANTIFYING GHG EMISSIONS

Table 3: An overview of key parameters in existing models and methodologies

	UK Renewable Transport Fuel Obligation (UK RTFO)	EU Renewable Energy Directive (EU RED)*	JRC, EUCar, Concawe (JRC)	Greenhouse Gases Regulated Emissions & Energy Use in Transportation (GREET)	ERG Biofuel Analysis Meta-Model (EBAMM)	California Low Carbon Fuel Standard (CA-GREET)*
Location	www.dft.gov.uk/rfa	http://ies.jrc.ec.europa.eu/WTW	http://ies.jrc.ec.europa.eu/WTW	http://www.transportation.anl.gov/modeling_simulation/GREET/index.html	http://rael.berkeley.edu/ebamm/	http://www.arb.ca.gov/fuels/lcfs/lcfs.htm
Fuel chain coverage	International Large number fuel chain pathways Liquid biofuels	International Focus on liquid biofuels	International Substantial number fuel chains	USA focus Liquid biofuels, fossil fuels and solid biomass.	USA focus Corn ethanol and switchgrass	USA focus Liquid biofuels but includes Brazilian cane
Metric	gCO_2eq/MJ	gCO_2eq/MJ	gCO_2eq/MJ	gCO_2eq/MJ	gCO_2eq/MJ	gCO_2eq/MJ
System boundaries	Well-to-wheel (excl transport from refinery).	Well-to-wheel	Well-to-wheel	Well-to-pump *and* well-to-wheel Includes variety of end use scenarios.	Well-to-pump	Well-to-wheel
Co-product treatment	System expansion & allocation by market value	Allocation by energy	System expansion	All methods available in the tool	System expansion	System expansion for some and allocation by energy
Direct land use change emissions	Included only if a change reported	Not included by default	Not included	Limited	Not included	Under development
Annualised land use emissions	20 years	20 years	n/a	-	n/a	30 years
Indirect LU emissions	Not included	Not yet included	Not included	Not included	Not included	Will be included in final version
Global Warming Potentials - # IPCC Assessment Report	3rd report Some emission factors based on earlier reports	3rd report Some emission factors based on earlier reports	3rd report Some emission factors based on earlier reports	3rd report	3rd report Some emission factors based on earlier reports	4th report
Economic / price effects included	No	No	No	No	No	No

*Methodologies and results are draft and under discussion/consultation

KEY METHODOLOGICAL CHOICES THAT INFLUENCE RESULTS

Co-product treatment

All crops produce co-products during the cultivation or harvest stage and in their processing (Table 2). One of the key decisions in any methodology is how to account for the emissions that these co-products may have produced or may save. Table 4 illustrates the various options and which organizations are using which approach.

There is no internationally agreed approach to co-product treatment. ISO series 14040-14049 is an international standard series on lifecycle assessment but does not specify a single method to follow. It establishes a hierarchy of available options based on the principle that where possible any treatment by allocation should be avoided and system expansion used instead.

The choice of co-product treatment method is the single biggest methodological decision to affect the GHG balance.

Why do people choose different approaches?

The choice of co-product treatment in practical tools and studies has been based on a number of considerations including the aims and objectives for the specific application of the tool. A regulatory tool for example must consider the practicality of parties providing data, the administrative burden associated with undertaking calculations on a regular basis and the requirement to verify claims that are made whereas a scientific research study does not encounter the same limitations.

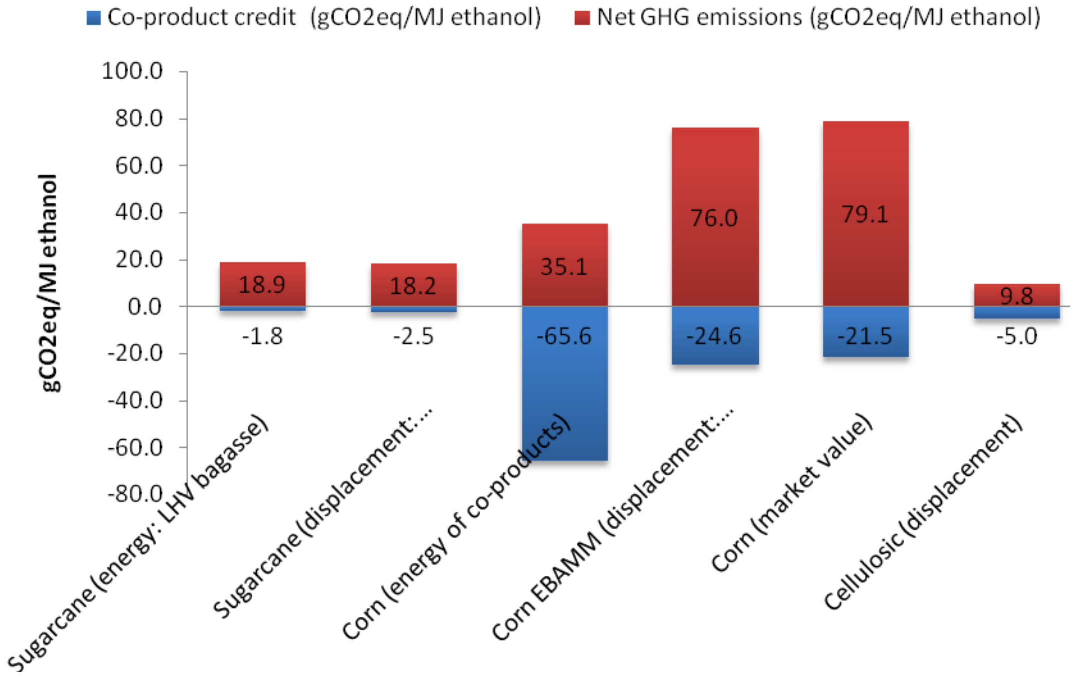
In assessing the GHG balance of the fuel, system expansion is widely acknowledged as the most appropriate approach to assessing the GHG impact of co-products. In some cases parties disagree as to the level of confidence that can be attributed to knowing what product the co-product has displaced. For some calculations, undertaken on a site specific basis it may be well-known that, for example, DDGS is burnt in a boiler rather than being used for animal feed. For other cases, co-product use may be uncertain and more difficult to assess- in these cases, the results are uncertain and the approach and calculations may be open to criticism. For other cases of other co-products and in a global situation it may be much more difficult to assess these interactions. In these cases the results are uncertain and this approach may be criticized. To avoid these challenges the European Renewable Energy Sources Directive (RED) specifies energy allocation for reporting purposes under regulation but will undertake a policy impact evaluation through a system expansion (or displacement) approach.

Table 4: Comparison of co-product treatment methods and tools using different approaches

	How does it work?	Why is this approach used?	Why is this approach not used?	Who uses this approach?
System Expansion (also called substitution or displacement)	Assumes the co-product substitutes another product. The co-product credit is then based on the GHGs of the avoided product.	It represents the consequences of production of the co-product – attempts to reflect a 'real' GHG balance that incorporates marginal impacts.	Determining exact uses of co-products and therefore consequences may be difficult and therefore may not represent actual impacts. Large number of co-products (biorefinery) may require substantial effort to determine use.	• GREET • EBAMM • JEC • UK for specific chains • ISO 14040 to 14049 recommends
Allocation by market value (price)	GHG emissions are allocated to products proportionally according to their market value.	Data is available. Price can determine a co-product's use and therefore provide a link to the GHG impact of the co-product. As price increases (demand increases) more of the emissions are attributed to the co-product.	Price varies geographically and over time influencing GHG balances.	• UK for specific chains
Allocation by energy content	GHG emissions are allocated to products proportionally according to their energy content.	Data is available. Minimum administrative effort needed to obtain data and undertake calculations. GHG balances do not change over time – less confusing for stakeholders.	The allocation process does not account for GHG impacts of the co-product production.	• EU Renewable Energy Directive • CA-GREET
Allocation by mass	GHG emissions are allocated to products proportionally according to their mass.	Data is available. Minimum administrative effort needed to obtain data and undertake calculations. GHG balances do not change over time – less confusing for stakeholders.	The allocation process does not account for GHG impacts of the co-product production. Can't allocate process heat/electricity by mass.	

Source: Adapted from E4Tech (2008)

Figure 7: Illustration of different co-product treatment methods on the resulting GHG balance.



The co-product credits for biofuels (in blue) can substantially reduce the overall GHG emissions (in red) when applied.

Source: Various – see Table 21 in Annex B

The impact of co-product treatment is well illustrated in the case of corn (this example in Figure 7 is for a dry mill process). At an average 2007-2008 US spot price of $468/t ethanol (FO Licht, 2009) and an illustrative DDGS price of $100/tonne, DDGS from a dry mill process represents 27% of the overall energy output and therefore 27% of total emissions are attributed to DDGS. If emissions from the same process are allocated on an energy basis the emissions attributed to DDGS represent 65% of the total[5]. For corn ethanol and any process that yields co-products with a high energy value, energy allocation results in a substantially more favorable net calculated carbon intensity of corn ethanol than a system expansion approach or allocation by market value. As the percentage of emissions allocated to the co-product increases the emissions associated with the biofuel decrease.

In the case of biodiesel, the choice of co-product treatment can also significantly affect the results of calculations. Figure 8 illustrates the results of different treatment methods for biodiesel feedstocks. It illustrates that, in general, an energy allocation approach produces lower GHG emissions for biofuel than the displacement (or substitution) approach owing to the large volume and relatively high energy content of the by-products. This effect is greater in the soy-biodiesel fuel chain than for palm or rapeseed biodiesel as it has a larger volume of co-product (soymeal).

Soybeans and rapeseed when crushed yield a high protein meal and oil. In the substitution approach, the meal is assumed/ modeled to displace another protein feed. The emissions avoided through this displacement are credited to the process, resulting in negative values for the pressing/ crushing stages for some chains.

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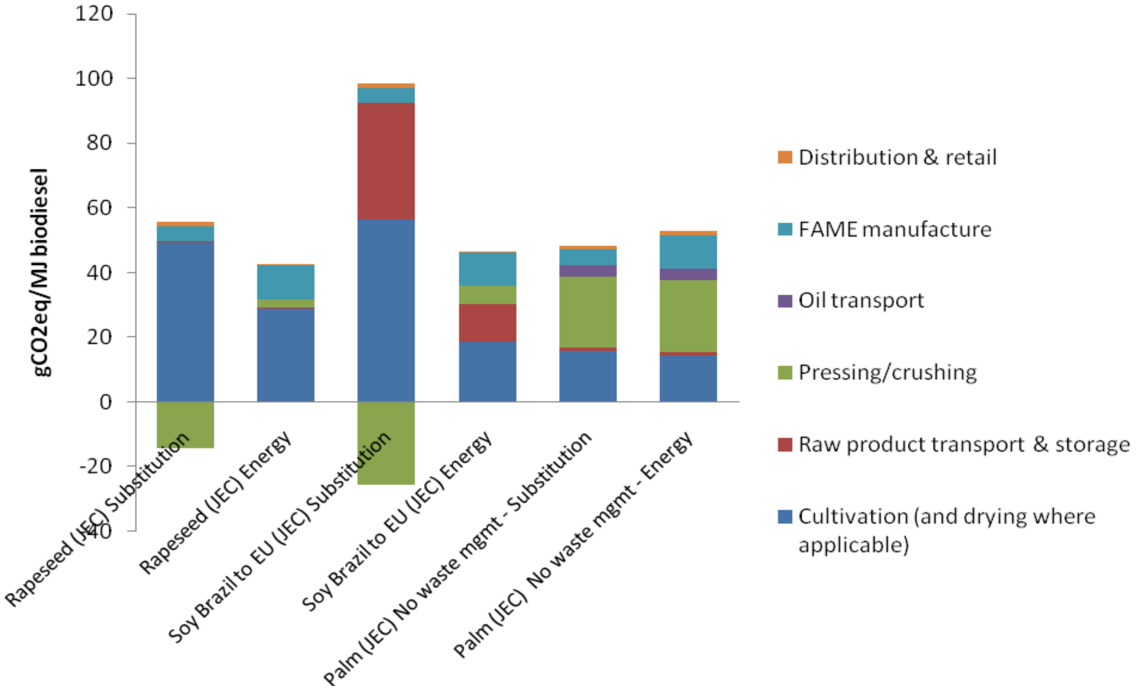
[5] Assuming 8414MJ ethanol/t corn; 0.5t DDGS / t corn (252kg DDGS/ t corn) and 21.79M/kg DDGS

The displacement (or substitution) approach in the study (JEC, 2008) assumes soymeal is credited at the pressing stage with the avoided emissions from the product it substitutes assumed to be EU produced wheat. Rapemeal is credited with the avoided emissions from soymeal which it is assumed to displace (1kg rapemeal is assumed to substitute 0.8kg soymeal).

The energy allocation process allocates emissions based on the energy content of each product at the pressing stage and therefore only a portion of the total emissions that the soybeans or rapeseed have attracted at that point are allocated to the oil as a biofuel feedstock. This approach has been followed in the draft CA-GREET process (version 2) (CARB, 2009) where approximately 46% of the emissions up to and including the pressing, are allocated to the soy oil that enters the biodiesel chain.

The net well-to-tank GHG emissions for soy are greater when substitution is used: emissions are 43.5gCO_2eq/MJ for an energy allocation treatment and 72.8gCO_2eq/MJ when substitution is used.

Figure 8: Comparison of different co-product treatments on GHG balances



Source: JEC (2008)

Future ‘biorefineries’ that are expected to produce a much larger number of co-products could make co-product treatment using substitution or energy allocation difficult and a time consuming approach. Allocation by economic value could be simpler but may but be as appropriate in some cases for example where agricultural co-products such as straw are used as animal bedding.

A COMPARISON OF FUEL CHAIN GHG EMISSIONS

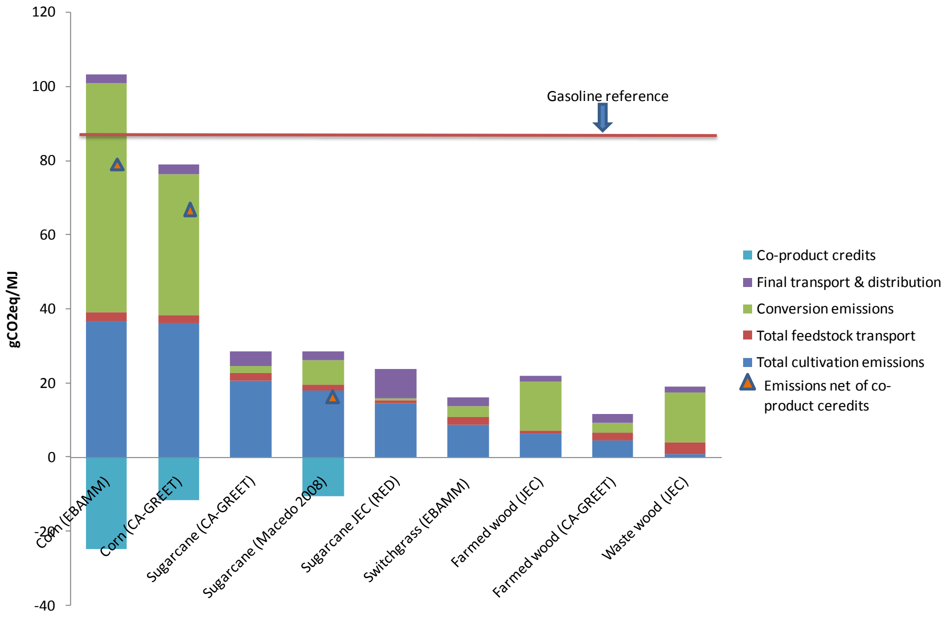
A majority of studies illustrate biofuels can deliver positive GHG balances and in some cases such as sugarcane they are substantially positive compared to the fossil reference. The assumptions under which the studies have been performed largely represent average conditions. Therefore each feedstock will have a range of GHG savings or emissions depending on the model assumptions or real-life practices. Each component of the biofuel chain is examined separately in this section to describe the conditions which can influence GHG emission calculations.

The fossil reference differs in different approaches: 87gCO2eq/MJ (JEC, 2008) or 95gCO2eq/MJ (CARB, 2009) however Figure 8 illustrates that, despite different co-product treatments, biodiesel feedstocks would reduce the well-to wheel emissions compared to diesel. The cultivation stage is a key contributor to the overall GHG balance and raw product transport for soy to Europe contributes significant emissions to that fuel chain. Oil transport, distribution and retail, as well as the conversion process to fatty-acid-methyl ester (FAME) biodiesel represent much smaller emissions within overall GHG balance.

GHG emissions from palm oil can be improved substantially by the introduction of waste management practices. Open-air waste treatment ponds produce methane (a more potent GHG than carbon dioxide) and by avoiding this practice the carbon intensity of the fuel chain (measured in gCO_2eq/MJ) can be reduced by around 39% (JEC, 2008).

Most ethanol fuel chains also produce GHG emission reductions compared to fossil gasoline. As with biodiesel, the cultivation stage represents a large proportion of emissions for all feedstocks. Conversion emissions are also a significant contributor to emissions. Final transportation of ethanol for use in the EU or US is relatively large in proportion to overall emissions within the fuel chain but not in comparison to transport emissions in other feedstocks for example.

Figure 9: Comparison of existing studies on typical ethanol well-to-tank emissions.



Source: CARB v2 (2009), JEC (2008), Macedo et al (2008), JEC (2008), UC Berkeley (2006)

Co-products can provide benefits that are included within the GHG balance for example corn ethanol produces a range of co-products that account for some of the emissions within the processing step. Sugarcane processing can be optimized to produce excess electricity from improving the efficiency of bagasse combustion. The excess electricity, when exported, is credited with displacing fossil electricity.

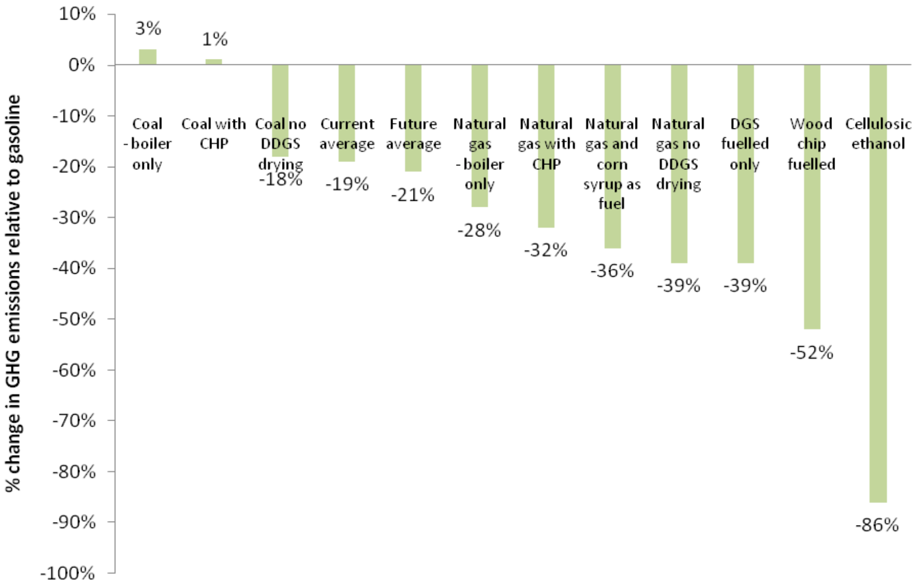
The remainder of the section explores some of the background detail that contributes to the fuel-chain emissions and explains how these values can vary depending on different activities.

Processing and conversion approaches

Processing and conversion technologies play a key role in influencing the GHG balance of biofuels.

A positive GHG balance for average corn ethanol in the EBAMM model (UC Berkeley, 2006) for example relies on the co-product credit to generate a net reduction compared to gasoline (discussed in the next section). However Figure 10 illustrates considerably worse performance would be demonstrated if coal was used for process energy and better performance if natural gas or biomass was the energy source.

Figure 10: Well-to-wheel GHG emission changes for corn ethanol under different processing scenarios compared to cellulosic ethanol.



Source: Redrawn from Wang et al, 2007

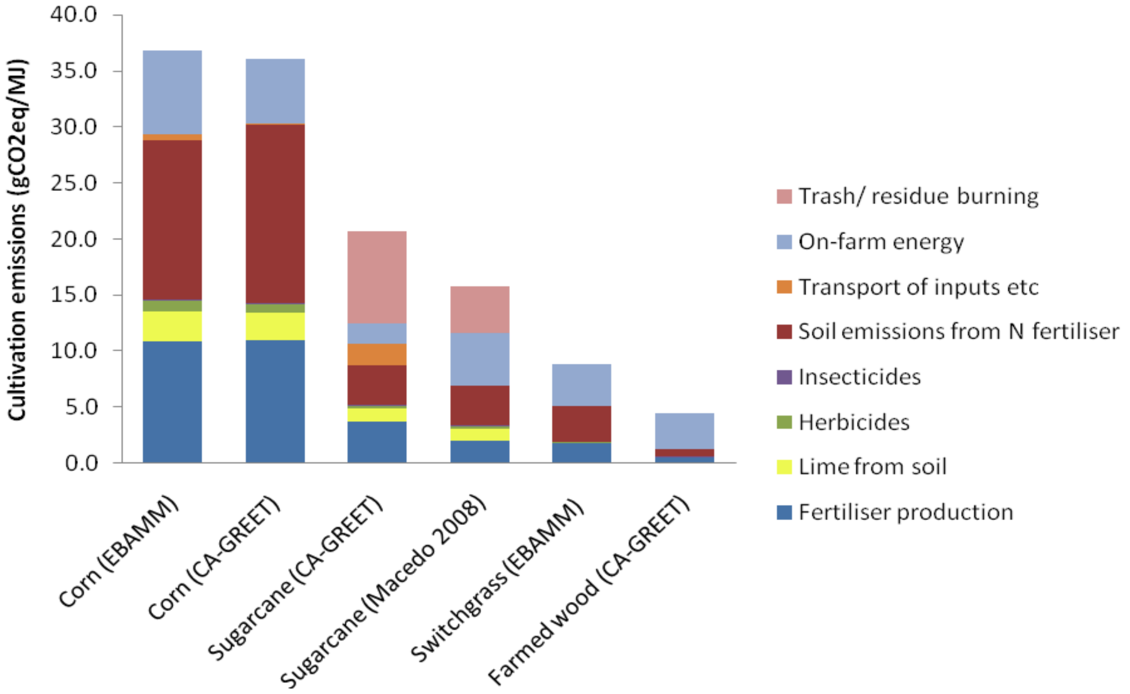
Sugarcane demonstrates particularly robust GHG savings through the use of bagasse as an energy source (represented by Sugarcane JEC in Figure 9) but potential still exists to improve boiler efficiency in many instances that would enable greater electricity production and export which would further improve GHG emissions. Macedo et al (2008) has estimated that with an increased uptake of high-efficiency boilers in Brazil by 2020 the GHG balance of ethanol from sugarcane will have improved sufficiently to increase net avoided emissions by c.40% compared to the 2006 reference case[6].

Current technologies for commercial production of biofuels use well recognized conversion and process pathways such as methyl esterification of vegetable oils for biodiesel and fermentation of sugars for ethanol. Technologies which are currently being developed for biofuels can utilize a wider range of crops such as low input and high yielding woody and herbaceous crops. The cultivation aspect of these crops represent good GHG balances e.g. for switchgrass and farmed wood (Figure 9) as the systems as generally low input but of important note is the contribution of emissions from enzyme and chemical manufacture in the conversion stage of ethanol from cellulosic routes that are sometimes included such as in the JEC pathways (JEC, 2008) but not necessarily in all cases.

Cultivation stage

The GHG emissions associated with fertilizer production and subsequent nitrous oxide (N_2O) emissions from the field are substantial contributors to cultivation emissions illustrated in Figure 11.

Figure 11: Comparison of the emissions associated with cultivation of ethanol feedstocks.



Source: UC Berkeley (2006), CARB v2 (2009), Macedo et al (2008)

Energy crops such as switchgrass generally have low GHG emissions in the cultivation stage. These woody and herbaceous crops generally require less fertilizer on an annual basis than annual crops such as corn which translates into a large benefit in current

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[6] Ethanol production is credited with the avoided emissions from fossil fuel use where bagasse is used for energy purposes.

calculations (emissions from soil and those from fertilizer production emissions are largely derived from type and volume of fertilizer application).

Trash burning (which releases methane and N_2O) within the sugarcane ethanol fuel chain is a substantial contributor. In the State of Sao Paulo in Brazil (representing over 80% of the country’s sugarcane production) this practice is being phased out and therefore emissions will be further reduced. The higher emissions for trash burning in the CA-GREET model assumes that burning of trash takes place on 100% of fields.

The application of lime to soil also results in CO_2 release when the carbonates in lime dissolve in water. Lime application rates and subsequent CO_2 release are uncertain (IPCC factors are generally used in methodologies) and can be a key component of GHG balances where high liming rates are found.

On-farm energy identified in Figure 11 relates to the diesel, gasoline, electricity and other energy requirements associated with planting, harvesting, irrigating and drying of crops. Highly mechanized processes represent a greater share of emissions. However for sugarcane, Macedo et al (2008) also consider embodied emissions for machinery and equipment manufacture[7] unlike the other studies illustrated in this paper. Embodied energy in equipment manufacture and buildings is usually low in comparison to energy flows associated with energy production (Macedo et al, 2008) and in comparison with the CA-GREET draft results illustrates that the inclusion of GHG emissions from construction does not significantly alter the GHG balance.

Figure 12 calculates the emissions associated with biodiesel feedstock cultivation expressed as gCO_2eq/MJ biodiesel. The data for feedstock production per hectare have been converted based on energy yields of biofuel per hectare. However, not all the inputs are related to the biofuel production chain e.g. soy meal that is produced from soy bean. The cultivation emission profile however illustrates the relative significance of cultivation inputs to the overall GHG balance. Soil N_2O emissions and nitrogen fertilizer production emissions are large in most fuel chains. On-farm energy in the form of diesel is also substantial in the case of soy cultivation. Herbicide application represents a more substantial contribution for soy than any other feedstock. Lower yields of oil from soy and rapeseed in comparison to palm relate to higher relative emissions per MJ of fuel produced.

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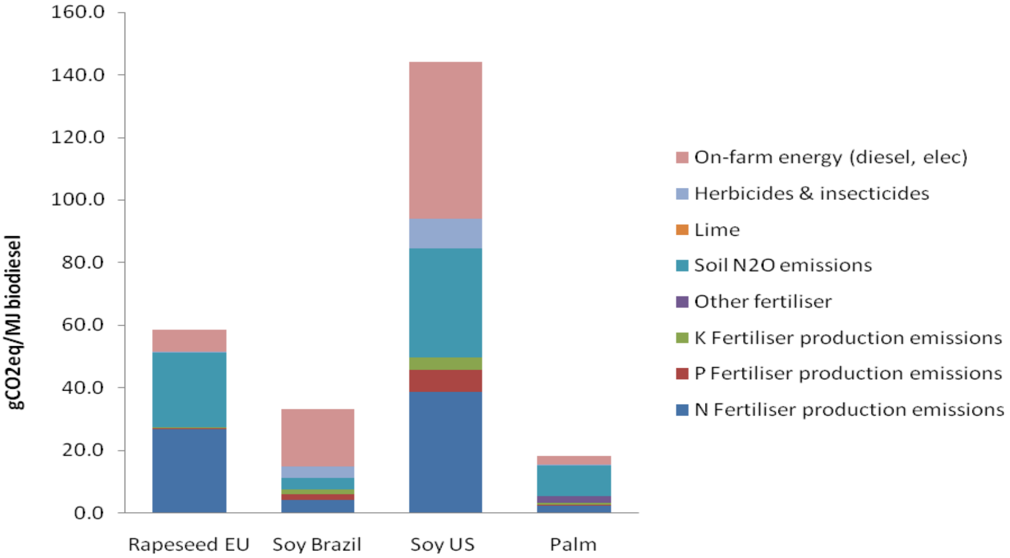
[7] $1.25tCO_2$ per t iron-steel. Electricity used in manufacture was not included in this Brazilian study as the contribution of hydroelectricity delivers a very low electricity emission co-efficient.

Figure 12: Comparison of the emissions associated with biodiesel feedstock cultivation. (No co-products treatment or allocation included).

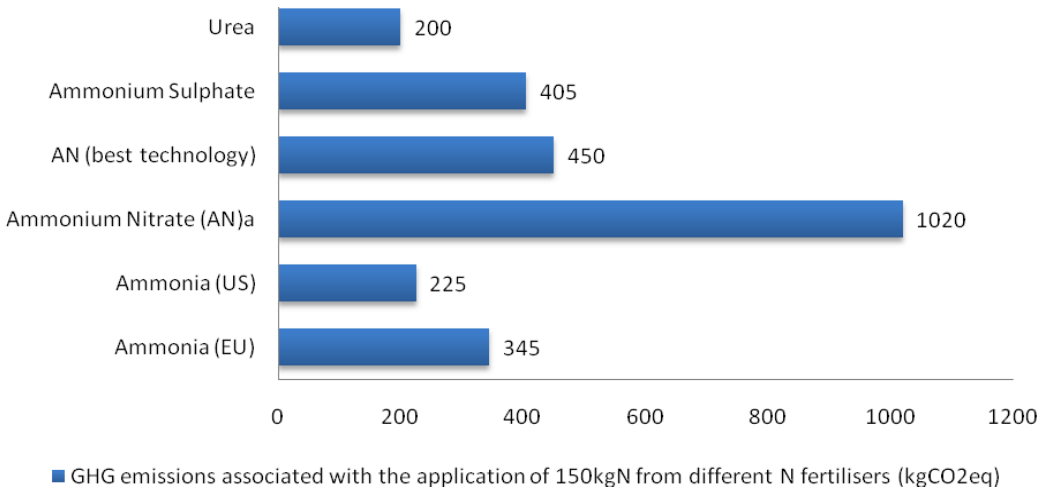


Sources: Input data from various sources including JEC (2008), RFA (2008a), Foreman & Livezy (2002). Emission factors assume 1.325% N2O from N application and other emission factors identified in Annex B.

A more detailed look at the underlying emission factors used in the cultivation stage illustrate that there are key choices that impact the emission profile. For example, Figure 13 illustrates that the type of nitrogen fertilizer used can make a substantial difference to the cultivation emissions. This is because energy required for the production of fertilizers differs and results in sometimes substantially different emission factors. Ammonium nitrate manufacture requires nitric acid which in its manufacture is the source of substantial GHG emissions. Urea doesn't require nitric acid therefore its emission factor is lower at present.

Fertilizer emission factors are themselves calculated through an LCA approach therefore some variations in results may be generated by differences in methodological approaches. Wood & Cowie (2004) illustrated the variety of results for the same production pathways and noted the lack of transparency in many studies which makes it difficult to compare results. Examples of methodological differences are allocation procedures (between fertilizers, industrial CO_2) and choices of global warming potential. An unlocked and transparent workbook updated calculations is available for ammonium nitrate in Western Europe (North Energy Associates, 2007). The basic source of data refers to typical production in Western Europe in 1995. However, it allows modification of data including the GWPs. Using the IPCC 2nd Assessment Report produces a result of 7.107kg CO_2eq/kg N. Using the 3rd Assessment Report produces a result of 6.925 +/- 0.260 kg CO_2eq/kg-N. Using the 4th Assessment report produces 6.979 +/- 0.260 kg CO_2eq/kg-N (Mortimer, *pers comm*).

Figure 13: Published data on GHG emissions from production of fertilizers.



Source: Wood & Cowie (2004)

Figure 13 illustrates that the use of best available technology to reduce emissions from ammonium nitrate production can reduce emissions substantially. In a review document for the IEA (Wood & Cowie, 2004) the use of best available technology reduces the emissions involved in the production of ammonium nitrate around 56% (6.80kgCO_2/kg-N compared to 3.0kgCO_2/kg-N).

Incentivizing the use of urea to reduce biofuel WTW emissions (based on this data) however could have serious consequences in increased levels of agricultural ammonia emissions compared to ammonium nitrate use (ammonia can increase acidification) (HGCA, 2008). The trade-offs associated with fertilizer choice between potentially increased acidification and increased GHG emissions should be recognized: regulation or incentivisation for GHG emissions for biofuels may inadvertently lead to a different environmental issue.

The current emission factors used for fertilizer manufacture within biofuel GHG calculations may be substantially altered if regulations or voluntary markets for GHG emission reductions are implemented. Cost-effective equipment to reduce GHG emissions in the manufacture of nitric acid for ammonium nitrate production would have a large impact on the many of the results of current methodologies that do not consider the use of best available technology for fertilizer production.

Emissions factors for fertilizer use vary between WTW studies (see Table 17 Annex B). In GREET and the EBAMM and draft CA-GREET the nitrogen fertilizer emission factor for production is 4kgCO_2eq/kg-N and relates to a combination of use of different N fertilizer types compared to 6.8kgCO_2eq/kg-N used within RFA (2008a) model for ammonium nitrate use only.

What impact does irrigation have on GHG emissions?

The emission profile associated with irrigation for different crops is illustrated well between corn and sugarcane. The EBAMM model calculates that emissions associated with the use of energy for irrigation contribute only 0.1% of the total emissions per hectare (1% of emissions when calculated as gCO_2eq/liter) assuming an average US grid emission factor. The EBAMM worksheet illustrates that energy associated with irrigation varies widely between studies: the worksheet detailing a Pimental and Padzek study results in irrigation energy contributing 25% of the emissions on a gCO2eq/liter basis as greater energy requirements are calculated and the electricity grid has a greater contribution from coal-fired sources.

For sugarcane, the most frequently modeled data on GHG emissions for sugarcane is from Brazil, primarily the center-south region, where cultivation is rain fed and there is no need for irrigation. However, for some regions of the world where irrigation will be adopted for sugarcane, irrigation may contribute additional emissions associated with its energy requirement.

Energy requirements differ according to irrigation technology, depth and distance to pump, crop type, soil type etc. Illustrative data (Alfaro & Marin, 1991) detail the energy requirement for the different irrigation systems in Latin America, according to potential and actual efficiencies. In order to provide an example of the implications of the energy requirements on GHG emissions, the data for yield, fertilizer application etc for the center-south of Brazil (Macedo et al, 2008) were used to calculate illustrative emissions associated with irrigation[8]. The results indicate that irrigation energy can substantially increase emissions compared to sugarcane baseline (an 82% increase in the case of conventional irrigation compared to rainfed) (see Table 20 and Figure 21 Annex C).

Transport

In the US, most biofuel production facilities are located close to corn and soybean acreage in the Midwest and therefore situated far from major consumption centers on the East and West coasts (EIA, 2007). Limited rail and truck capacity has complicated the delivery of ethanol particularly however, the transport phases are, in general, a smaller contributor to overall emissions within GHG compared to the feedstock cultivation stage. Large transport distances (c.7000km from Brazil to Europe) calculated within the sugarcane ethanol transport phase for the JEC study only increases the transport emissions for sugarcane by around 3.6g/MJ for a European destination and 1.5gCO_2eq/MJ for a US destination (draft CA-GREET model) compared to corn ethanol transported within the US for use in the US. Switchgrass transported in the US has a higher percentage contribution to the overall emissions compared to corn owing to a low overall emission profile but in general it has lower absolute emissions for transport.

Soybean transport for crushing and processing in the EU represents just over half of the fuel chain emissions according to JEC calculations[9]. The efficiency of bulk marine transport is 36% greater than truck transport in the study but despite this, the substantially greater distance, lower bulk density of soybeans and higher carbon intensity of fuel (heavy fuel oil) contributes to the substantial overseas transport emissions (35.9gCO_2eq out of total transport emissions of 37.2gCO_2eq/MJ).

This engineering approach to allocating emissions does not assume a baseline for transport i.e. it does not establish a ‘business as usual’ baseline and measure changes. For example, if corn or the DDGS were already being transported say 5,000km for animal feed in a business as usual case, a system change to locally using corn for ethanol and DDGS in a cattle lot would reduce the emissions compared to the baseline.

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[8] This is likely to underestimate emissions in parts of the world where fertiliser application is greater than that for Brazilian sugarcane.

[9] The calculation is a net GHG balance (including co-product credits) therefore the percentage would be reduced if the transport contribution was calculated as a proportion of gross fuel-chain emissions (without co-product credits).

Table 5: The contribution of all transport emissions within selected fuel chains

	Corn (EBAMM)	Corn (CA-GREET)	Soy US (CA-GREET)	Soy Brazil to EU (JEC)	Sugarcane (JEC)	Switch grass	Farmed wood (JEC)	Waste wood
gCO_2eq/MJ	5.0	4.9	2.7	37.2	8.6	4.3	2.4	4.7
% of total emissions	6%	7%	11%	51%	37%	26%	11%	25%

See Table 18 & Table 19 Annex B for details of transport emission factors

Source: UC Berkeley (2006), CARB (2009), JEC (2008)

Technological Improvements & GHG emissions

Yield improvement (for a given fertilizer application rate) provides substantial improvements for the GHG balance of biofuel. Figure 14 illustrates that of the selected crops, improving the biofuel yields by 20% improves the GHG saving for all crops and particularly for sugarcane ethanol.

Understanding the impact of different agronomic factors on yield, such as planting date, planting depth, plant population, row spacing, irrigation, fertiliser application (rate and timing) as well as interactions they have is essential to enable producers to optimize their production systems and improve the GHG balance.

Yield increases are a key element of improving GHG balances of biofuels. Perlack *et al* (2005) assume a 25% increase in corn yields by 2025[10] and a 50% increase before 2045 is possible and FAO predicts a yield increase slowdown from 1.2% to 0.9% per year. Worldwide there are many non-agronomic factors that relate to yield attainment such as short term weather impacts, development of economic infrastructure and political unrest. Concerted efforts to improve yields much address these issues where possible given the substantial opportunities that yield increases represent in avoiding some of the negative impacts of agricultural expansion such as increased GHG emissions from land-use change. However, sustainable residue harvest is site specific and USDA recommends tools[11] to predict sustainable removal rates that could ultimately enable soil carbon sequestration benefits to be realized and balanced against using the residue for displacing fossil energy requirements.

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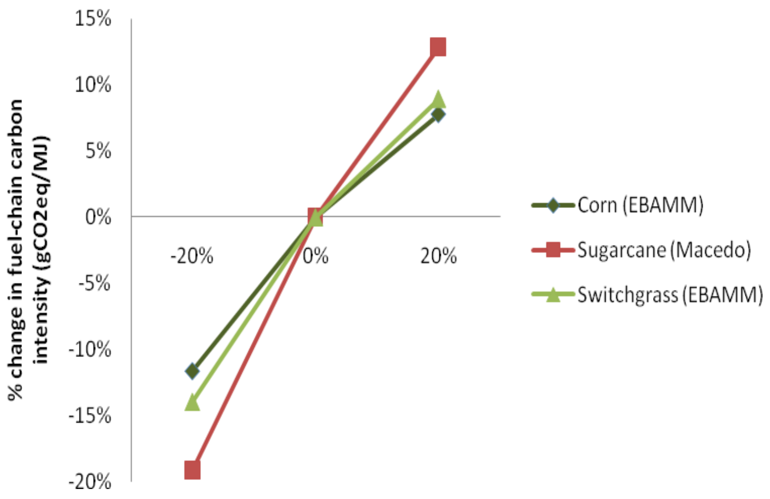
[10] To 173 bushels/acre

[11] RUSLE2, WEQ, and SCI

Figure 14: Sensitivity of fuel chain GHG savings to changes in yield (liters biofuel per hectare). See Annex C for detailed sensitivity analysis.



Source: UC Berkeley (2006), Macedo *et al* (2008)

Improvements in conversion practices should also be considered use of co-products may be used to offset fossil fueled electricity improve GHG emission reductions when employed to offset primary high associated GHG emissions such as coal or oil (see

Figure 10).

Within the cultivation stage, further developments could also substantially reduce GHG emissions. The development of nitrogen-fixing bacteria that enable yield maintenance while reducing fertilizer application has reportedly made substantial progress in Brazil (Embrapa Agrobiologia, 2008) and fertilizers with nitrification inhibitors which can reduce N_20 emissions from soil by up to 40% (Delgrosso, *pers comm., Smeets et al 2008*). Figure 17 illustrates the substantial positive impact these developments can have on the GHG balance of biofuels.

Influential parameters on the GHG balance

The results below are based on representative or illustrative studies therefore as circumstances differ or as the significant parameters are addressed, new parameters become important. For example, yield and energy use in conversion are influential parameters for GHG emissions for average corn ethanol in the US. If these are lowered over time, or differ between different locations, parameters such as N fertilizer and co-product credit become the most influential parameters. See Annex C for graphical representations of these sensitivities.

	Parameter # 1	Parameter #2	Comment	Source
Corn ethanol	Yield (t/ha)	Energy in bioethanol conversion	Co-product credits and nitrogen fertilizer have high influence. If energy conversion is lowered then N fertilizer influence increases in significance.	Based on 'ethanol today' in EBAMM (UC Berkeley, 2006)
Sugarcane ethanol	Yield (t/ha)	Electricity & bagasse surplus (co-	On-farm energy (diesel) also high influence and for some high efficiency plants already exporting is the #2 parameter.	Macedo et al 2008

	Parameter # 1	Parameter #2	Comment	Source
		products)		
Ethanol from switchgrass	Yield (t/ha)	N fertilizer	Credit for renewable energy in process energy is essential. Coal-based process increases base-case emissions (142gCO2 vs 9.8gCO2eq). Chemicals in conversion process are estimated to be higher in other studies and therefore significant parameters (JEC,2008)	Based on EBAMM model, UC Berkeley (2006)
Ethanol from forest residue	Conversion (chemicals)	Transport	Overseas transport of residues is an influential parameter but co-product credits for wood to electricity would be more significant if no overseas transport.	Based on JEC (2008)
Ethanol from straw	Conversion (chemicals)	Collection of straw	Overall emissions are much lower than most chains therefore the absolute influence of each parameter is lower when compared to other chains.	Based on JEC (2008)
Biodiesel – soy	Yield (t/ha)	Co-product credit for soymeal	Transport to crush (domestic) almost as high influence as co-product credit. Overseas transport not significant.	Based on RFA (2008a)
Biodiesel – oil palm	Yield (t/ha)	POME discharge	Co-product credit for palm kernel oil and glycerin are as significant as POME discharge. No co-products of electricity export are included and could offer substantial GHG benefits based on this influential parameter.	Based on RFA (2008a)
Biodiesel - rapeseed	Yield (t/ha)	N fertilizer	Co-product credits for rapemeal are almost as significant as N fertilizer. Based on illustration of production based in Germany	Based on RFA (2008a)

UNDERSTANDING LIMITATIONS OF APPROACHES TO DATE

In addition to the limitations of using attributional rather than consequential LCA approaches and a lack of market (or economic) influence within the calculations there are some issues that have not been sufficiently addressed to date.

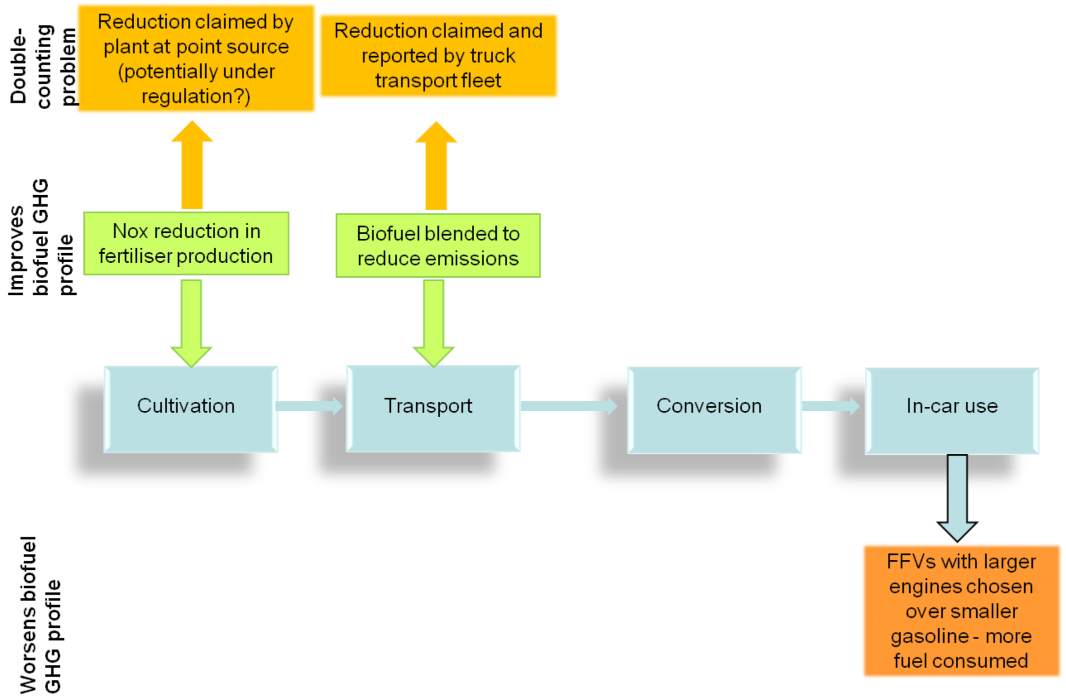
Future policy & risk of double-counting

Current approaches to accounting for GHG emissions at the ‘wheel’ of the vehicle require a large number of input values from further upstream including those from other industries e.g. the chemical industry. In many cases default values may be set to represent these inputs that attempt to represent typical situations e.g. the emissions associated with making nitrogen fertilizer.

Current and future regulations in other sectors, already included within the biofuel well-to-wheel calculation, may have implications for the well-to-wheel calculations and indeed the default values that are set. The proposal (now agreed) for a Directive as an amendment to EU Directive (2003/87/EC) related to the EU Emissions trading scheme (EUETS), states that as of 2013 CO_2 emissions from petrochemicals, ammonia and aluminium should be included in the EUETS, including N_2O emissions from the production of nitric acid. The emissions associated with nitric acid are already included within the emissions applied to fertiliser production (ammonium nitrate) and therefore if emission reductions are claimed by the petrochemical industry, they cannot also be claimed by the biofuel well-to-wheel calculation as this would double-count the emission reduction.

The Clean Development Mechanism (CDM) of the Kyoto Protocol, EUETS and similar schemes that measure & verify GHG emission reductions, have in the past focused on boundaries that are drawn relatively tightly to ensure that verification is robust. Effects beyond the boundary have been categorised as “leakage”. In the case of the biofuel well-to-wheel calculations, while drawing wider boundaries may help to refine the accuracy of the “leakage” component, it may tend to increase problems associated with double-counting and verification. In addition, difficulties associated with setting an appropriate “baseline” and proving “additionality” for CDM-projects become harder when biofuel mandates are agreed in countries and the current well-to-wheel calculations do not provide a solution for these problems. As an indication of these difficulties, there is no methodology approved for crop-based biofuels within the CDM process, despite a number of submissions.

Figure 15: Illustration of the biofuel WTW calculation with potential issues of double counting related to GHG accounting through different initiatives.



‘New’ co-products

One co-product not covered in great detail to date in WTW calculations is CO_2 produced from the fermentation process in ethanol production.

CO_2 can be generated in stand-alone fossil-fuelled combustion plants where the flue gas is extracted and can also be recovered through specific by-product recovery systems from a variety of sources such as ammonia production plants, geothermal plants and mineral processing plants for example.

A significant portion of the raw CO_2 used for liquefaction/purification in the US traditionally came from the fertilizer industry through ammonia plants, however owing to changes in the market these plants are reducing in number in the US which offers an opportunity for ethanol plants to utilize their CO_2 from fermentation (Rushing, 2005). Biomass and fossil-fired co-generation plants also offer an opportunity to capture and extract CO_2 from the flue-gas.

The extent to which the CO_2 from the ethanol plant would displace stand-alone direct fired CO_2 combustion plants and create actual emission reductions rather than simply replace CO_2 from other by-product recovery sources would need to be considered. However this significant development is one that should be considered.

Innovative models for biofuel production are attempting to make use of all outputs. A US company, Panda Ethanol, for example notes that ash from biomass sources is a non-toxic co-product and while it can be used successfully as a soil additive and may reduce fertilizer requirements, it could also be utilized in materials such as cement and cinder blocks or also doubles as a road bed material.

BNDES (2008) list approximately 20 different types of co-products that are currently or soon to be commercialized that could be produced from the sugarcane to ethanol process. Lysine, for example, as an essential amino-acid for animal feed (currently imported to Brazil) and citric acid used as a food preservative as well as for cleaning industrial equipment and in the manufacturing of detergents and other hygiene products.

Owing to the attributional nature of GHG emission calculations (see page 8), these future development and pathways have not yet been assessed to understand the potential significance on GHG emissions that co-products may play in the expansion of biofuels.

Land Use, Land Use Change and GHG emissions

Opportunities for increasing carbon sequestration through better management practices can be realized and has not been included within well-to-wheel analyses to date but the potential size of this opportunity has a wide range throughout the literature. There is evidence that, through judicious management, it is possible to increase the soil organic carbon pool in some soils and agro-ecosystems.

Box 3: Average reported figures for soil carbon sequestration

The reduced fuel associated with reduction in tillage results in GHG savings. The USA uses a national estimate of 0.045tonC/ha/yr (0.182tonneCO_2eq/ha.yr) reduction in emissions from CCT in non-irrigated corn & soybean compared to conventional tillage. (Murdock et al, 2007).

The soil carbon sequestration potential varies depending on soil type and surface texture, climate conditions and crop rotation. A USA average national sequestration rate for conservation tillage is identified as 0.377tonC/ha.yr (1.52tCO_2eq/ha.yr) (Murdock, et al 2007) but the results of studies illustrate that blanket claims of soil carbon cannot be taken for granted.

Amado & Bayer (2008) cite soil carbon sequestration estimates with a relatively low crop residue input at 0.12tC/ha.yr (0.44tonneCO_2eq/ha.yr) in subtropical environments and 0.03tonneC/ha.yr +/-0.07 (0.11tonneCO_2eq/ha.yr) for tropical climates. This is improved to 0.36tC/ha.yr (1.32tonneCO_2eq/ha.yr) and 0.42tC/ha/yr (1.54tonneCO_2eq/ha.yr)for higher residue input and cover crop inclusion.

Anderson-Teixiera et al (2009) report that miscanthus and switchgrass increased soil carbon by an average of 0.1-1tonneC/ha.yr (0.367-3.67tCO2eq/ha.yr) in the top 30cm and suggest that soil carbon sequestration under perennial grasses represents a substantial opportunity to improve the GHG performance of biofuels. They also report that even a 25% removal of corn stover reduces soil carbon.

Reducing tillage has been cited as a best management practice both to reduce emissions and increase soil carbon sequestration. Sometimes this is clarified further into ‘no-till’ or ‘intermittent-till’ where tillage practices include strip till, ridge till, mulch till. Tillage practice can also be combined with crop residue removal, either completely or with some retention. Data on soil carbon sequestration rates varies substantially and is not only affected by harvesting method. Climate, residue harvesting and soil type amongst other factors all play a role.

Crop residue contributes to soil organic matter and nutrient increases, water retention, and microbial and macro invertebrate activity. These effects typically lead to improved plant growth and increased soil productivity and crop yield, however some studies have illustrated these residues can contribute to increased N_20 emissions and reduced yields (Six et al 2002). While the addition of residues can assist in building up soil carbon, N_20 emissions from crop residues are not always taken into account in biofuel GHG calculation methodologies but are potentially important[12]. More research is needed to investigate the interactive effects of tillage, fertilizer application and crop rotation as they affect C-sequestration, CH_4-uptake and N_2O-fluxes, especially in tropical soils, where data is lacking.

GHG emissions associated with *changing* from one land use type to another can be considerable and sometimes sufficient to negate the GHG benefits of biofuel. Some studies have attempted to quantify these emissions to calculate emission factors that would be added to the fuel-chain emissions. The key variables in the calculations are often highly uncertain and can lead to substantially different results depending on the scale of assessment (site specific vs national or international). The key variables are:

- The volume of land that has been converted
- The soil carbon stock of the reference land type
- The above and below ground carbon stock of the reference land type

The time period over which emissions from change of land use is accounted is also important. Allocating total emissions over a shorter period produces higher values than if they are distributed over a longer time frame. Many methodologies (identified in Table 3) have attributed emissions over a 20 year timeframe in accordance with IPPC guidelines and that results in the identification of ‘annual’ emissions associated with land-use change that are added to the fuel-chain emissions.

Carbon payback time

The concept of a carbon payback time has been used as sustainability metric for biofuels (RFA, 2008a & 2008b, Gibbs *et al* 2008, Kim & Dale, 2008, Searchinger *et al* 2008). It is defined by calculating the emissions associated with changing a reference land use to cropland to produce a biofuel and dividing by the emissions saved by that biofuel in displacing fossil fuel.

Gibbs et al (2008) conducted a regional assessment of the yield potential for crop production and credited the associated biofuel production with displaced fossil gasoline and diesel

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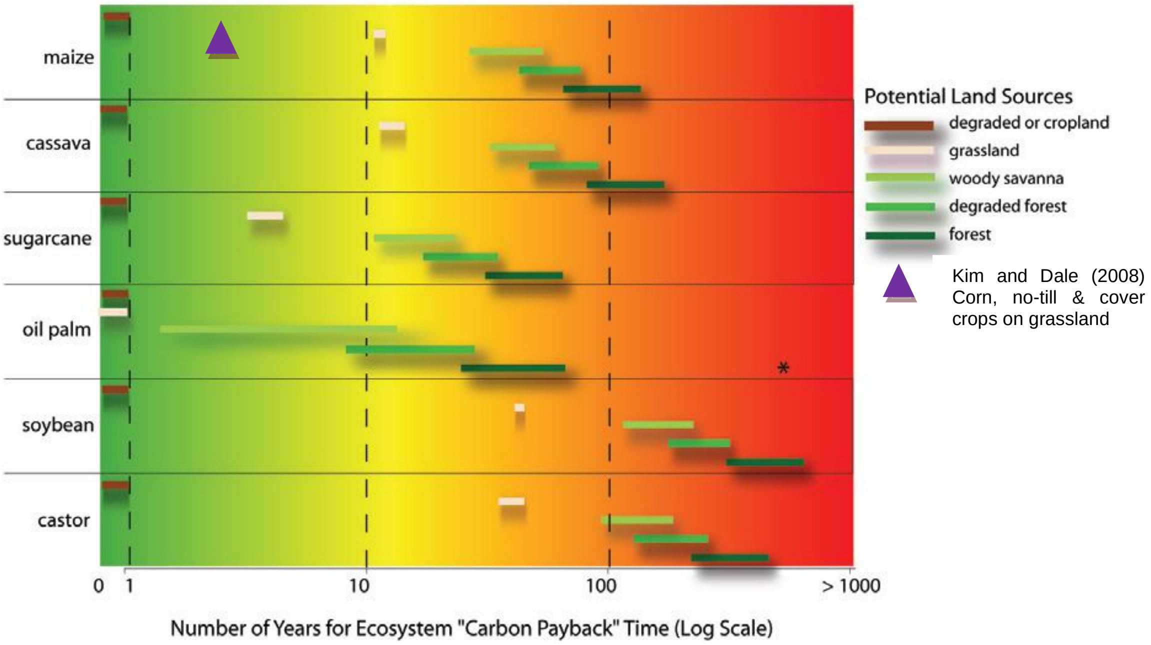
[12] The C/N ratio of crop residues appears to be a key variable in determining the amount of N2O produced but here are no process-based models that integrate above- and belowground dynamics with respect to C and N for biomass crops. Rather than rely on IPCC, one suggestion is for landscape scale estimations of N_2O emissions from residues based on area-based quantities of nitrogen in crop residues by crop type (JRC, 2004).

emissions. The results are shown in Figure 16, which illustrates the carbon payback times for different feedstocks and reference land uses. The examples do not include the well-to-wheel emissions of biofuels which would increase the payback time[13] but represent a high yield situation (crop-specific yields across the world are in the 90th percentile) and therefore represent faster payback periods than a current yield scenario.

Figure 16: Carbon payback times for a high yield scenario (90th percentile) from Gibbs (2008) & Kim & Dale (2008).



Note: Gibbs (2008) assumes no well-to-wheel emissions, no benefits of land management or co-product residue optimisation (land use change emissions and displaced liquid fossil fuel only). Background N_2O emissions from natural vegetation are not included. Asterisk refers to peatland conversion payback period of 918 years.

Source: Gibbs et al 2008, Kim and Dale (2008)

The range of carbon payback years for each of the bars in Figure 16 illustrates it is not generally possible to make definitive claims about the specific carbon impacts of land use change based on generic land categories such as woody savannah. Above & below ground carbon data differ substantially across woody savannah biomes in South America for example and represent significantly different emissions associated with conversion to cropland. Soil carbon stocks also vary substantially and are not robustly mapped at detailed scales. Some methodologies for land use change include changes in soil carbon stocks down to 1m, whereas typically the soil disturbance in land use change to cropland is not influenced down to 1m but is related to the top 30cm (Brown, *pers comm*.) which other studies account for (Anderson-Teixeira *et al*, 2008). Peatland conversion is widely agreed to yield carbon payback periods of hundreds of years but is also influenced by factors such as the depth of peat and subsequent cultivation practices.

Figure 16 illustrates that applying crop-specific yield improvement (Gibbs *et al* improved yields to the 90th percentile globally) to biofuel calculations still generate GHG emissions associated with land use change. The carbon payback time is improved, but not negated by improving yields. However, the potential for management practices to deliver additional GHG emissions reductions that impact the payback period are not accounted for by Gibbs et al (2008) and are illustrated by Kim and Dale (2008). The time taken to "pay-back" land use

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[13] On a logarithmic scale this is unlikely to represent a substantial deviation from the results as illustrated.

change emissions from for changing grassland to corn and using corn ethanol to displace gasoline was calculated as 12 years[14]. If management practices were changed to no-till compared to the reference case, this payback period was reduced to 4 years. If no-till was combined with the use of cover crops, this was calculated to reduce the calculated payback further to 3 years for conversion of grassland. The extent of the benefits of management practices varies on site-specific bases, however they can make a substantial difference to GHG calculations if maintained over a long time period to ensure the benefit is actually realized and counted as a carbon stock improvement.

Data uncertainties & variability

N_20 Emissions

N_2O emissions within the biofuel chain pathway are one of the largest sources of GHG emissions illustrated by studies to date and also one of the most uncertain parameters in the calculation. According to the 4th IPCC report, the Global Warming Potential of this GHG is 298 times greater, weight for weight, than CO_2 (IPCC, 2006).

Significant direct anthropogenic emissions of nitrous oxide occur from agricultural soils through:

- The use of nitrogen (N) fertilizers and animal manure.
- Mineralization of soil organic matter and crop residues.

Indirect emissions can also occur because N is leached from fertilized soils into ground water (where it is denitrified). Most of the uncertainty is related to the difficulty in estimating the emissions of N_2O that occur from soil and in estimating the indirect emissions that relate to specific activities.

Direct N_2O emissions have been shown to increase with the nitrogen application rate and a standard methodology proposed by the IPCC (2006) allows estimates of direct and indirect N_20 emissions based on nitrogen application without any other detail required. This method is used to calculate national anthropogenic emissions of N_2O from the use of fertilizers and animal manure and, in the absence of complete coverage of detailed soil biogeochemical models,

N_2O emissions typically account for 10% to 80% of the GHG balance (Smeets et al, 2008) but represent a significant uncertainty in the fuel-chain calculations to date.

While biogeochemical models for N_2O are available in the EU and US, there is a substantial lack of data from tropical and sub-tropical climates.

The use of an IPCC default based on N application is commonly used in methodologies but is a ‘top-down’ approach developed for national averages and unsuited to ‘bottom-up’ calculations that vary on a site-by-site basis.

In the EU, modeling shows N_2O emissions vary by a factor of more than 100 from one wheat field to another (JRC, 2008).

Optimized crop management can substantially reduce N_2O emissions.

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[14] Description of Kim & Dale (2008) reference case: Current average tillage (60% conventional tillage), dry mill,, use of cornstover in the dry meal for CHP (subst. of coal), utilisation of wood for from forest for CHP (subst. of coal), Most recent data on yield and energy usage, 1 kg DDGS substitutes (from Greet model) 0.95 kg of dry corn grain and 0.3 kg of dry soybean meal and 0.03 kg of N in urea.

this default approach (fertilizer-induced emission) is often used in methodological tools to calculate GHG emissions from biofuels.

The IPCC default for *direct and indirect* emissions assumes 1.325% of all N applied is emitted as N_20 (see Annex B) but the uncertainty associated with the calculation means that this can range from approximately 0.01% to 3.5% of nitrogen applied (Smeets *et al* 2008).

This linear rate of N_2O emission associated with fertilizer addition is subject to considerable uncertainty. The actual N_2O flux is related to soil temperature, water, and texture; anaerobic soil conditions (reduced oxygen supply); microbial and fungal populations and type; legumes in crop rotation; crop residue type and amount (C:N ratio); type of N fertilizer applied; fertilizer placement in soil; tillage system; year to year variations in climate and irrigation systems. In the EU for example, N_2O emissions within model results, varied by a factor of more than 100 from one EU wheat-field to another (JRC, 2008b).

Stehfest & Bouwman (2006) have conducted the most comprehensive assessment of measurement data on NO and N_2O emissions to identify the factors that significantly influence these parameters for agricultural fields and soils under natural vegetation. The conclusions of this study are that factors with a significant influence on N_2O emissions from agricultural soils were:

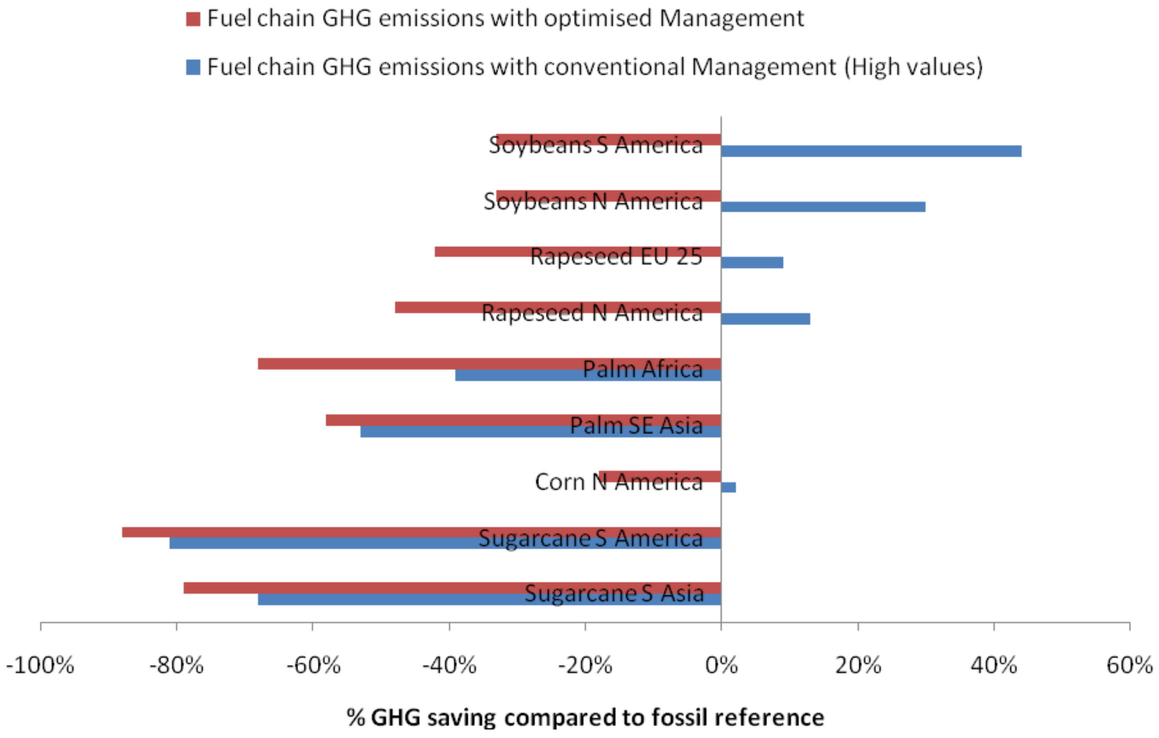
- Environmental factors (climate, soil organic C content, soil texture, drainage and soil pH);
    - For example, the oxygen and moisture status and gas diffusion in agricultural soils depend on soil texture and drainage. Fine textured soils have more capillary pores and hold water more tightly than sandy soils. As a result, anaerobic conditions may be more easily reached and maintained for longer periods in fine textured soils.
- Management-related factors (N application rate per fertilizer type, type of crop, with major differences between grass, legumes and other annual crops);
- Factors related to the measurements (length of measurement period and frequency of measurements).

The study also found the most important controls on NO emission include the N application rate per fertilizer type, soil organic-C content and soil drainage.

Based on the results of the study, Stehfest and Bouwman (2006) published a model that allows a more detailed approach to be used in calculating the N_2O emissions. This is detailed in Annex B, though the authors stress that despite the greater level of detail, uncertainties still remain which can affect the resulting GHG balance by more than 100%.

A study of GHG emissions specific to biofuels based on the Stehfest and Bouwman methodology has been undertaken by Smeets et al (2008). The N_2O emissions within the calculation relate to two crop management systems; conventional management and optimized crop management. Optimized management includes an optimized nitrogen fertilization regime and the use of nitrification inhibitors, which reduce N_2O emissions. The optimized management case gives insight into the technical potential to reduce N_2O emissions, which is substantial as Figure 17 illustrates. The study is focused on direct N2O emissions and excludes the GHG emissions due to positive or negative changes in above-or belowground biomass, soil organic matter and litter that result from the conversion of land into energy crop plantations or through better management practices which would impact GHG balances.

Figure 17: Impact of an optimized crop management system for N_2O on the well-to-wheel GHG balance for biofuels



Source: Smeets et al (2008).

WHAT IMPACTS ON DECISION-MAKING MIGHT VALUING GHG REDUCTIONS HAVE?

It is widely acknowledged that biofuels represent a more expensive way to reduce GHG emissions than other forms of renewable energy. However the costs of production vary widely depending on the type of crop, climate, cultivation and process practice as well as transport requirement and costs of energy and labor.

The biomass feedstock costs broadly represent about two-thirds of the biofuel production costs and therefore lower feedstock production costs represent improved biofuel economics. Some feedstocks at present including corn and rapeseed rely on incentives to make them competitive with fossil fuels. Long-term commitment to sugarcane in Brasil has led to a very cost competitive feedstock for biofuel production that is cited at an average of US$0.22/liter (Zuurbier, 2008) or a breakeven with oil at $35/bbl[15]. Other countries with high fossil fuel dependency and low agricultural costs of production that may benefit substantially from the

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[15] On a volume basis assuming 159litres per barrel.

development of biofuels may not require substantial incentives to deliver a cost-effective alternative for fossil fuels.

Other feedstocks have higher production costs and based on a compilation of incentives for feedstock production, processing and biofuel sales, Steenblik (2008) estimated the costs of saving GHG emissions with corn and EU rapeseed at \$520/tCO_{2eq} for US corn ethanol and \$1000-\$1340 for EU rapeseed biodiesel. This is substantially higher than the approximate \$10-\$20/ tCO_{2eq} achieved in carbon trading schemes. By comparison, removal and sequestration of carbon dioxide from coal fired power plants in the US has been estimated to fall in the range \$34 to \$70 /tCO_2 for retrofitting existing plants (DOE, 2008).

Within the transport sector there are more limited options to address emissions with alternative compared with electricity generation for example and the production of biofuels and their feedstock offers opportunities to produce co-products for animal feed or process energy requirements in addition to leading the transition towards bioplastics for example. Biofuels are not the sole solution to address GHG emissions in the transportation sector but can play a role if pursued effectively.

There are a number of mechanisms that could realize revenue associated with GHG reductions for biofuel. Incentives can be targeted at various parts along the supply chain from upstream at the feedstock producers' biofuel producers or downstream at the biofuel suppliers. Bioelectricity production associated with co-products (e.g. bagasse cogeneration) already has approved Clean Development Mechanism (CDM) methodologies for use in claiming Certified Emission Reductions[16] (CERs). The voluntary emission reduction (VER) market could deliver opportunities for generating value assuming robust and credible methodologies are developed. Policy measures associated with rewarding biofuels in accordance with their GHG performance have also been proposed by the UK Government (DfT, 2007).

Downstream incentives

Biofuel mandates, such as those pursued in the EU, generally require a fossil fuel supplier to supply a certain percentage of biofuel. Incentives that could be provided on the basis of the GHG savings of the biofuel would be received by this downstream party. There are potentially important considerations for providing incentives based on the GHG saving for biomass fuels that relate to the reference carbon intensity i.e. what is being displaced.

Double-counting could become a significant issue

Well-to-wheel calculations include a large volume of input data owing to the wide system boundaries. Some from the input data may be influenced by current or future regulations. Emissions associated with fertilizer manufacture for example are included within the biofuel WTW calculation but are also now included within the EU emissions trading scheme from 2013. A reduction in emissions from the fertilizer plants cannot be claimed both by the manufacturing plant and the biofuel through its WTW calculation (this would double-count the benefit). Consideration must be given to how WTW calculations could evolve to avoid these issues associated with future regulations and carbon markets.

The level of incentives depend on what is being displaced

A biomass fuel that demonstrated 80% GHG saving against a high carbon intensity reference (e.g. a coal-based electricity grid) translates into greater savings calculated as

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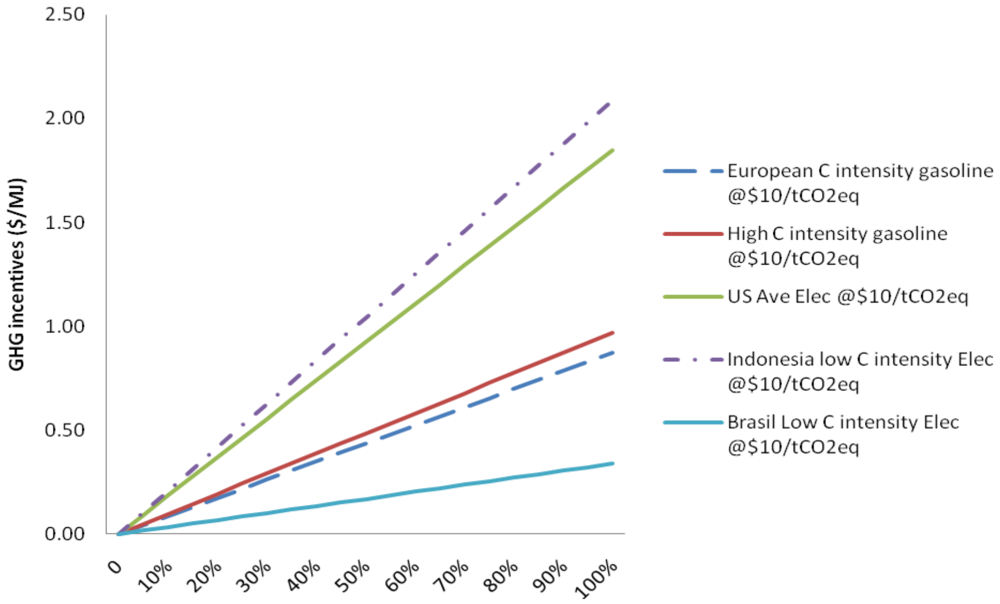
[16] AM0015 – Bagasse-based cogeneration interconnected to the electricity grid.

gCO_2eq/MJ_{fuel} than if the 80% GHG saving is related to a lower carbon intensity grid, or liquid fossil fuel. For example when the incentive is based on a price of $\$10/tCOt_{eq}$ Figure 18 illustrates that owing to a higher carbon intensity of electricity (e.g. US average) compared to that of fossil gasoline, bioelectricity from the feedstocks potentially used for biofuel production (e.g. agricultural residues or herbaceous energy crops) would receive higher incentives for the same percentage GHG saving. Electricity producers would in theory be able to offer a higher price for the feedstock. If however the carbon intensity of the electricity grid or the fossil fuel to be displaced is relatively low (e.g. high hydropower contribution), the biofuel producers may receive higher GHG incentives for saving more carbon by replacing fossil gasoline than by displacing low carbon electricity.

Figure 18: GHG incentives ($/MJ fuel or electricity) according to percentage GHG saved under different reference scenarios.



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Source: Compiled from JEC (2008) and Table 15 for grid intensities.

The influence of incentives on costs of production

GHG-based incentives for biofuels could influence their costs of production. Table 6 provides an indication of the potential reductions that would be represented by a GHG incentive at $\$10/tCO_2eq$ saved. GHG-based incentives expressed as in $/MJfuel above are illustrated in Table 6 as $/liter using the energy content of the fuel (22.1MJ/liter ethanol and 33.1MJ/liter biodiesel).

The cost reductions that could be delivered through GHG-based incentives at $\$10/tCO_{2eq}$ are not substantial in comparison with current volume-based subsidies. The greatest cost reduction is for a future cellulosic ethanol scenario which provides an incentive of $0.0152/liter ($0.058/US gal). US tax credits or rebates range from $0.08/gal ($0.02/liter) to $1.55/gal ($0.41/liter) for ethanol in different states and therefore the extent to which the incentives are significant in relation to the volume-based incentives differs. In the EU, excise tax credits for ethanol range from the equivalent of $0.04/liter ($0.15/gal) to $0.89/liter ($3.37/gal) and $0.22/liter ($0.83/gal) to $0.63/liter ($2.39/gal) for biodiesel (Steenblik, 2008) which is significantly greater than GHG incentives at $\$10/tCO_2eq$ would represent.

In many cases a GHG incentive would have to be substantially greater to replace even the smallest of current subsidies. For example a volume-based incentive for a typical corn-ethanol of \$0.08/gal (\$0.02/liter) would represent a GHG incentive of \$140/tCO_2eq. However this assumes that the corn ethanol GHG saving is not optimized (i.e. no use of biomass for heat and power) and that no account is taken of any soil carbon sequestration benefits. Greater reductions in production cost can be realized by improving GHG savings through adopting cost-effective technology improvements and management practices.

Table 6: Costs of production including incentives for GHG savings associated with average biofuel GHG savings

	GHG saving (gCO_2eq /MJ)*	GHG saving (gCO_2eq /liter)	Current biofuel production cost 2007 (\$/liter)[1]	GHG incentive (\$/liter) @\$10/tCO_2eq	Production cost with GHG incentive (\$/liter)	% change in biofuel production cost
Sugarcane	66.9	1478	0.27	0.0148	0.25	5.5%
Corn	6.4	141	0.74	0.0014	0.73	0.2%
Switchgrass	68.9	1522	0.66[2]	0.0152	0.64	2.3%
Cellulosic (future)	68.9	1522	0.26[3]	0.0152	0.25	5.8%
Soy Brazil	13.6	450	0.64	0.0045	0.64	0.7%
Rapeseed	44.9	1486	1.74	0.0149	1.72	0.9%

* Based on a number of existing studies; Macedo et al 2008, EBAMM (UC Berkeley, 2006), JEC (2008)

[1] Cost data from OECD-FAO (2008). Note that feedstocks costs especially oils are at a high point in 2007

[2] Assumes \$2.50/gal

[3] Assumes \$1/gal

See Annex C for an overview of costs of production for biofuels (\$/bbl oil equivalent) with GHG incentives.

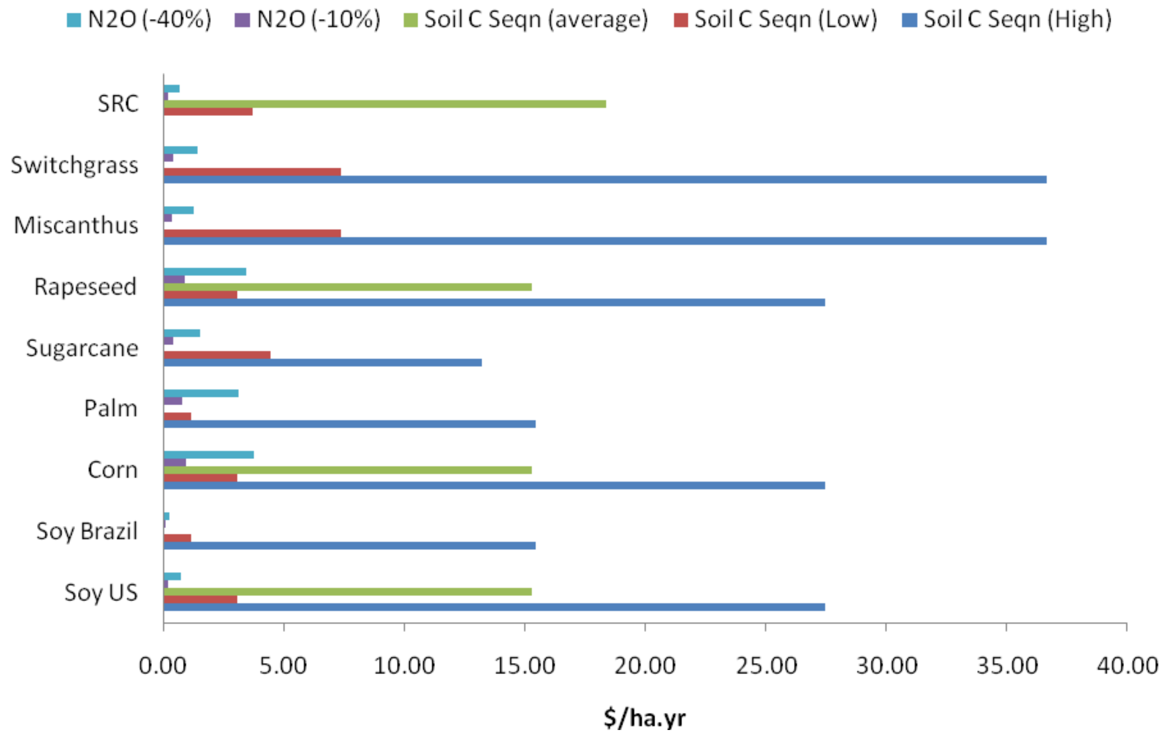
Upstream GHG Incentives

Incentives based on carbon sequestration and reduced GHG emissions could also be delivered further upstream at the feedstock cultivation stage. Figure 19 illustrates the potential size of incentives that feedstock producers might receive directly for participating in a scheme that delivers GHG emission reductions or carbon sequestration rather than relying solely on a cost pass-through from incentives aimed at biofuel producers.

The incentives relate to soil carbon sequestration and a reduction in N_2O emissions. Soil carbon sequestration rates are reported in Box 3. Where only average data is available for crops the high and low soil carbon sequestration rates are calculated as +80% or -80% of the average. Switchgrass and miscanthus are assumed to sequester up to 1tC/ha.yr (3.667tCO_2eq/ha.yr) (Anderson-Teixeira) and SRC data on soil carbon sequestration at 0.5tC/ha.yr (1.8tCO_2eq/ha.yr) (Grogan & Matthews, 2001). It should be recognized that these data are indicative and can vary substantially from site to site.

Soil N_20 emission reduction is based on fertilizer induced emissions (1.325% of applied N). Fertilizer application rates are based on data from USDA (2002 & 2006), JEC (2008), Macedo et al (2008), EBAMM model (UC Berkeley, 2006) and RFA (2007). A 10% reduction in soil N_2O emissions represents the ‘low’ case in Figure 19 and a 40% reduction represents the high case[17].

Figure 19: Incentives ($/ha) for GHG emission reduction or soil carbon sequestration in the cultivation step based on $10tCO₂eq*



For crops with relatively low baseline soil N_2O emissions such as soy, switchgrass, miscanthus, woody short rotation coppice and sugarcane, there is a greater potential for financial returns based on soil carbon sequestration. Where soil N_2O emissions are relatively high a 40% reduction could represent greater benefits in the cases where soil carbon sequestration potential is low (palm, rapeseed and corn).

These feedstock-specific results are illustrative. In reality, soil N_2O emissions vary substantially and within one feedstock such as soy a high baseline N_2O emission rate could represent greater potential reductions and vice versa. In addition, soil carbon sequestration rates vary substantially with management practices and climate and are not guaranteed across every site.

Impact of incentives on feedstock producers

To what extent could these upstream and downstream incentives for GHG saving influence the returns per hectare ($/ha) and play a role in decision-making for land allocation?

First, it is assumed that 50% of the downstream GHG incentives (received by the biofuel supplier) identified in Table 7 are passed on to the feedstock producer[18]. The incentives are

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[17] The reduction is based on soil N_2O emissions only and has not accounted for any reduction in the fertiliser production emissions.

based on the GHG saving for each feedstock type and translated into $/tonne feedstock (see Annex C). Yield data are used to calculate this incentive in $/hectare.

Secondly, the upstream incentives direct to the feedstock producer associated with soil carbon sequestration and N_2O reduction are taken into account for specific fuel chains to calculate the net economic return ($/ha).

Table 7: Potential incentives ($/ha) for farmers based on biofuel incentives based on GHG performance received by biofuel producers or suppliers: Note this represents 100% of the downstream incentive.

	tonne/ha	Source	$/ha @$10/ tCO_2eq	$/ha @$20/ tCO_2eq	$/ha @$30/ tCO_2eq
Sugarcane	87.1	Macedo et al 2008	107	213	320
Corn	8.75	EBAMM model	5	9	14
Switchgrass	13.45	EBAMM model	75	150	225
Willow	10	JEC model	51	102	153
Palm (CPO)	3.5	UK RTFO model	49	98	148
Soy US	0.48	UK RTFO model	8	16	25
Soy Brazil	0.48	UK RTFO model	2	4	5
Rapeseed	1.26	UK RTFO model	16	32	48

With a high GHG saving value and high yields per hectare sugarcane would receive substantial incentives per hectare compared with other feedstocks. Switchgrass also performs well, however the yield figure is that from the EBAMM model (UC Berkeley, 2006) and is much higher than the 5.8t/ha modeled in Illinois (Khanna *et al*, 2008). The incentive would be reduced to $32/ha (at $10/$tCO_2$eq) if the yield was as low as 5.8t/ha (assuming costs of production were the same). Palm oil with good methane management practices would also receive relatively high incentives for the biodiesel. Despite good GHG savings from soy biodiesel (assuming no land-use change) the low yields translate into low incentives on a $/hectare basis. It is interesting to note that corn, with substantially lower estimated GHG savings compared to US soy (in the models used), has only marginally higher incentives per hectare owing to the difference in yield per hectare.

Table 8 illustrates the extent to which these incentives, both transferred from the biofuel producer and those accrued directly to the feedstock producer through management practices identified in Figure 19, may play a role in influencing land allocation decisions.

Increases in net returns are greatest for sugarcane (+19%) and only marginally improved for palm and rapeseed (+2 to +5%). These results are dependent upon prices assumed for products and though although averages for 2006-2007 are used in the calculation this still represents a historically high price period for feedstocks. GHG incentives would represent a greater proportion of revenues if prices fall.

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[18] A GHG incentive enables biofuel producers to pay a certain amount per tonne for the feedstock and retain some of the incentive to lower the costs of production and increase revenues (Annex C). It is assumed unlikely that a biofuel producer or supplier would pass-through the entirety of an incentive directly to the feedstock producer.

Table 8: Comparison of baseline economic returns ($/ha) for different biofuel feedstocks and net economic returns accounting for potential GHG incentives.

	Sugar cane	Corn	Switch grass (ethanol)	Soy US	Soy Brasil	Palm	Rapeseed
Return ($/ha .yr)	354	463	-	269	186	2200	826
50% incentive pass through ($/ha.yr)	53	2.4	37.5	4.1	0.9	24.6	7.9
Soil C seqn $/ha.yr (high)	13.2	27.4	36.7	27.4	15.4	15.4	27.4*
40% decrease N_20 ($/ha.yr)	1.5	3.7	1.4	0.7	0.2	3.1	3.4
Total Incentive ($/ha.yr)	*67.7*	*33.5*	*75.6*	*32.2*	*16.5*	*43.1*	*38.7*
Return incl GHG incentives ($/ha.yr)	***421.7*** ***(+19%)***	***496.5*** ***(+7%)***	***-***	***301.2*** ***(+8%)***	***202.5*** ***(12%)***	***2243.1*** ***(+2%)***	***864.7*** ***(+5%)***

Source: Net returns collated from literature sources and include fixed and variable costs. USDA data (2008) is used for US corn and soy, CONAB data (2008) for Brazilian soy, Federal University of Rio de Janeiro data for sugarcane (published in OECD 2008), University of Newcastle (2006) statistics for rapeseed and data on palm oil synthesized from various sources.

* Assumed same as corn

The data already show the substantial returns that corn offers relative to soy (again related to high feedstock prices), however it appears unlikely that valuing GHG emissions for biofuels will redress any balance between net returns for soy relative to corn. Outside these rotational decisions, a change in crop would be necessary to move to advanced biofuel production and in this example switchgrass is used for illustrative purposes. The opportunity costs for switchgrass production are assumed to be those of corn including GHG incentives ($496.5/ha) and switchgrass costs of production are assumed to be $351/ha (Khanna et al 2008). The breakeven for a switchgrass producer is therefore $847.5/ha without GHG incentives and is reduced to $772/ha if the incentives for GHG in Table 8 are included. At 5.8t/ha this results in a breakeven cost per tonne of $133/t and at 15t/ha this breakeven cost is reduced to $51/t. The cost pass-through from the downstream GHG incentive is also significant and without it the incentives for farm-based activities would have to be $20/tCO$_2$eq to maintain these breakeven returns.

Any ethanol price that allows ethanol plants to pay more for switchgrass as a feedstock will allow them to pay more for corn (Babcock, 2007). If comparative feedstock cost is the key issue, GHG incentives could allow switchgrass to compete with corn. OECD (2008) estimates for corn prices averaged $113.2/t from 2002/3 to 2006/7 and are forecast to increase to $189.0/t in 2010/11 and decrease again to $164.6/t in 2017/18. USDA forecasts $141.7/t corn for both 2010/11 and 2017/18 (USDA, 2008). At these estimates, and without any pass-through from downstream incentives, a price of $20/tCO$_2$eq could deliver changes in land-allocation decisions for low-yielding switchgrass. These calculations do not account for any removal of subsidies in the production process which would require lower feedstock production costs to be competitive with fossil alternatives.

The calculations provide some indications of how GHG incentives could impact, land-allocation decisions. However these decisions are often made with consideration of other less obvious factors that may influence crop choice including herbicide and fertilizer programs, desired rotation, crop insurance and loan values for crops, potential crop pest problems, livestock feed needs, and erosion concerns on sloping soils.

FINDINGS

There are biofuel-specific GHG models that allow standard-setting organizations to assess key drivers of emissions within fuel chains. Focusing on improving GHG emissions through adopting better practices for site-specific situations today and into the future has to be a key goal for addressing biofuel sustainability. The net global impact has yet to be quantified robustly.

Reported well-to-wheel GHG emissions can vary according to methodological decisions, the use of different emission factors and uncertainties in data e.g. N_2O emissions from soil. Well-to-wheel GHG emissions can also vary substantially on the basis of different cultivation practices and fuels used to process biofuel. It is not possible to classify biofuel as ‘good’ or ‘bad’ on the basis of the feedstock they are developed from alone.

Soil N_2O emissions have great significance in the calculation but are not often modeled in great detail for GHG calculations. Many studies rely on default IPCC emission factors that were developed for national average reporting and not site-specific assessments. Data from tropical climates is especially lacking. Some research that has undertaken more detailed assessments have concluded that sugarcane and palm oil (without land use change) have robust GHG savings even with N_2O variability and other feedstocks with high N_2O emissions such as soy, can be considerably improved by optimizing management practices (Smeets *et al,* 2008). Uncertainty in the statistical models used for these calculations still remains and should be recognized.

Soil carbon sequestration has not been addressed within GHG calculations for biofuels and could play a valuable role in improving GHG balances. This is perhaps particularly true for lignocellulosic feedstocks (Anderson-Teixeira, 2009) as well as for improved management of residues for annual crops such as corn. There may be a trade-off between increased carbon sequestration and the value of using residues for power generation where it would be particularly attractive to replace coal for example. Site-specific circumstances will determine a sustainable harvesting strategy.

Emissions associated with some types of land use change can negate GHG savings associated with biofuels and lead to long ‘carbon payback times’. Focusing on how to improve situations where this has occurred, and indeed in developing better practices in general, such as effective co-product utilization, improving cultivation management practices to increase soil carbon sequestration and reducing N_2O emissions can play a valuable role in improving the net GHG balance of biofuels and may improve ‘carbon payback time’.

Incentives for GHG reduction ($/t$CO_2$eq) are unlikely to represent a large proportion of net returns ($/ha) at $10/t$CO_2$eq but land allocation decisions for advanced biofuel crops could be influenced by GHG incentives that reduce the breakeven returns (used as a proxy for land allocation decision). High yields per hectare and soil carbon sequestration rates are key and incentives focused on the feedstock production stage may require incentives in the range of \$20/tCO_2eq.

Current system boundaries for well-to-wheel calculations of biofuels are wide and encompass emissions in other sectors, such as those associated with producing fertilizers. These systems boundaries therefore raise the risk of double-counting emissions and emission reductions through the biofuel well-to-wheel calculation. Market mechanisms such as voluntary carbon markets and the regulatory EU ETS, are delivering incentives to reduce emissions in sectors covered within the biofuel well-to-wheel calculation. This creates difficulties in setting baselines for future emission reductions for biofuels and also risks double-counting GHG reductions in both the regulated sector (e.g. the chemical industry producing fertilizer) and the transport sector (for biofuel).

A substantial volume of work on biofuel GHG emissions has already emerged from Europe, Brasil and the USA. While harmonization of methodologies would be desirable, differences in objectives and applications ultimately limit the potential for this goal to be reached. Emerging scientific evidence is contributing to the debate on GHG emissions for biofuels and the tools and methodologies will evolve over time. However, the tools have predominantly been developed for regulatory purposes and their scope is not global. Engagement with key producer countries to obtain robust regional input data (including emission factors) would add substantial value to the body of work underway at present.

ANNEX A: GHG CRITERIA IN SUSTAINABILITY STANDARDS

The scope of these sustainability standards either covers biofuel production as a whole or focuses solely on the feedstock cultivation phase and could therefore be used for any commodity pathway e.g. palm oil in the food sector. The organizations vary in their structure or governance arrangements and their voluntary or mandatory nature but a majority are multi-stakeholder voluntary approaches. Some of the organizations are not considering specific GHG criteria within whereas others are developing methods to quantify these GHG emissions or are interested in promoting practices that can reduce GHG emissions.

Table 9: Some organizations involved in developing sustainability standards for biofuels / biofuel feedstock (not exhaustive).

	Type	Consensus based?	Membership
International/Regional			
Global Bioenergy Partnership (GBEP) (GHG and sustainability taskforces)	Voluntary	Y	International policy-makers (policy-makers)
Roundtable on Sustainable Biofuels (RSB):	Voluntary	Y	Multistakeholder
Better Sugarcane Initiative (BSI):	Voluntary	Y	Multistakeholder
Roundtable on Sustainable Palm Oil (RSPO):	Voluntary	Y	Multistakeholder
Roundtable on Responsible Soy (RTRS):	Voluntary	Y	Multistakeholder
Sustainable Agriculture Network / Rainforest Alliance	Voluntary	Y	Multistakeholder
European Committee for Standardization (CEN)	Voluntary	Y	Multistakeholder - European
International Organization for Standardization	Voluntary	Y	Multistakeholder - International
Asia-Pacific Partnership Clean Development & Climate	-	-	Multistakeholder - Limited International
IADB 'Biofuel Sustainability Scorecard'	Voluntary	N	n/a
World Bank biofuel sustainability scorecard	Voluntary	-	n/a
Brazil INMETRO	-	-	n/a
UK Renewable Transport Fuel Obligation	Mandatory reporting	N	Regulatory
German Government	Voluntary reporting	N	Regulatory
U.S.			
CA Low Carbon Fuel Standard (LCFS):	Mandatory	N	Regulatory
Sustainable Biodiesel Alliance	Voluntary	Y	Multistakeholder - national
American National Standards Institute (ANSI), Sustainable Agriculture Standard	Voluntary	Y	Multistakeholder - national
Southern Bioenergy Roadmap: SAFER	Voluntary	n/a	Multistakeholder - regional
Council on Sustainable Biomass Production:	Voluntary	Y	Multistakeholder - regional

Table 10 Selected GHG criteria from standards or guidelines agreed or in draft (*emphasis added*).

Organization	Criteria
Mandatory	
European Union	35% minimum threshold for GHG savings (compared to a defined marginal European fossil reference). From 2017 is 50%. After 2017 is 60 % for biofuels (where) production has started from 2017 onwards. Grandfathering for facilities in operation on January 2008.
UK RTFO	Requires economic operators to report on their GHG balance (and provides calculator and methodology). No cut-off. Carbon criteria in sustainability standard: • Preservation of above & below ground carbon stocks from 1st Nov 2005 - specifies no longer than 10-year carbon payback period
Germany	Requires economic operators to report on their GHG balance
Voluntary	
Global Bioenergy Partnership	(*Draft*) 'Greenhouse gas emissions' and 'Land use change (including indirect and indirect)'
Sustainable Agriculture Network addendum (3rd draft)	The farm must implement practices to diminish its emissions of greenhouse gases and increase carbon dioxide sequestration. Such practices include: • soil cover, planting of perennial plants, proper sourcing and management of fertilizers and types of fuels, management of effluent ponds and manure, use of clean technologies, improvement of energy efficiency, reduction in tillage and participation in local or regional initiatives aimed at GHG reduction and carbon dioxide sequestration.
Roundtable on Sustainable Biofuels - Version Zero.	Producers and processors shall reduce GHG emissions from biofuel production over time. Emissions shall be estimated via a consistent approach to lifecycle assessment, with *system boundaries from land to tank* Preferred methodology : • Functional unit shall be CO_2 equivalent (in kg) per Giga Joule [$kgCO_2equ/GJ$] • The greenhouse gases covered shall include CO_2, N_2O and CH_4. The most recent 100-year time horizon Global Warming Potential values and lifetimes from the IPCC shall be used[19]. • GHG emissions from direct land use change shall be estimated using IPCC Tier 1 methodology and values. Better performance than IPCC default values can be proven through models or field experiments • GHG emissions from *indirect* land use change, i.e. that arise through macroeconomic effects of biofuels production, shall be

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[19] See Table 11 for IPCC Global Warming Potentials

Organization	Criteria
	minimized e.g. maximizing use of waste and residues as feedstocks; using marginal, degraded or previously cleared land; improvements to yields; and efficient crops; International collaboration to prevent detrimental land use changes; and avoiding the use of land or crops that are likely to induce land conversions resulting in emissions of stored carbon.
Roundtable on Sustainable Palm Oil	No explicit GHG criteria Requires collection and monitoring of key data including fertilizer and fossil fuel use for compliance with some criteria Principle 4 contains criteria that are designed to promote optimal yields which will influence the GHG balance. Criterion 5.4 Efficiency of energy use and use of renewable energy is maximized.
Roundtable on Responsible Soy (RTRS) *DRAFT*	No explicit GHG criteria. Includes requirement for: • Maintenance of carbon at sites with high capacities for above or below ground carbon • Restoration of degraded lands for soy cultivation
ANSI *DRAFT* (Note that this original draft has been set-aside but is used for illustrative purposes)	Agricultural Production Plan required that covers **carbon sequestration and storage** and will describe the approach for increasing the level of terrestrial carbon stabilization, sequestration and storage resulting from agricultural production processes including, for example, maintenance of agro-ecosystem health, planting of buffer zones, and planted wind-breaks or through off-site carbon stabilization strategies. Energy Resource Management principle: Producer 'is required to monitor energy consumption, pursue increased energy efficiency in the production, handling and transport of agricultural products, and ***calculate the energy and associated greenhouse gases per agricultural production unit***.' Producer shall calculate the Energy Efficiency Index for the agricultural product (EEIP), as well as the associated Greenhouse Gas Index (GHGIP). The Producer shall provide the EEIP and associate GHGIP to downstream recipients who are conformant handlers.
Sustainable Biodiesel Alliance	Sustainable biodiesel results in net GHG emissions reductions compared to fossil fuels when analyzed via a life-cycle assessment. Fossil energy used in growing, transporting and processing biodiesel must be considered. Converting land from wilderness or grasslands to plant biodiesel feedstock crops also releases GHG and is not sustainable.
Inter-American Development Bank (IDB)	A color-coded sustainability scorecard used for evaluating biofuel funding applications is in draft form and covers fuel-chain GHG savings as well as crop rotation details. http://www.iadb.org/scorecard/scorecard.cfm?language=English Fuel-chain GHG savings at: • 60% or greater = excellent (deep green) • 35% to 60% = good (light green) • 0% to 35% = satisfactory (light yellow) • Less than 0% = unsatisfactory (red)
World Bank/ WWF *DRAFT scorecard*	Life cycle GHG assessment concludes that the project is...carbon negative (including emissions from direct and indirect land use change).

ANNEX B: EMISSION FACTORS

Emission factors are required in order to obtain the CO_2eq emissions associated with a specific input or activity. The emissions factors are based on the fundamental background assumption of different global warming potentials.

Table 11 Global Warming Potentials

	CH_4	N_20	CO_2
2nd IPCC Assessment report	21	310	1
3rd IPCC Assessment report	23	296	1
4th IPCC Assessment report	25	298	1

Table 12: Emissions factors in use within different tools and studies specifically related to N_20 emissions from soil.

	Total fraction on N resulting in N_2O (kg N_2O-N/ kg N)	Effective emissions factor ($kgCO_2eq$/ kg N)
GREET baseline	0.015 (1.5% of N)	6.977
UK RTFO baseline	0.01325 (1.325% of N)	6.163[1]
JEC baseline	From EU soil emissions model	-
North Energy baseline direct & indirect (IPCC)[a]	0.0132 (1.3% of N)	6.140
North Energy Low[a]	0.0058 (0.6% of N)	2.698
North Energy High[a]	0.0348 (3.5% of N)	16.187
EBAMM baseline direct & indirect emissions[b]	0.015 (1.5% of N)	6.977
EBAMM Low[b]	0.003 (0.3% of N)	1.391
EBAMM High[b]	0.055 (5.5% of N)	25.420

[1] Except for soybeans – treated as special case due to high N content of residue.

a) Based on calculations in Mortimer et al which were based on IPCC 2006 guidelines.
b) Based on data contained in EBAMM which were based on IPCC 2006.

Evaluation of Total Soil N_2O Emissions with the IPCC Tier 1 Approach

ANNEX A from North Energy Associates (2008) Support for the Review of the Indirect Effects of Biofuels[20]

Using the Tier 1 approach outlined in IPCC 2006, the d**irect soil N_2O emissions from the application of N fertiliser** has a likely value of 0.0100 kg N_2O-N/kg N (1.0%) or 0.0157 kg N_2O/kg N, and a range of 0.0030 – 0.0300 kg N_2O-N/kg N (0.3 – 3.0%) or 0.0047 – 0.0470 kg N_2O/kg N (IPCC 2006, Table 11.1).

Indirect soil N_2O emissions are assumed to arise from volatilisation and leaching/run-off. The pathway to N_2O from the atmospheric deposition of N through volatilisation involves the conversion of ammonia (NH_3) and oxides of nitrogen (NO_x). The IPCC 2006 Tier 1 approach assumes that the amount of N_2O emitted indirectly through volatilisation depends on the product of the fraction of N fertiliser that volatilises as NH_3 and NO_x and the emissions factor for N_2O emissions from the atmospheric deposition of N on soils (IPCC 2006, Equation 11.9). The fraction of N fertiliser that volatilises as NH_3 and NO_x has a likely value of 10% and a range of 3 – 30% (IPCC 2006, Table 11.3). **The emissions factor for N_2O emissions from the atmospheric deposition of N on soils** has a likely value of 0.0100 kg N_2O-N/kg N and a range of 0.0020 – 0.0500 kg N_2O-N/kg N (IPCC 2006, Table 11.1). This results in indirect soil N_2O emissions from volatilisation with a likely value of 0.0010 kg N_2O-N/kg N (0.1%) or 0.0016 kg N_2O/kg N, and a range of 0.0001 – 0.0055 kg N_2O-N/kg N6 (0.01 – 0.55%) or 0.0002 – 0.0086 kg N_2O/kg N.

The IPCC 2006 Tier 1 approach assumes that the amount of N_2O emitted indirectly through leaching/run-off depends on the product of the fraction of N fertiliser that leaches or runs off and the emissions factor for N_2O emissions from such leaching and run-off (IPCC 2006, Equation 11.10). The fraction of N fertiliser that leaches or runs off has a likely value of 30% and a range of 10 – 80% (IPCC 2006, Table 11.3). T**he emissions factor for N_2O emissions from leaching and run-off** has a likely value of 0.0075 kg N_2O-N/kg N and a range of 0.0005 – 0.0250 kg N_2O-N/kg N (IPCC 2006, Table 11.3). This results in indirect soil N_2O emissions from leaching/run-off with a likely value of 0.0023 kg N_2O-N/kg N (0.23%) or 0.0035 kg N_2O/kg N, and a range of 0.0004 – 0.0088 kg N_2O-N/kg N (0.04 – 0.88%) or 0.0006 – 0.0138 kg N_2O/kg N.

Consequently, the total soil N_2O emissions using the IPCC 2006 Tier approach has a likely value of 0.01325 kg N_2O-N/kg N (1.33%) or 0.0208 kg N_2O/kg N, and a range of 0.0058 – 0.0348 kg N_2O-N/kg N (0.58 -3.48 %) or 0.0091 – 0.0547 kg N_2O/kg N.

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[20] Available from http://www.renewablefuelsagency.org/reportsandpublications/reviewoftheindirecteffectsofbiofuels/consultancystudies.cfm

Table 13: Model, effect values and constant for the Stehfest and Bouwman model (2006) used as an alternative to the IPCC default to calculate direct N2O emissions from agricultural fields and natural vegetation.

$$\log(N_{emission}) = A + \sum_{i=1}^{n} E_i \qquad (1)$$

Agricultural fields		Natural vegetation	
<u>*Constant (c)*</u>	-1.5160	<u>*Constant (c)*</u>	-2.8900
<u>*Effect values (ev)*</u>		<u>*Effect values (ev)*</u>	
N application rate		*N application rate*	
Variable	0.0038 x N application rate in kg N ha^{-1} yr^{-1}	N/a	N/a
Soil organic C content (%)		*Soil organic C content (%)*	
<1	0	<1	0
1-3	0.0526	1-3	0.6683
>3	0.6334	>3	1.0918
Soil pH		*Soil pH*	
<5.5	0	<5.5	0
5.5-7.3	-0.0693	5.5-7.3	-0.2750
>7.3	-0.4836	>7.3	-2.4179
Soil texture		*Soil texture*	
Coarse	0	N/a	N/a
Medium	-0.1583	N/a	N/a
Fine	0.4312	N/a	N/a
Climate		*Climate*	
Temperate_C	0	N/a	N/a
Temperate_O	0.0226	N/a	N/a
(sub)Tropical	0.6117	N/a	N/a
Tropical	-0.3022	N/a	N/a
Crop type		*Vegetation type*	
Cereals[a]	0	Coniferous[b]	0
Grass	-0.3502	Deciduous[b]	0.0115
Legume	0.3783	Grass	-0.7941
Other[d]	0.4420	Rain forest	0.4995
Wetland rice	-0.8850	Savannah	-0.6881
None	0.5870	Tropical dry forest	-0.5811
		Desert[c]	N/a
Soil drainage		*Soil drainage*	
N/a	N/a	Poorly drained	0
N/a	N/a	Well-drained	-1.0462
Bulk density (g cm^{-3})		*Bulk density (g cm^{-3})*	
0-1	0.9941	N/a	N/a
1-1.25	-0.3786	N/a	N/a
>1.25	-0.8597	N/a	N/a
Length of experiment (year)		*Length of experiment (year)*	
1	1.9910	1	3.6120

[a] The category cereals excludes maize, which is in ‘other crops’.

[b] The measurements for temperate coniferous and deciduous forests stem from areas with high N deposition (≥ 10 kg N ha^{-1} yr^{-1}). For coniferous and deciduous forests with low N deposition (<10 kg N ha^{-1} yr^{-1}) we correct the N_2O emissions calculated by the model. The correction factor is the ratio of N deposition for forests with low N deposition to 10 kg N ha^{-1} yr^{-1}.

[c] For deserts, no effect values are given. Instead assumed a default N_2O emission of 0.1 kg N_2O-N ha^{-1} yr^{-1} based on Bouwman *et al.* (1993).

[d] This category includes sugar cane. Although this crop has some biological N fixation, it is not usually classified as a legume. The rate of N fixation in sugar cane production is much lower than that in leguminous crops, but higher than that of other non-leguminous crops (see further the main text. An analysis of the results of N_2O emission measurements that are included in the dataset of Stehfest and Bouwman (2006), from which the statistical model is calculated, showed that the N_2O emissions are similar to those of ‘other crops’.

Table 14: Electricity emission factors from various studies and methodologies.

Emission factors used in studies may vary substantially. For example the use of electricity within a conversion process may have a large emission factor for coal-based electricity systems or a lower coefficient for an electricity grid that is dominated by nuclear, hydroelectric or renewable energy. The emission factor can have a significant influence on the GHG balance depending on its value and the size of the contribution it makes to the supply chain. The choice of using ***marginal*** or ***average*** values for the fossil reference system also influences the final result.

Country	g CO_2 / kWh	kg CO_2 / MJ	Note	Reference
Brazil (low)	78.11	0.0220	Brazil CDM data in 2006 for the emissions related to power generation in Southeast-Midwest Region	Macedo et al (2008)
Brazil (high)	180	0.0500		Macedo et al (2008)
Brazil	121	0.0340	Average build margin	CDM report Feb 09 http://www.iges.or.jp/en/cdm/report_ers.html
Brazil	540	0.1500	Operating margin (average)	CDM report Feb 09 http://www.iges.or.jp/en/cdm/report_ers.html
Brazil	338	0.0940	Combined build & operating average	CDM report Feb 09 http://www.iges.or.jp/en/cdm/report_ers.html
Indonesia	776.2	0.2160	Build margin	UK RFA (2008)
Indonesia	753	0.2090	Build margin (average)	CDM report Feb 09 http://www.iges.or.jp/en/cdm/report_ers.html
Indonesia	912	0.2530	Operating (average)	CDM report Feb 09 http://www.iges.or.jp/en/cdm/report_ers.html
Indonesia	832	0.2310	Combined build & operating average	CDM report Feb 09 http://www.iges.or.jp/en/cdm/report_ers.html
United Kingdom	472.9	0.1310	Marginal average	UK RFA (2008)
United States	574.7	0.160	Marginal average	UK RFA (2008)
United States	248.4	0.0690	Operating - 1% oil-fired, 14.9% NG, 53.8% Coal, 18% Nuclear, and 12.3% other (hydro, wind, etc), and accounting for an 8% distribution loss.	EBAMM

Table 15: US electricity emission factors by eGRID region.

	kgCO_2eq/ Mbtu	kgCO_2eq/ MJ
NEWE (New England)	111.1998	0.117
NYCW (New York City)	104.1627	0.110
NYLI (Long Island, NY)	180.9885	0.191
NYUP (Upstate NY)	93.536	0.099
RFCE (Mid Atlantic)	146.4199	0.154
SRVC (Virginia/Carolina)	149.5563	0.158
SRTV (Tennessee Valley)	201.8366	0.213
SRMV(Lower Mississippi)	135.8106	0.143
SRSO (SE US, Gulf Coast)	197.4664	0.208
FRCC (Most of Florida)	168.9224	0.178
RFCM (Most of Michigan)	206.3762	0.218
RFCW (Ohio Valley)	205.4303	0.217
MROE (Eastern WI)	242.3268	0.256
SRMW (Middle Mississippi)	244.6227	0.258
MROW (Upper Midwest)	242.0177	0.255
SPNO (KS-Western MO)	262.0664	0.276
SPSO (TX Panhandle-OK)	221.0419	0.233
ERCT (Most of TX)	176.7271	0.186
RMPA (CO-Eastern WY)	249.8624	0.264
AZNM (Southwest US)	175.0721	0.185
NWPP (Northwest US)	120.0487	0.127
CAMX (Southwest Coast)	95.1978	0.100
HIMS (HI excluding Oahu)	191.0109	0.202
HIOA (Oahu Island)	232.2375	0.245
AKMS (Most of Alaska)	66.5416	0.070
AKGD (So/Central Alaska)	161.6548	0.171
National Average	175.5362	0.185

Source: http://www.energystar.gov/ia/business/evaluate_performance/Emissions_Supporting_Doc.pdf

Table 16: Fossil fuel reference data.

Fossil fuel reference data are used in order to calculate the GHG savings associated with biofuels. The emissions associated with the production of bioethanol are compared with a gasoline reference and emissions from biodiesel production are compared against a diesel reference. The following calculation is detailed within the RFA (2008) Technical Guidance[21]. Potential differences in drivetrain efficiencies have not been accounted for.

$$\text{Fuel chain \% GHG saving} = \frac{\text{(carbon intensity of fossil reference – carbon intensity of biofuel)}}{\text{Carbon intensity of fossil reference}} \text{ X 100}$$

Fossil fuel reference	Total (WTW (gCO_2 eq/MJ)	Source
European marginal average gasoline	85	http://ies.jrc.ec.europa.eu/uploads/media/WTT%20App%202%20v30%20181108.pdf Middle eastern crude is assumed as the marginal fuel
European marginal average diesel	86	As above
CARBOB gasoline	96.9	http://www.arb.ca.gov/fuels/lcfs/011209lcfs_ulsd.pdf
ULSD (CA-GREET)	95.3	http://www.arb.ca.gov/fuels/lcfs/011209lcfs_ulsd.pdf
EBAMM conventional gasoline (derived from GREET	85	EBAMM_1_1Jan (UC, Berkeley 2006)
EBAMM diesel derived from GREET	91	EBAMM_1_1Jan (UC, Berkeley 2006)
Brazil	82	Macedo et al 2008

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[21] The RFA calculation assumes that there is no efficiency penalty of combustion for biofuels as the guidance is intended for low blend biofuels.

Table 17: Fertilizer production emission factors.

	GREET[a]	IEA Task 38 report[b]	JEC 2007	Macedo et al 2008
(kgCO_2eq/kg N)				
Ammonia	-			-
European average		2.3		
Best Technology		2		
USA		1.5		
Ammonium Nitrate (EU average)	-	6.8		-
Best technology		3		
Urea (EU average)	-	1.33		-
Best technology		0.91		
NPK (Urea / TSP / MOP) EU average mix	-	2.60		-
Composite nitrogen	4.0			3.97
Single superphosphate (SSP) as P_2O_5	-	0.095		-
Triple superphosphate (TSP) as P_2O_5	-	0.354		-
Rock phosphate as P_2O_5	-	0.354		-
Mono ammonium phosphate (MAP) as P_2O_5	-	0.596		-
Unidentified phosphate	1.6	-		1.3
Potassium Chloride	0.71	0.33		0.71
Lime ($CaCO_3$)	0.07		0.124	0.1
Magnesium	-			-

a) Used by EBAMMv1.1 in modeling data

b) Wood & Cowie (2004) Wide range data – figures here are identified as ‘Kongshaug 1998’ in the report.

Note: Emissions coefficients for fertilizers can be given on the basis of individual elements (i.e. P, K etc) or of their “oxides” (i.e. P_20_5). The latter is the standard way in which fertilizer application rates are reported and care should be taken to make sure the correct emission factor is used.

Table 18: Transport energy intensity based on various studies

	MJ/ t-km	Emission factor	gCO_2eq/ t-km	Reference
Truck OECD N America	1.46	86.4	126.144	RFA (2008) cites WBSCD/IEA (2004) Transport spreadsheet model - Mobility 2030 Project. IEA/OECD and WBSCD. Uses JEC emission factors
Truck OECD Europe	1.53	86.4	132.192	
Truck Latin America	1.8*	86.4	155.52	
Rail OECD N America	0.19	87.3	16.587	
Rail OECD Europe	0.38	87.3	33.174	
Rail Latin America	0.24	87.3	20.952	

* At 33MJ/liter diesel this is approximately 0.03liters/t-km. This value varies substantially and can substantially impact emissions from some fuel chains. In Brazil, efficient loading in the center-south can use only 0.0125l/t_{eth}-km which is 0.4MJ/t_{eth}-km (Macedo, *pers comm*.)

Table 19: GHG emissions associated with transporting different products within the European JEC well-to-wheels study.

	Distance (km)	tkm/MJ product	gCO2/MJ product
Truck			
Wood chips	50	0.004	0.33
Sugar cane	20	0.004	0.30
Rapeseed	50	0.002	0.17
Soybeans	50	0.126	0.74
Marine			
Soybeans	5500	1.833	32.59
Sugarcane ethanol	5500	0.380	4.11
Palm oil	5500	0.283	2.83

Source: Data supplied by Jean-Francois Larive (CONCAWE)

ANNEX C: BACKGROUND DATA & CALCULATIONS

Figure 20: A breakdown of the contribution to total *cultivation* emissions for corn in the EBAMM model (gCO_2eq/ha)

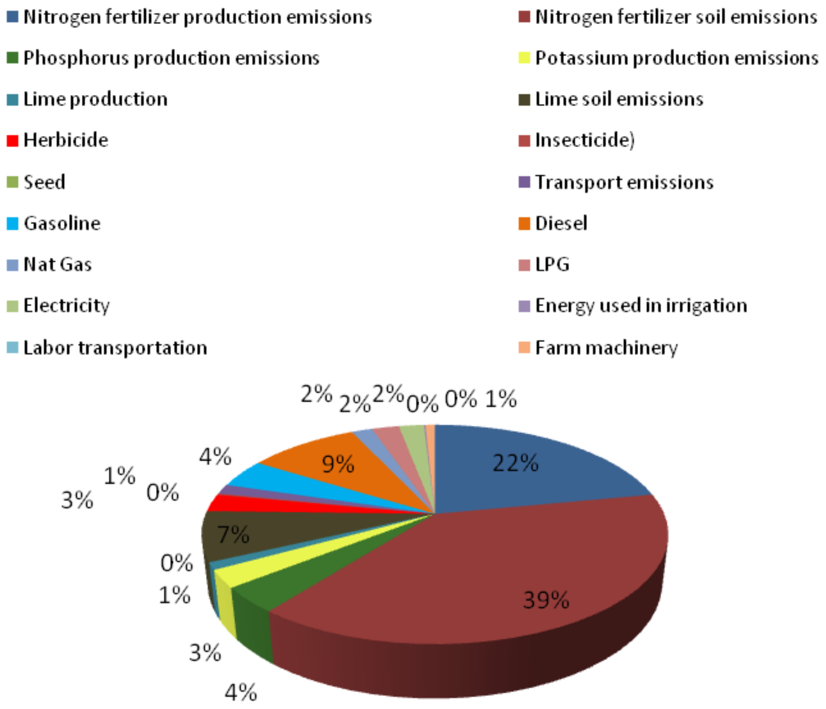


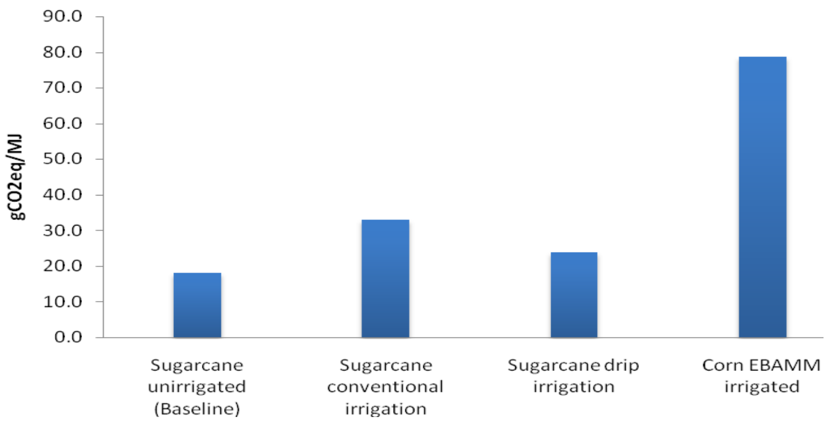
Table 20: Irrigation efficiency background data. Illustrative for Latin America.

Irrigation System	Energy requirements (kWh/ha.yr)		GHG emissions (kgCO_2eq/ha.yr)[1]	
	Theoretical Potential	Actual Efficiency	Theoretical Potential	Actual Efficiency
Sprinkler				
Conventional	1897	2846	1174	1762
Center Pivot	3612	n/a	2236	
Localized				
Drip	765	1084	474	671
Microsprayers	957	1355	592	839

[1] Assumes grid emission factor of 0.62tCO_2eq/MWh

Source: Alfaro & Marin (1991)

Figure 21: Comparison of GHG balance (gCO_2/MJ) for irrigated sugarcane compared to un- irrigated sugarcane and irrigated corn. Energy requirements vary and therefore this is illustrative only.



Note: Average grid intensity for electricity of sugarcane irrigation assumed as $0.832tCO_2eq/MWh$ (Indonesia marginal see Table 14 Annex B)

Source: Macedo et al 2008, Alfaro & Marin (1991), EBAMM v1.1 model (UC Berkeley, 2006).

Table 21: Background data on co-product treatment.

The data in the tables below support Figure 7 in the main report.

	Co-product credit (gCO_2eq/MJ ethanol)	Net GHG emissions (gCO_2eq/MJ ethanol)	GHG savings %	Assumption
Sugarcane (energy: LHV bagasse)	-1.8	18.9	77.8%	176MJ bagasse /t cane;
Sugarcane (displacement: electricity)	-2.5	18.2	78.6%	High grid emission from Macedo 2008
Corn (energy of co-products)	-65.6	35.1	58.8%	65% of emissions attributed to co-products
Corn EBAMM (displacement: various)	-24.6	76.0	10.6%	From EBAMM. Various displacement
Corn (market value)	-21.5	79.1	6.9%	21% of emissions attributed to co-product
Cellulosic (displacement)	-5.0	9.8	88.5%	From EBAMM: avoided $105.8gCO_2eq$/liter ethanol
Corn CA-GREET (displacement)	-11.5	64.8	23.8%	From CA-GREET includes animal feed & fertilizer

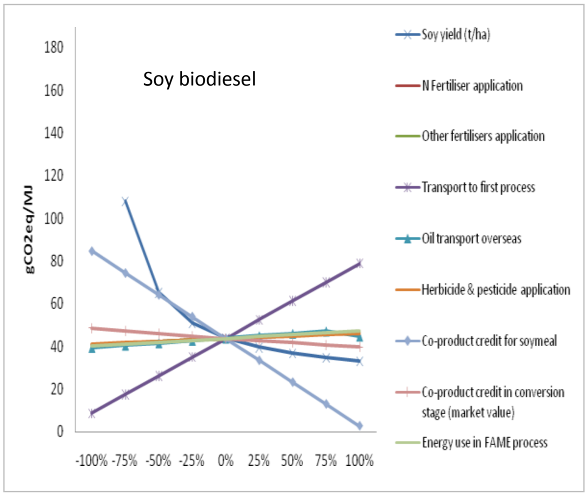
Data on co-product treatment	
Corn market value	
468	$/t Ethanol market value (2007-2008 USA spot) FO Lichts
100	$/t DDGS (2002-2004 wholesale)
21%	Market value
368	$/t Ethanol market value (2006-2008 Brazil ex-dist anhydrous) FO Licht, 2009.
145	$/t DDGS Jan 09 top price (USDA AMS www.ams.usda.gov/mnreports/sj_gr225.txt)
Sugarcane based on market price of electricity export	
9.70	$/MWh for industrial customers 2008 (from http://rad.aneel.gov.br/reportserverSAD?%2fSAD_REPORTS%2fSAMP_TarifaMedCConsumoRegiao&rs:Command=Render)
103.50	$MWh across all tariffs 2008 (reference as above)
96.00	MJ electricity /t cane
0.27	MWh / t cane
24.22	$/t cane for electricity
368	$/t = price of ethanol (FO Licht 2009)
9.20	t cane per tonne ethanol
223	$/tonne ethanol (electricity)
61%	emissions attributed to electricity
Calculation for allocation based on energy content (Corn dry mill)	
396.91	liters ethanol/ t corn
8414.40	MJ ethanol / t corn
0.50	t DDGS/t ethanol
21.79	MJ/kg DDGS
0.50	t ethanol/ t corn
251.65	kg DDGS/t corn
5483.51	MJ DDGS/t corn
65%	emissions attributed to co-products

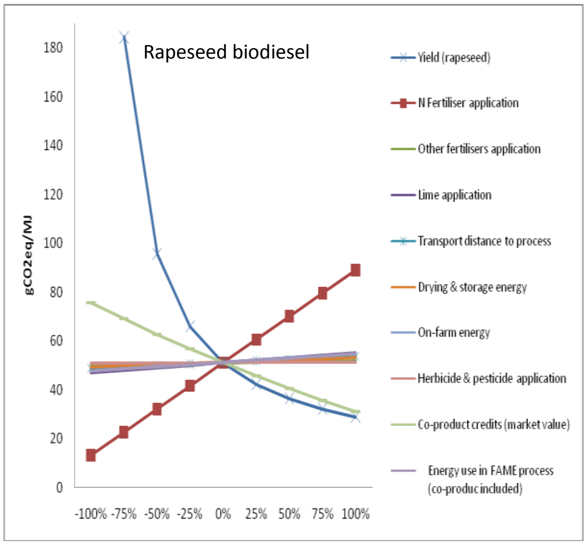
Table 22: A comparison of co-product treatment methods for corn and sugarcane.

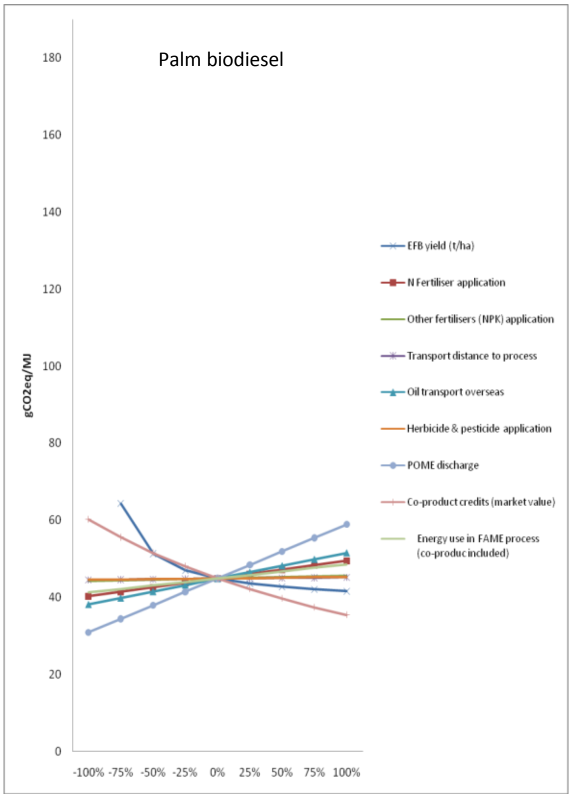
	20% + ethanol price	20% - ethanol price	20% + co-product price	20% - co-product price
Corn dry mill				
Energy	65%	65%	65%	65%
Market value	18%	27%	26%	17%
Sugarcane[1]				
Market value (electricity)	61%	61%	73%	48%

[1] Based on Macedo (2008) calculations for 2006 scenario

Figure 22: Sensitivity analysis for biodiesel fuel chains

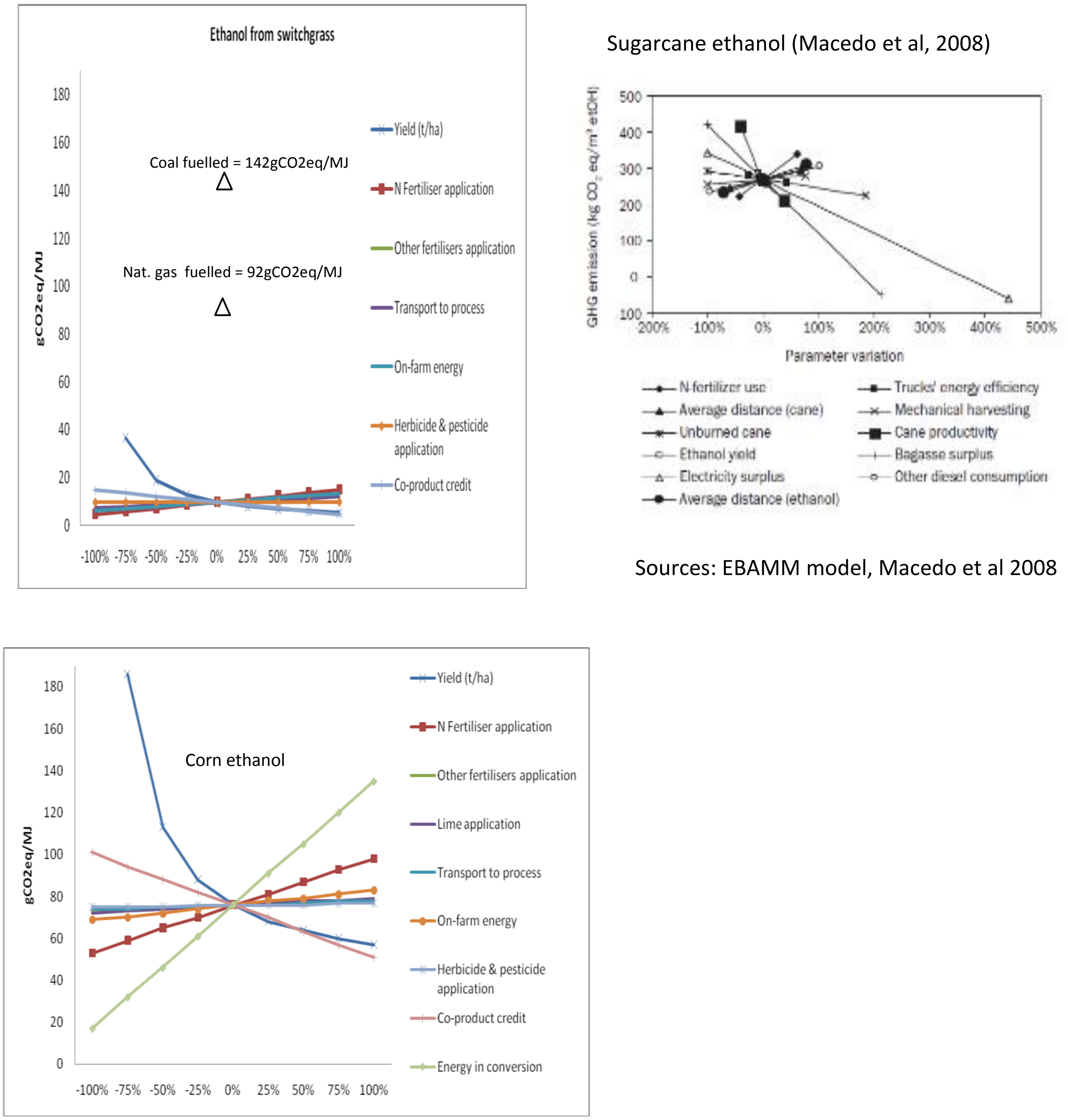






Sources: Based on RFA (2008a) and JEC (2008)

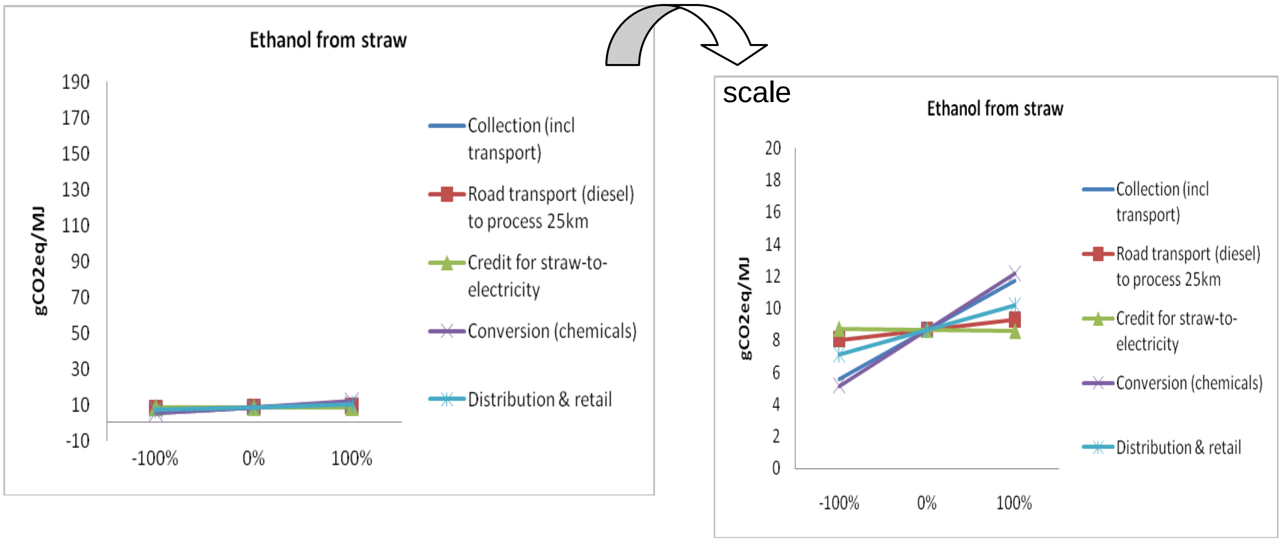
Figure 23: Sensitivity analysis of ethanol fuel chains

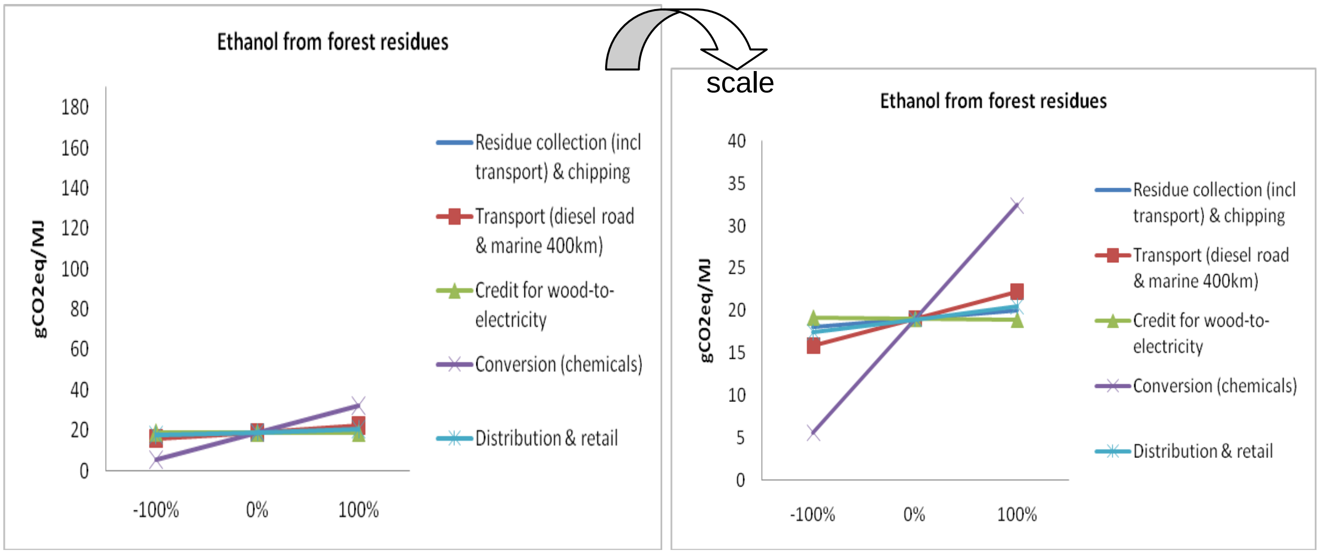

Ethanol from switchgrass

Sugarcane ethanol (Macedo et al, 2008)

Corn ethanol

Sources: EBAMM model, Macedo et al 2008





Source: Based on JEC, 2008

Table 23: Incentives per tonne of feedstock based on the GHG saving of the biofuel from that feedstock.

This table provides the background data for figures in the main body of the paper. It assumes that the incentive provided on the basis of the GHG saved is passed through 100% to the feedstock producer/supplier.

	tCO_2eq saved/MJ	Source	MJ biofuel from 1 tonne feedstock	$/tonne (f) @ $10 /$tCO_2$eq	$/tonne (f) @ $20/ tCO_2eq	$/tonne (f) @ $30/ tCO_2eq
Sugarcane	0.000067	Macedo 2008	1830	1.22	2.45	3.67
Corn	0.000006	CA-GREET	8395	0.54	1.07	1.61
Switchgrass (ethanol)	0.000069	EBAMM	8098	5.58	11.15	16.73
Forest residue (ethanol)	0.000063	JEC model	8098	5.10	10.20	15.31
Palm	0.000063	JEC model	22389	14.06	28.12	42.17
Soy US	0.000061	CA-GREET	28003	17.12	34.25	51.37
Soy Brazil	0.000014	UK RTFO model	28003	3.81	7.62	11.43
Rapeseed	0.000045	JEC model	28003	12.57	25.14	37.70

Figure 24: Sensitivity of breakeven prices ($/t switchgrass) against the carbon incentive and the opportunity cost for switchgrass vs corn.

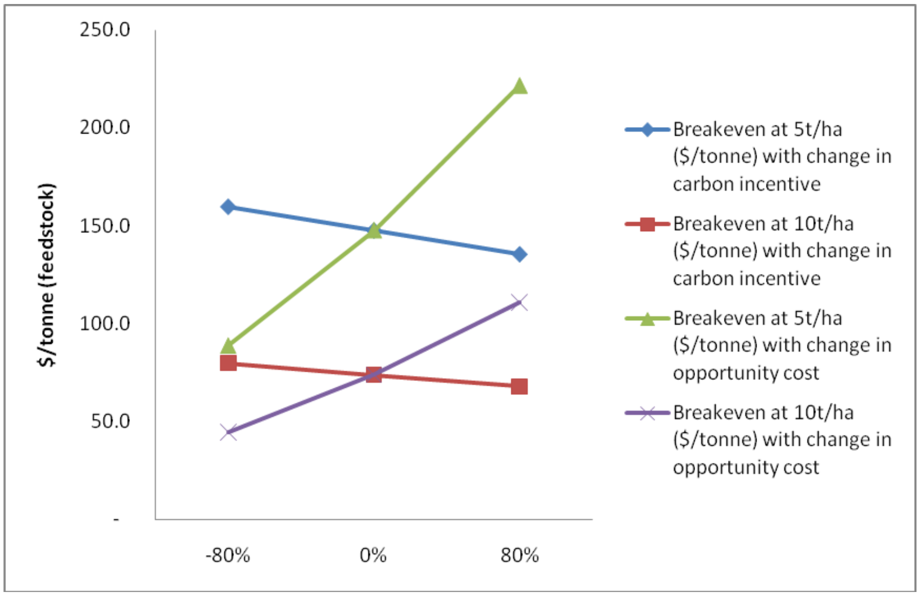
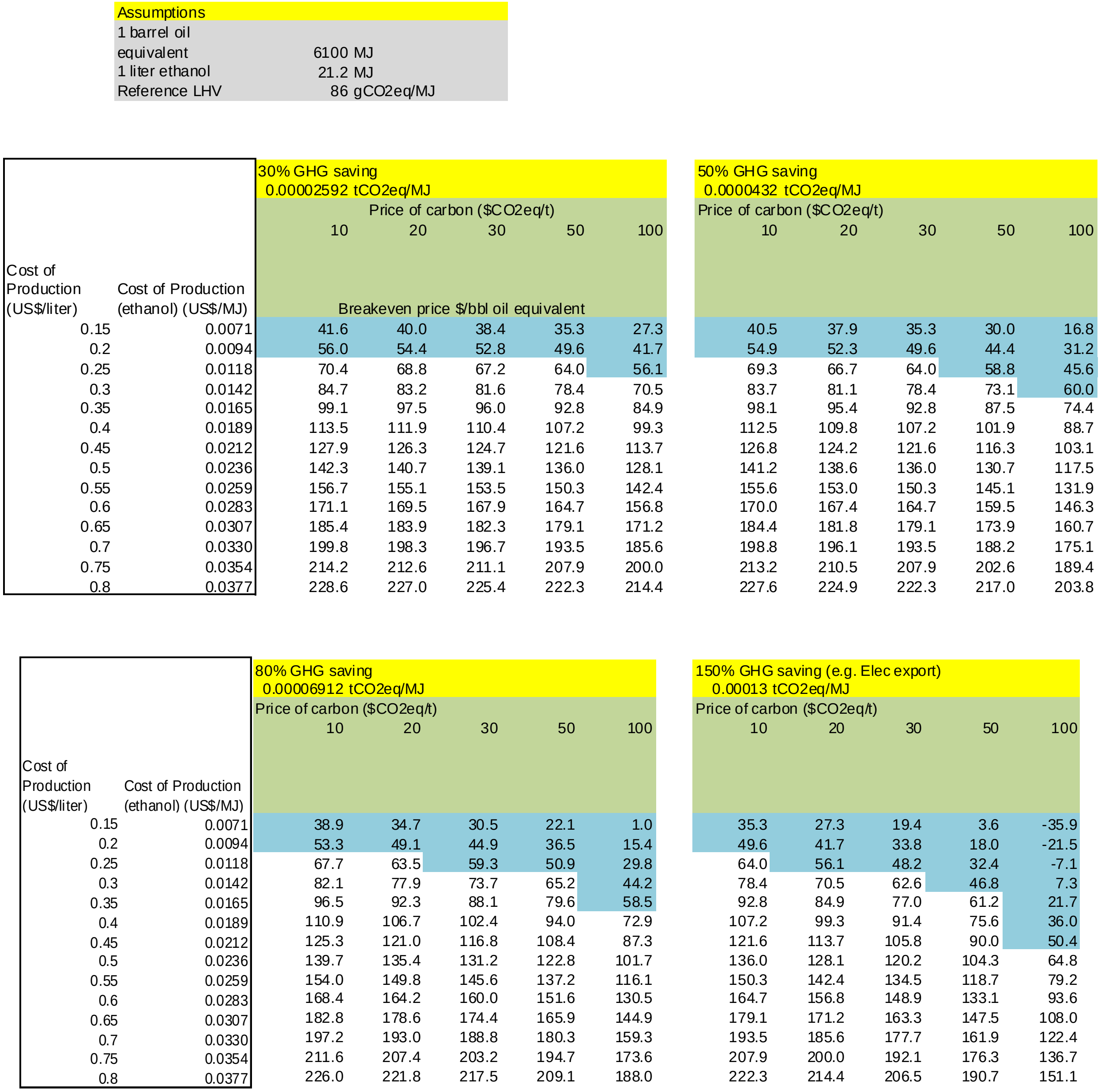


Figure 25: Sensitivity analysis of GHG incentives and costs of production in $/bbl oil equivalent.



Assumptions	
1 barrel oil equivalent	6100 MJ
1 liter ethanol	21.2 MJ
Reference LHV	86 gCO2eq/MJ

		30% GHG saving 0.00002592 tCO2eq/MJ					50% GHG saving 0.0000432 tCO2eq/MJ				
		Price of carbon ($CO2eq/t)					Price of carbon ($CO2eq/t)				
		10	20	30	50	100	10	20	30	50	100
Cost of Production (US$/liter)	Cost of Production (ethanol) (US$/MJ)	Breakeven price $/bbl oil equivalent									
0.15	0.0071	41.6	40.0	38.4	35.3	27.3	40.5	37.9	35.3	30.0	16.8
0.2	0.0094	56.0	54.4	52.8	49.6	41.7	54.9	52.3	49.6	44.4	31.2
0.25	0.0118	70.4	68.8	67.2	64.0	56.1	69.3	66.7	64.0	58.8	45.6
0.3	0.0142	84.7	83.2	81.6	78.4	70.5	83.7	81.1	78.4	73.1	60.0
0.35	0.0165	99.1	97.5	96.0	92.8	84.9	98.1	95.4	92.8	87.5	74.4
0.4	0.0189	113.5	111.9	110.4	107.2	99.3	112.5	109.8	107.2	101.9	88.7
0.45	0.0212	127.9	126.3	124.7	121.6	113.7	126.8	124.2	121.6	116.3	103.1
0.5	0.0236	142.3	140.7	139.1	136.0	128.1	141.2	138.6	136.0	130.7	117.5
0.55	0.0259	156.7	155.1	153.5	150.3	142.4	155.6	153.0	150.3	145.1	131.9
0.6	0.0283	171.1	169.5	167.9	164.7	156.8	170.0	167.4	164.7	159.5	146.3
0.65	0.0307	185.4	183.9	182.3	179.1	171.2	184.4	181.8	179.1	173.9	160.7
0.7	0.0330	199.8	198.3	196.7	193.5	185.6	198.8	196.1	193.5	188.2	175.1
0.75	0.0354	214.2	212.6	211.1	207.9	200.0	213.2	210.5	207.9	202.6	189.4
0.8	0.0377	228.6	227.0	225.4	222.3	214.4	227.6	224.9	222.3	217.0	203.8

		80% GHG saving 0.00006912 tCO2eq/MJ					150% GHG saving (e.g. Elec export) 0.00013 tCO2eq/MJ				
		Price of carbon ($CO2eq/t)					Price of carbon ($CO2eq/t)				
Cost of Production (US$/liter)	Cost of Production (ethanol) (US$/MJ)	10	20	30	50	100	10	20	30	50	100
0.15	0.0071	38.9	34.7	30.5	22.1	1.0	35.3	27.3	19.4	3.6	-35.9
0.2	0.0094	53.3	49.1	44.9	36.5	15.4	49.6	41.7	33.8	18.0	-21.5
0.25	0.0118	67.7	63.5	59.3	50.9	29.8	64.0	56.1	48.2	32.4	-7.1
0.3	0.0142	82.1	77.9	73.7	65.2	44.2	78.4	70.5	62.6	46.8	7.3
0.35	0.0165	96.5	92.3	88.1	79.6	58.5	92.8	84.9	77.0	61.2	21.7
0.4	0.0189	110.9	106.7	102.4	94.0	72.9	107.2	99.3	91.4	75.6	36.0
0.45	0.0212	125.3	121.0	116.8	108.4	87.3	121.6	113.7	105.8	90.0	50.4
0.5	0.0236	139.7	135.4	131.2	122.8	101.7	136.0	128.1	120.2	104.3	64.8
0.55	0.0259	154.0	149.8	145.6	137.2	116.1	150.3	142.4	134.5	118.7	79.2
0.6	0.0283	168.4	164.2	160.0	151.6	130.5	164.7	156.8	148.9	133.1	93.6
0.65	0.0307	182.8	178.6	174.4	165.9	144.9	179.1	171.2	163.3	147.5	108.0
0.7	0.0330	197.2	193.0	188.8	180.3	159.3	193.5	185.6	177.7	161.9	122.4
0.75	0.0354	211.6	207.4	203.2	194.7	173.6	207.9	200.0	192.1	176.3	136.7
0.8	0.0377	226.0	221.8	217.5	209.1	188.0	222.3	214.4	206.5	190.7	151.1

Table 24: Overview of risks associated with nitrogen loss in cultivation

	High risk	Lower risk
Leaching of nitrate-N	Light textured soils	
	UAN solutions	Anhydrous ammonia (82%N)
	Ammonium nitrate	
Dentrification of nitrate-N	Heavy & poorly drained soil	
	Compacted soil	Add nitrification inhibitors
	UAN solutions (28%N)	Anhydrous ammonia
	Ammonium nitrate	Sidedress N application instead of pre-plant
Volatization of urea-based products (dry urea (46%N) or liquid UAN solutions)	Light texture soil	Applied by injection or mechanically incorporated
	Surface application	
	Warm sunny days after application	Rainfall immediately following application
	High residue cropping systems	Polymer coated urea Add urease inhibitors
	Soil surface pH>7.0	Sidedress N application instead of pre-plant
Nitrogen immobilization (temporary)	High residue no-till cropping systems	

Source: Adapted from http://www.agry.purdue.edu/ext/pubs/2006NLossMechanisms.pdf

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