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## **Understanding and managing uncertainties to improve biofuel GHG emissions calculations**

by

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## **Abstract**

In support of a major report on facilitating greenhouse gas (GHG) accreditation schemes for UK biofuels (Woods *et al.* 2008), this report reviews the nature and extent of uncertainties surrounding GHG emissions for biofuel feedstock production. It proposes (i) possible approaches for biofuel accreditation and (ii) research to improve GHG accreditation. The approaches recommended here are now recognised in the new HGCA GHG Calculator.

The major remaining uncertainties in estimating GHG emissions are in fertiliser manufacture, hence, fertiliser choice, the extent and attribution of in-field nitrous oxide (N<sub>2</sub>O) emissions, choice of land use reference system, how productivity affects land use, and how to allocate GHG emissions to co-products. Other significant uncertainties are in GHG emissions from inherent soil lime, effects on soil carbon of minimal tillage and indirect effects of using straw as fuel, how to attribute fuel use to biofuel crops, and indirect effects of biofuel production on carbon loss from organic soils. It is suggested that prohibitive costs of accreditation might be avoided if it is recognised that wheat, and possibly other grains, holds a 'signature' of its associated GHG emissions, and this could be revealed by analysis before processing, particularly for nitrogen (N).

# 1 Executive Summary

Carbon reporting of the greenhouse gas (GHG) emissions associated with the production of biofuels is important to ensure that climate change benefits are delivered. Over 60% of the GHG emissions of producing biofuel can come from producing the crop feedstock, but the level of these emissions can vary greatly and it is possible that growers could adopt practices to reduce them. Reliable and practical reporting of these GHG emissions will, therefore, be important for both the biofuels and agricultural industries. Unfortunately, many of the greatest uncertainties in calculating the GHG emissions of biofuels lie within feedstock production.

This report (as part of project MD-0607-0003 - Woods et al 2008) sets out the main issues involved in providing estimates of GHG emissions from biofuel feedstocks, the uncertainties involved and the likely pathways towards resolving these issues.

The issues surrounding the calculation of GHG emissions from feedstock production can usefully be divided into those of approach, and those of a technical nature.

## 1.1 Issues of Approach

### 1.1.1 *Farm vs Field level reporting*

Implications arise from choosing whether carbon reporting is required at individual field level or the farm level. There are questions of practicality, fairness and representativeness if either extreme is used. There are potential repercussions for estimating yields and nitrogen (N) fertiliser use, especially from growing for different crop markets (e.g. bread wheat). Inputs including N are usually recorded at field level whilst yields and fuel use are often not. Sensitivity analyses are required to assess the variabilities in GHG emissions both within and between farms. It is suggested that for reporting purposes 'representative' fields or blocks of land that are managed for the same market and in a similar way should be used.

### 1.1.2 *Appropriate Reference Systems*

The concept of using 'bare land' as a reference system for the 'alternative land use' to biofuel production is becoming increasingly inappropriate. It is therefore best to avoid reference land uses in calculations where possible, working on an absolute basis, and comparing GHG emissions for different land uses. An approach for using 'comparative reference systems' is explained. For fair comparisons to be made, much further LCA work on other crop production systems and land uses globally are required. Reference land use issues are complicated further by the potential for changes in productivity; if increased demand for crop feedstocks can be met by increases in yield per hectare rather than increased area of cropping, consequences on land-use change could be reduced. Recognition of yields is therefore important.

### 1.1.3 *Allocation of GHGs to Co-products*

The production of biofuels produces several co-products (including straw, animal feeds (DDGS or rapemeal) and surplus electricity). There is considerable controversy over how the GHG emissions associated with these co-products should be dealt with in the calculation of the GHG intensity (greenhouse gas emissions per unit output\_ of the biofuel; whether by substitution credit or allocation by mass, energy content or market value. Whilst there is no strictly 'correct' answer, it is important that the method used reflects the reality of the situation. It is tentatively suggested that using allocation by price gives the most realistic representation. The consequences of using different allocation/substitution approaches needs to be examined with the view to which best reflects reality, an approach for doing this using 'shadow credits' is outlined. The following conclusions on allocation are made:

- the guiding principle and ultimate test of any allocation procedure is that it should accurately reflect changes that actually happen or are likely to happen in the real world,
- allocation by means of physical characteristics, such as mass, energy content, etc., is rarely appropriate and should only be used, with caution, in very special circumstances,
- allocation using substitution credits is deceptively simple and attractive but can have major drawbacks which become apparent when actual consequences are considered carefully and properly by applying comparative reference systems,
- comparative reference systems must be established to reflect actual changes in the real world and the specific question under consideration by any given life cycle assessment,
- any substitution must be based on appropriate functional units for the products and services under consideration,
- substitution credits for by-products which displace other by-products are zero,
- allocation by price is the most realistic and practical means of addressing joint products in life cycle assessment provided that there is confidence that relative market prices reflect actual changes in the real world,
- it is necessary to examine circumstances where relative market prices do not reflect actual changes in the real world, such as the occurrence of market failures, market distortion, etc.,
- shadow prices, based on careful evaluation, should be adopted for any products and services that are not traded outside the process(es) under consideration, and
- both life cycle assessment practitioners and their audiences need to accept and appreciate that subsequent results can vary over time as both circumstances and the question under consideration change.

The GHG emission implications of burning co-products for energy need to be explored in more detail, since the burning of high value high protein animal feed may not be regarded as its most appropriate use, especially once all indirect land use change implications are considered.

## 1.2 Technical Issues

### ***1.2.1 GHG emissions from N fertiliser manufacture***

The manufacture of nitrogen (N) fertiliser produces GHG emissions both from the large amounts of carbon dioxide (CO<sub>2</sub>) from the energy required to fix ammonia, but also from nitrous oxide (N<sub>2</sub>O) created in the production of nitric acid. N<sub>2</sub>O has approximately 300 times the global warming potential of CO<sub>2</sub>. Typically, about 30% of the GHG emissions associated with the production of a biofuel may be from N fertiliser manufacture, of which roughly half is due to CO<sub>2</sub> and half due to N<sub>2</sub>O.

However, there is considerable variation in the GHG emissions associated with N fertiliser from different manufacturing plants and between different N fertiliser products. In particular, some factories use N<sub>2</sub>O abatement technologies that considerably reduce emissions; and the production of urea does not involve nitric acid so GHG emissions for this N fertiliser can be considerably lower than those for ammonium nitrate. Hence there is considerable uncertainty over the most appropriate GHG emissions factors to use in GHG intensity

calculations for N fertiliser manufactured in the UK, with ammonium nitrate predominantly used rather than urea, but generally from relatively 'clean' plants. There is a need to assess the difference in GHG emissions factors of different N fertiliser products, to ensure that appropriate values are used for the UK situation. It is also important that, if use of urea is effectively incentivised by carbon reporting methodologies, full consideration is also given to the likely impacts on national and global emissions of ammonia (which causes acidification, not global warming).

### **1.2.2 In-field N<sub>2</sub>O emissions**

As well as N<sub>2</sub>O emissions resulting directly from N fertiliser manufacture, there can also be large N<sub>2</sub>O emissions from crop soils, especially after artificial or organic fertilisers have been applied. Typically, in-field N<sub>2</sub>O emissions can be responsible for over 30% of the calculated GHG intensity of a biofuel. However, this is by far the most uncertain of all GHG emissions.

Direct measurement of N<sub>2</sub>O emissions is difficult, laborious and expensive so cannot be used widely, and is a significant limitation to fundamental research. There have been two main methods used in estimating N<sub>2</sub>O emissions from crops for life cycle assessment purposes. The IPCC methodology calculates emissions from N additions to the soil using default emission factors (e.g. Williams et al., 2006a). Alternatively, a modelling approach can be used to estimate typical emissions from a given region, assuming typical inputs, for example using the soils model DNDC (e.g. Edwards et al. 2006). As well as direct N<sub>2</sub>O emissions, indirect emissions subsequently arising from leached nitrate and volatilised ammonia should also be accounted for. Assuming average N fertiliser rates of around 200kg N/ha, N<sub>2</sub>O emissions are estimated to be around 4-6 kg N<sub>2</sub>O/ha, giving 1200-1800 kg CO<sub>2</sub> eq/ha.

Emissions of N<sub>2</sub>O come from microbial processes in the soil converting N into different forms (nitrification and denitrification) and these are inherently variable with temperature, moisture and soil conditions; hence the 'real' soil N<sub>2</sub>O emissions from an individual biofuel crop may actually be close to zero or could be several times greater than those calculated. In terms of estimating N<sub>2</sub>O emissions for accreditation purposes, it is not practical to attempt to determine or estimate actual soil N<sub>2</sub>O emissions for each individual crop. However, variations in likely N<sub>2</sub>O emissions caused by soil type, climate, region or management factors could, with significant further work, be accounted for. Such work is required to ensure that estimates of N<sub>2</sub>O are appropriate for UK regions and management systems. It is possible that best estimates could be substantially higher or lower than those calculated using IPCC defaults.

As N<sub>2</sub>O emissions are linked to the amount of N available in the soil, it is appropriate for accreditation purposes that emissions are driven by N fertiliser application. The IPCC approach (2006 Guidelines) provides the simplest, most transparent and defensible basis for quantifying N<sub>2</sub>O emissions. The emission factors used in these calculations need to be verified for UK conditions. Potentially, different emission factors could be used for different regions or management practices, and could be used for RTFO reporting purposes.

Emissions from non-fertiliser sources of N (biological fixation, deposition, organic manures, crop residues, and changes in soil organic matter) can be difficult and impractical to quantify, and how they are dealt with could have serious implications for incentivising on-farm practices.

Organic manures tend to be applied every few years as part of a rotation, and much of the nitrogen they contain is not available to the crop to which they are applied, giving problems of apportionment. Only part of the N content of manures may be susceptible to N<sub>2</sub>O emission, and N fertiliser use on a crop where manure has been applied will only be displaced by the available N content. Any accreditation system should seek to encourage best practice with regard to manure management so that their other environmental impacts are recognised, hence the use of manures on biofuel crops should not be unfairly penalised.

There are similar uncertainties over emissions from the incorporation of crop residues; emissions may be dependent on the carbon:nitrogen (C:N) ratio of the residues. The practice of incorporating residues (and organic manures) can help increase soil organic matter, potentially sequestering carbon, so care is needed that good practice is not unduly penalised.

Mineralisation of soil organic matter can give N<sub>2</sub>O emissions, with cultivation encouraging such mineralisation. However, measuring or estimating these changes is difficult and impractical, and in many cases may be considered negligible. N<sub>2</sub>O emissions from mineralised soil organic matter will be greatest in highly organic, humose or peaty soils.

Accounting for N<sub>2</sub>O emissions from non-fertiliser sources using an adapted IPCC approach seems likely to allow the fastest progress. These issues will need to be reviewed before economic incentives are derived for achieving low carbon intensities, or perverse practices could be encouraged.

In terms of producing a conservative methodology for dealing with N<sub>2</sub>O emissions, it is recommended here that:

- organic additions are accounted for using the IPCC approach on the basis of *available* N content rather than *total* N content.
- crop residues are accounted for using the IPCC approach assuming a modest N addition that is included irrespective of yield, N fertiliser and whether or not straw is removed.
- appropriately large emissions should be assumed for cropping on organic and humose soils.

A number of areas are identified where further research will aid understanding and potentially lead to improved calculations:

- emissions of N<sub>2</sub>O from crop residues and from organic manures over the long term requires further research work. It remains to be proven that N returned to the soil in crop residues should be treated the same as applied N from fertiliser.
- field measurements are required to assess the scale of N<sub>2</sub>O emissions from UK organic and peaty soils.
- the relationship between N<sub>2</sub>O emissions and N applied is currently assumed to be linear; however there is mounting evidence that this may be non-linear, with N applications above the optimum for the crop giving much greater emissions than applications at or below the optimum.
- consideration is needed of baseline N<sub>2</sub>O emissions, i.e those that would occur anyway if no crop or an unfertilised crop was grown, and agreement globally on how to account for these emissions.
- N<sub>2</sub>O emissions are affected by unaccountable, fine-scale changes in weather, specifically rainfall and temperature, but it may be possible to account for climatic differences between regions by using UK-DNDC to develop different emission factors, perhaps on a postcode basis. Consideration would be needed to ensure that such a system was not divisive. Soil texture can also be an important factor determining N<sub>2</sub>O emissions, interacting with climate. However, whilst differences between clay soils in the wetter West and light sandy soils in the drier East may be significant, differences between less extreme soil types are less consistent. Further consideration is needed to account for differences between soil types, with the possible development of a soil-climate matrix using UK-DNDC
- fertiliser type and timing of application may significantly affect N<sub>2</sub>O emissions; further field research is required.



- cultivation type and method of residue incorporation may significantly affect N<sub>2</sub>O emissions; further field research is required

There is a need to reconcile the IPCC approach to N<sub>2</sub>O emissions, DNDC model outputs and findings from recent work (e.g. Crutzen et al., 2007). Whilst the work of Crutzen et al. (2007) suggests that real N<sub>2</sub>O emissions from biofuel cropping may be higher than calculated from the IPCC approach, there is considerable evidence from field experimentation and modelling that the IPCC approach may significantly *overestimate* the real N<sub>2</sub>O emissions from cropping in the UK. In this case, biofuel production in the UK could be unfairly penalised.

It is suggested here that it may be possible to develop an approach to estimating N<sub>2</sub>O emissions from crop N% content that may cut through the issues outlined above, and allow verification to occur. This makes use of the observation that N% is relatively constant at the optimum N rate. Further work is required to develop and validate this as an accurate and usable measure for accreditation purposes.

### ***1.2.3 In field CO<sub>2</sub> emissions/sequestration from Soil Organic Matter***

The soil can act as either a sink or a source for CO<sub>2</sub>, dependent on various soil factors and agricultural practices. Soils may lose Soil Organic Matter (SOM) by mineralisation through cultivation, so emitting CO<sub>2</sub> to the atmosphere. Alternatively, SOM may increase through cropping, or from repeated addition of crop residues or organic manures. The potential savings in CO<sub>2</sub> through sequestration by straw incorporation are potentially large (of the order of 1 or 2 tonnes CO<sub>2</sub>/ha). Large savings may also be possible through direct drilling or reduced cultivations, but there is uncertainty over the scale of these effects. As the amount of animal manures, compost and sewage sludge is determined by how much is produced, not by agricultural demand, it is questionable whether their incorporation should be considered within the biofuel crop production system. As carbon returned to the soil in farmyard manure (FYM) is largely in the form of straw that will have previously come off the land elsewhere, the issues of incorporating straw and FYM need to be considered together.

In terms of including calculations of SOM in accreditation schemes there are several difficulties; carbon sequestered in the soil is affected by past and future management; it cannot be considered to be cumulative over years and it is difficult to verify. As the practices to increase SOM are required on a long-term basis, are finite and are quite quickly reversed on reversion back to conventional practice, it would be unjustified to allocate GHG savings from such practices to an individual biofuel crop year, and impossible to verify that savings were not lost in subsequent years.

Consideration of SOM may be more important in other parts of the world, or with perennial crops, so there may be a case for its inclusion on a regional scale to allow fair comparisons with other global crop systems.

If SOM is to be used in GHG emission calculations then a practical approach for accounting for changes and dealing with the issues will need to be developed. To ensure that benefits are not over-represented, accounting for CO<sub>2</sub> sequestration from straw incorporation or zero-tillage would require that any negative consequences on N<sub>2</sub>O emissions are also included appropriately. A pragmatic first solution may be to count a CO<sub>2</sub> credit for all straw that is not burned, assuming that, if the straw is not burned, it will find its way back to the land at some point through FYM.

### **1.2.4 CO<sub>2</sub> released from organic soils**

SOM changes due to land use change, i.e. converting from grassland to arable, can give very large emissions of CO<sub>2</sub> over a period of years (>3500 kg CO<sub>2</sub>/ha.yr). Even with very efficient biofuel production it could take decades in such circumstances before sufficient GHG emissions reductions have accumulated from the use of biofuels to give a truly net positive GHG emissions saving (Edwards et al., 2006; Searchinger et al, 2008).

The long-term nature of these effects makes them very difficult to deal with in a quantitative way in accreditation systems. It is also difficult to attribute such land-use change directly to cropping for biofuels; if grassland is converted to arable it is likely that the crops for food as well as for fuels will be grown in the rotation. Further problems arise if land-use change results indirectly from cropping for biofuels, i.e. when biofuel crops displace food crops so more land is required to produce food. In such a case, it does not really matter whether biofuel crops or food crops are grown on the previously uncropped land, the change is partially due to the introduction of biofuels and the consequent GHG emissions remain the same. Because of these difficulties, the CO<sub>2</sub> consequences of indirect land use change have generally been excluded from assurance and certification systems. It has been suggested that such issues of land use change are more appropriately dealt with through government legislation. The publication of two major studies assessing the indirect land use change effects of biofuels (Searchinger et al., 2008; Fargione et al., 2008) came after this report was drafted. These issues are being studied in more detail in the recently announced 'Gallagher Review' on the indirect effects of biofuels.

### **1.2.5 Lime acidification and CO<sub>2</sub> release**

This study has identified a potentially significant source of CO<sub>2</sub> from cropping that has previously not been recognised. Whilst CO<sub>2</sub> release from the acidification of lime applied to agricultural crops can be calculated using IPCC methodologies, the acidification of naturally-occurring limestone or chalks on cropped land has been ignored. As a large proportion of UK arable land is naturally calcareous, emissions could be substantial, estimated to be 200-300 kg CO<sub>2</sub>/ha. Many of the soil processes causing soil acidification are associated with fertiliser application. There is a clear need for clarification of this issue and for further research to relate fossil C release from soils to agronomic practices. It would seem sensible for estimation of fossil C release from soils (and liming materials) to be based on fertiliser use rather than on use of lime.

### **1.2.6 Grain drying**

There are a number of issues surrounding mechanical grain drying. It is recommended in this work that GHG emissions from grain drying should be estimated on a per tonne basis considering:

- the proportion of grain dried on a whole farm basis
- average moisture content removed
- the energy requirement of the drying system used per % moisture content removed per t grain.

There are no good data on the amount of grain drying, or types of grain dryers now used on UK farms. There is a need to survey farm practice to estimate the size and importance of the issues around grain drying, and to develop sensible and realistic default values. There may be an opportunity to begin this survey process through the HGCA farm audit questionnaires.

### **1.2.7 Farm Diesel usage**

The GHG emissions from farm diesel use generally make up a relatively small proportion of the total GHG emissions from cropping, around 5-10% using the GHG calculator. However, estimating farm diesel usage associated with growing a specific biofuel crop for accreditation purposes could require quite detailed, time consuming, records, additional to current practice. There is a need to develop a system that uses simple information on cultivation, agrochemical application and harvesting operations, together with machinery types and energy usage to estimate farm diesel usage for different farms, accounting for different soil types. Further reliable information is required for diesel use of modern farm machinery under different soil types. For instance, diesel used in transporting grain by agricultural tractor needs to be considered, because it is relatively expensive in GHG terms.

## **1.3 Research Requirements**

1. Fertiliser management (mainly N) and impact assessments:
  - i. Provide detailed analyses of in-field N<sub>2</sub>O emissions. Evaluate the appropriateness of the IPCC emission factors for N<sub>2</sub>O emissions from UK arable biofuel cropping. Approaches for dealing with organic manures, crop residues, organic soils and baseline emissions from non-cropped land need to be developed and evaluated. Given the relative paucity of published data on N<sub>2</sub>O emissions from arable soils, and the large expense of experimental N<sub>2</sub>O measurement, the UK-DNDC model will play a useful role in resolving these questions.
  - ii. There is a need to evaluate the most appropriate GHG emissions factors for fertiliser manufacture for ammonium nitrate and other N fertiliser products in the UK. The variation in manufacturing emissions between products, manufacturing plants and country of origin needs to be assessed.
  - iii. Exploration of how N fertiliser rates could be optimised for GHG emissions savings could be very instructive for the agricultural and biofuels industry. The N fertiliser rates that maximise GHG emissions savings should be determined (see Kindred et al., 2007a), and the economic costs of optimising GHG emissions savings should be assessed.
  - iv. The potential for using grain N% (or grain protein) as a 'signature' for GHG emissions from nitrogen needs to be evaluated.
2. Quantify the CO<sub>2</sub> emissions resulting from the acidification of applied and inherent lime in soils. The current understanding in the literature needs to be reviewed, and there may be a need for experimentation.
3. Develop globally agreed standardised allocation procedures for co-products.
4. Develop and employ standardised comparative reference systems – requires the development of a global land use inventory.
5. Review the scope for better crop productivity to mitigate GHG emissions arising from extension of arable land use at a global scale, and assess the extent to which increased commodity prices might attract the requisite investment to increase crop productivity.

## **2 Introduction**

This report has been developed in parallel to a report outlining the developments in the biofuel GHG calculator funded by the UK Home Grown Cereals Authority (Woods et al., 2008). Carbon Reporting of the GHG intensity of biofuels is important to ensure that climate change benefits are delivered. Over 60% of the GHG costs of producing biofuel can come from producing the crop feedstock, but the level of these costs can vary greatly and it is possible that growers could adopt practices to reduce these costs. Reliable and practical reporting of these costs will therefore be important for both the biofuels and agricultural industries. Unfortunately, many of the greatest uncertainties in calculating GHG intensities of biofuels lie within feedstock production.

This report (as part of project MD-0607-0033 - Woods et al., 2008) sets out the main issues involved in providing estimates of GHG emissions from biofuel feedstocks, the uncertainties involved and the likely pathways towards resolving these issues.

The key uncertainties that surround quantifying and accrediting the GHG emissions from growing biofuel crops are assessed below. These uncertainties can be divided into broad categories:

- i. those that predominantly arise from the diversity of approaches and biophysical environments that biofuel feedstocks can be produced from, and;
- ii. those that are more technical in nature and where a more robust evidence base is needed.

Much of the uncertainty lies in attempting to understand what level of detail is required in the monitoring and accounting procedures to provide a valid average for a field or farm level operation. There are also issues of fundamental scientific uncertainty where insufficient knowledge is available to provide an adequate level of precision. Despite these uncertainties often being possibly large enough to change the outcome of the GHG balance, considerable knowledge will be gained through learning-by-doing. Indeed, it may not be possible to gain sufficiently broad data sets through any other means. These issues are explored in more detail in subsequent sections.

## **3 Issues of Approach – uncertainty arising from diversity**

In general, this category of uncertainty arises from variations in location (e.g. soil), time (e.g. climate and history of land use), and in management practices. In theory, the uncertainties could be minimised by sufficiently detailed measurements, monitoring intervals and calculation methodologies. In practice the bureaucratic and regulatory burdens, in particular to farmers, would be onerous. This category of uncertainty is summarised below and options for managing this complexity whilst delivering viable and therefore credible outcomes are discussed in the conclusions.

### **3.1 Data collection issues at a field or farm level**

Preliminary HGCA funded auditing trials to date have taken data for one individual field per farm (Woods et al. 2008). Whether an individual field can be taken as being representative of the farm as a whole is questionable. Farm inputs of diesel power, chemicals and fertilisers

will differ between crops and between fields, and to some extent, can even differ within fields. The same is true of crop yields and crop quality. Variability between fields in inputs and outputs is likely to be greatest with differences in soil types and rotational position. How much these vary within a farm might depend on the size of the farm, the crops, varieties and rotations used, and inherent variability across soil types. On top of this, crop management and crop yields will depend on the market the crop is grown for; e.g. wheat grown for bread-making usually receives more nitrogen fertiliser (by around 40kg N/ha) and achieves lower grain yields (by around 10%) than wheat for animal feed, perhaps increasing GHG intensity from around 444kg CO<sub>2</sub> eq/t grain for feed wheat to 572kg for milling wheat, using the approach and figures used in Kindred et al. (2007a). The importance of this variability across whole farms needs to be considered further. This could be addressed to an extent in on-going HGCA farm audit trials.

Ideally, it may be argued that data on inputs and yields etc should be collected for each field in which a biofuels crop is grown. This would however have major cost implications in terms of data provision by the farmer, data collection by an auditor, and data analysis by the scheme administrator. This could potentially also unfairly burden farmers with smaller field sizes compared to those with larger fields; if 500t of grain came off a single field of 50ha this could only require 1 field input sheet, but 500t of grain coming from 10 adjacent fields of 5ha could require 10 field input sheets.

Traceability of grain back to individual loads of grain and their associated grain quality back to individual fields is an issue; grain from different fields is likely to be bulked together for storage, making it difficult to specifically marry up data on individual fields with individual lots of grain entering a biofuels facility. This is especially difficult if, as is likely, audits for crops occur in the year *after* the harvest of the biofuel crop; full crop information will not be known until after harvest, and due to busy farm workloads it is unlikely that auditing could commence much before November, i.e. after much of the grain may have left the farm.

If there is a need for segregation of grain that is destined for biofuel serious logistical problems could result on-farm, possibly leading to inefficiencies that could potentially increase the energy and management costs involved in grain storage.

If field-based assurance is impractical in terms of providing a valid representation of the GHG emissions resulting from the production of a batch of biofuel feedstock, could an whole-farm approach be adopted? Biofuel feedstocks are grown as part of the farm business, often with multiple outputs. How biofuel feedstock production interacts with the rest of a farms production needs to be understood and quantified. Having a carbon 'rating' for the whole farm system would encourage farmers to improve this carbon rating year on year and so may be an effective way of meeting the over-arching goal of reduced GHG emissions. However, to be a practical solution for biofuel producers to provide biofuels with a certified GHG balance, batch-based accounting would still be required. Whole farm accounting might reward integrational efficiencies and alleviate situations where carbon-saving practices on biofuel crops are offset by carbon-negative practices on food crops. However, a whole-farm accounting system may be likely to give rise to complexities and inequalities between farm systems (such as between arable and mixed arable-livestock farms) that would need to be fully thought through.

### ***3.1.1 Collecting data at a field or farm level***

Whilst farmers will have good records for some inputs at the field-level, (e.g. chemical and fertiliser inputs) due to management, regulatory and assurance requirements, other records may be less easily obtainable, for example diesel use per ha or field. It is possible that growers may be able to quantify data such as this more easily and accurately on an annual

basis across the whole farm. Even data on grain yields may not always be available or reliable on a field basis for all farms.

Dealing with data for grain drying presents difficulties at both the field or farm levels. There is unlikely to be detailed data on energy used in grain drying for individual fields. Drying can be very different between fields, farms and between years. Grain from many individual fields may not be dried at all, whilst for other fields the majority of the grain may be dried. Whether or not grain is dried will be dependent on weather conditions, harvesting capacity of the farm and attitude of the farmer.

It is also likely that there will be a geographical divide in the amount of grain drying with more in northern areas where harvest is later and there are fewer 'dry' combining days than in the southern and eastern regions of the country. The harvesting capacity in relation to the area to be harvested will also be important; to spread the large capital costs of combine harvesters farmers may maximise the area to be cut per machine. To do this it is often necessary to ensure that the combine is operating at all times when the crop will thresh, even if the grain has a fairly high moisture content, for example early in morning and late into the evening. In order to do this the farm must have the capacity to deal with and dry large quantities of moist grain. Farmers with more limited drying facilities, and ample harvesting capacity, are likely to be more reluctant to combine when the grain is above 15% moisture content.

Collecting data for things like diesel use at a farm level would be very difficult for farms which are not wholly combinable arable crops. Comparisons across farms with livestock, or large areas of potatoes or vegetable crops, may be unfair. It would also be very difficult where farms undertake a significant amount of contracting for other farmers, or if the farmer employs contractors to do some or all of the field work.

If a farm level approach is taken, is it right that a 1000ha farm could be represented by the same level and detail of data as a 50ha farm?

In order to avoid the problems associated both with working at the field and whole-farm levels, it is probably most appropriate to work at a level in between. The whole crop level may be simplest, but would not discriminate between crops grown for different markets, and as such might not encourage use of best-practice to optimise carbon balance for biofuels crops. It is probably necessary therefore to get down to the 'crop market level'. Farmers could be asked to give details of the representative (or averaged?) crop grown on the farm for the biofuels market. This could then require only one audit sheet per biofuel crop per farm. However, this may not fairly account for differences across the farm in yield levels and inputs caused by differences in previous cropping, soil type etc. It may be appropriate to get down to a 'crop management' level. Farmers would then be asked to identify blocks of crops that are treated the same in management terms and yield similarly; these could be individual fields or blocks of 100s hectares. A decision would be needed on how large a difference in management or yield level would need to be to constitute requiring a new audit certificate.

There is a need for a sensitivity type analysis to assess how large differences are between farms and within farms to help determine what level of data is most appropriate. This could be assessed as part of the on-going HGCA farm-audit trials.

If figures are not taken from individual fields there is a question over how to average figures of grain yield, inputs, nitrous oxide (N<sub>2</sub>O) emissions, grain drying etc over different fields; arithmetic means may not be the most appropriate method. If figures are aggregated there may be considerable difficulties in verifying or policing these figures.

### **3.2 'Game playing' and rotational issues**

It is possible that if calculations are made in a certain way then this could encourage farmer behaviour that may not be appropriate in the context of reducing greenhouse gas (GHG)

emissions from the farm in the round, or from the whole rotation. For example, it is possible that growers could do a number of things to reduce N rates and give a more positive calculated GHG balance for the biofuel crop but which may not actually benefit GHG emissions in reality. Super-optimal N applications to previous crops might be the most extreme example of this and would actually result in greater whole farm GHG emissions and increased diffuse pollution levels but would allow reduced reported N-application to the biofuel feedstock.

Growing the biofuel crops after crops already grown in the rotation that give large N residues (e.g. potatoes, oilseed rape), or in the rotation after manure applications, would also reduce N fertiliser requirements for the biofuel crop, but may not reduce N fertiliser use across the farm. Whilst this would not actually increase GHG emissions it could lead to GHG savings being represented as greater than they actually are, if over the whole rotation nothing has actually changed, i.e. more N is applied to other crops in the rotation. On the other hand, accreditation schemes could lead to more careful planning and discerning use of N in the soil and in manures. This comes back to the issue of accounting for the full rotation and questions whether taking results from a single field can really be seen as being fair and representative.

Grain drying is another example where 'game playing' could occur that wouldn't show real GHG benefits; the farmer could choose to harvest the biofuel crop in conditions where grain drying was minimised or not necessary, but would then have to dry more of the conventional crop.

Consideration will need to be given in any accreditation scheme to avoiding these 'game playing' issues. Ensuring that the data used is *representative* of the farm may help in this regard. There are however inevitable difficulties in verification.

### 3.3 Reference systems

Assessing the net impact of a biofuel chain on atmospheric GHG levels requires a notional estimate of what would have happened if the biofuel was not produced. This is because many of the GHG emissions assessed need to be calculated in relation to 'background' emissions. However, the choice of reference system for land-use means choosing the most likely alternative land management choice. Deciding on appropriate reference systems for comparison within the life cycle analysis (LCA), especially in terms of land-use, has major consequences for the calculated GHG emissions from biofuels, and results in expanded system boundaries, with consequent increases in the quantity of data required.

The current approach to reference land use proposed in the RTFO consultation addresses both alternative and previous land use (Department for Transport, 2007a and 2007b). Alternative land use concerns the overall GHG emission effects when the use of land is changed to growing crops for the production of biofuels, compared to what the land would otherwise be used for if biofuels weren't being grown. This is a potentially complex issue because of the consequences of displacing previously grown crops to other locations which may have very significant impacts on associated GHG emissions. The complexity derives from a combination of what crop is being displaced, where it will be grown instead, how it will be grown and what are the resulting GHG emissions, both from the land use and from any land use change. Faced with such complexity which can only be resolved by access to simulated market analysis and a global database of crop GHG emissions information, it has been recommended that the effects of alternative land use will not be included in the RTFO methodology. Instead, the emissions resulting from so-called 'indirect land use' will be addressed retrospectively by the RTFO administrator. Subsequent to the drafting of this report, important papers by Searchinger et al (2008) and Fargione et al (2008) attempt to quantify the indirect land use change consequences of biofuel production, and find these

consequences to be so large as to make most conventional crop-based biofuels untenable as a climate change mitigation measure. It is not possible to explore these issues further here, but they are the subject of the recently commissioned 'Gallagher Review' of the indirect effects of biofuels.

Previous land use mainly concerns the GHG effects of converting uncultivated land to biofuel crop production. Whilst this is unlikely to be a major issue in the UK, it can have serious consequences for biofuel crops that are grown elsewhere throughout the world. It has been recommended that reporting of GHG changes from previous land use should be accounted relative to 2005. If previous land use is not reported then it is proposed that no default value should be applied in the initial stages of the RTFO. This has been criticised because, generally, it provides no incentive to report or to avoid damaging land use changes and, specifically, it could disadvantage UK producers relative to foreign suppliers who choose not to report sensitive land use changes which result in significant GHG emissions. Hence, whilst the currently proposed approach may not affect UK producers directly, it could affect them indirectly. Spot market purchasers on the biofuels market are unlikely in the short term to be able to identify land use change associated with the biofuels purchased; the proposed RTFO approach allows time for information flows to be established. It is likely that this 'unknown' allowance for LUC will not follow through to a carbon-based scheme (from 2010).

Within life cycle assessment, reference systems are used to determine the avoided energy resource depletion and greenhouse gas emissions which occur when the activity under consideration displaces another activity. Reference systems are needed for land because, as a basic natural resource, it can have many different uses. These have implications in terms of primary energy inputs and associated greenhouse gas emissions. Hence, it is necessary to take into account avoided fossil energy resource depletion and greenhouse gas emissions. This is done by assigning credits to the biofuel produced when its production avoids emissions from either or both, alternative land use that would have had greater emissions or fossil fuel use.

Many different alternative land uses can be chosen. In many previous life cycle assessment studies of biofuels, maintained set-aside is taken as the alternative land use. There are at least three reasons for this. First, it is relatively easy to estimate the primary energy inputs and associated greenhouse gas emissions of maintaining set-aside land. Second, no tangible product is derived from set-aside land which would complicate the calculations. Third, it is likely that set-aside land will be used, in the early stages of development, to grow crops for biofuel production. Whilst the first two reasons are relevant to the practicality of life cycle assessment, it is only the third reason which is actually valid for the choice of reference system. A fundamental rule in selecting reference systems (and, as it happens, allocation procedures) is that they should reflect the reality of the situation which is being assessed. Unfortunately, this basic principle often seems to be forgotten in life cycle assessment, leading to the view that the choice of reference systems (and allocation procedures) is arbitrary and, hence, subsequent results are, inevitably, subjective.

The way to resolve such errors in the application of life cycle assessment is to always remember that the way in which calculations are performed and the results that are obtained is dictated solely by the question which the work is attempting to answer. It may seem obvious that "any final answer depends on the initial question" and that, as a consequence, the results of life cycle assessment vary depending on circumstances. However, it is all too often assumed that life cycle assessment produces unique and universal answers that can be used to address any and every question; "The answer to life, the universe and everything is 42, but do you really understand the question" (Adams, 1979). The proposed approach to resolving this problem is to create "comparative reference systems". The use of such an approach can be demonstrated by means of a relevant example (Mortimer 2006a).

Previously, oilseed rape (OSR) for biodiesel production in the European Union (EU) could be said to be most likely to be grown on "set-aside" land which is specified for non-food



production by the European Commission (EC). This situation can be represented by the comparative reference system illustrated in Table 1, which represents an essential “balancing” of inputs (in this example, land) and outputs (in this example, energy in the form of road transport fuels) before and after the switch, normalised, for simplicity, to 1 hectare of land in the United Kingdom (UK). The reference land use in this case would be maintained set-aside which releases a small amount of greenhouse gas emissions. In keeping with the use of reference systems, such emissions would constitute a credit which would be subtracted from the total greenhouse gas emissions of biodiesel production from OSR. Using data available from life cycle assessment studies, the results of these comparative reference systems and the net savings (relative change) in total greenhouse gas emissions are summarised in Table 2. These results answer the question “what is the magnitude of greenhouse gas emissions avoided by the production and use of this particular liquid biofuel in the UK relative to conventional fuels from fossil sources *in the near-term future*?” In this context, “near-term future” refers to the early years in the period up to 2010 when the next target for biofuel production has been set by the relevant EC Directive (Anon 2003).

**Table 1: Comparative Reference Systems for Biodiesel Production from OSR in the UK in the Near-Term Future**

Before		After	
<b>Inputs:</b>		<b>Inputs:</b>	
Land for Set-Aside Use in UK	1.0 ha	Land for OSR in UK	1.0 ha
<b>Outputs:</b>		<b>Outputs:</b>	
Diesel from Crude Oil in UK	40,335 MJ	Biodiesel from OSR in UK	40,335 MJ

**Table 2: Net Greenhouse Gas Emissions Savings for Biodiesel Production from OSR in the UK in the Near-Term Future**

Situation	Calculation	Total Greenhouse Gas Emissions (kg eq CO <sub>2</sub> )
<b>Before</b>		
Maintenance of set-aside land in UK	1 ha.a x 366 kg CO <sub>2</sub> /ha.a	336
Production and use of diesel from crude oil	40,335 MJ x 0.08755 kg CO <sub>2</sub> /MJ	3,531
<b>Sub-Total</b>		<b>3,897</b>
<b>After</b>		
Production of biodiesel from OSR in the UK	40,335 MJ x 0.0680 kg CO <sub>2</sub> /MJ	2,743
<b>Sub-Total</b>		<b>2,743</b>
<b>Grand Total</b>		<b>- 1,154</b>
<b>Net Savings</b>		<b>- 30%</b>

As the targets for biofuel production increase in the future, more land will need to be converted to these biomass energy crops. In the case of oilseed rape, it might be assumed that land currently used for existing food use (such as cooking oil) could be transferred to biofuel production. If this happened, the existing food product (cooking oil) displaced would have to be obtained from elsewhere. Consequently, a new comparative reference system has to be established that reflects the diversion of existing OSR cultivation into biodiesel production and the cultivation of OSR for cooking oil on currently unused land in, say Australia. This is represented by the comparative reference system shown in Table 3 which adapts data on oilseed rape production in the UK (El Sayed et al. 2003) and in Australia (Narayanaswamy et al. 2004). This comparative reference system provides the correct basis to answer the new question; “what is the magnitude of GHG emissions avoided by the

production and use of this particular liquid biofuel in the UK relative to conventional fuels from fossil sources *in the medium-term future?*” The results for total greenhouse gas emissions are summarised in Table 4 (El Sayed et al. 2003). It will be seen that instead of a net saving, a small net increase in total greenhouse gas emissions of 6% occurs due mainly to the less efficient production of OSR in Australia compared with the UK.

**Table 3: Comparative Reference Systems for Biodiesel Production from OSR in the UK in the Medium-Term Future**

Before		After	
<b>Inputs:</b>		<b>Inputs:</b>	
Land for OSR (food) in UK	1.0 ha	Land for OSR (fuel) in UK	1.0 ha
Unused Land in Australia	3.6 ha	Land for OSR (food) in Australia	3.6 ha
<b>Outputs:</b>		<b>Outputs:</b>	
Cooking Oil in UK	1,186 l	Cooking Oil in UK from Australia	1,186 l
Diesel from Crude Oil in UK	40,335 MJ	Biodiesel from OSR in UK	40,335 MJ

**Table 4: Net Greenhouse Gas Emissions Savings for Biodiesel Production from OSR in the UK in the Medium-Term Future**

Situation	Calculation	Total Greenhouse Gas Emissions (kg eq CO <sub>2</sub> )
<b>Before</b>		
Cooking oil from OSR in UK	1,186 l x 1.094 kg CO <sub>2</sub> /l	1,297
Production and use of diesel from crude oil	40,335 MJ x 0.08755 kg CO <sub>2</sub> /MJ	3,531
<b>Sub-Total</b>		<b>4,828</b>
<b>After</b>		
Cooking Oil from OSR in Australia	1,186 l x 2.007 kg CO <sub>2</sub> /l	2,380
Production of biodiesel from OSR in the UK	40,335 MJ x 0.0680 kg CO <sub>2</sub> /MJ	2,743
<b>Sub-Total</b>		<b>5,123</b>
<b>Grand Total</b>		<b>+ 295</b>
<b>Net Savings</b>		<b>+ 6%</b>

Biofuel targets may be higher in the more distant future. This means that there could be increased pressure to convert land which currently grows other food crops to OSR production for biodiesel. For example, land for current wheat production for food could be turned over to OSR production for biodiesel (although the actual dynamics of this illustrative example will depend on farming cycles, land suitability, etc., as well as the relative demands for biodiesel and bioethanol). Table 5, based on data from the UK (El Sayed et al 2003) and Australia (Narayanaswamy et al. 2004) provides an indication of the comparative reference system that would have to be considered if such a situation was to materialise in the longer-term future. Using the information related to the comparative reference systems in Table 5 it would be possible to answer the then-relevant question; “what is the magnitude of GHG emissions avoided by the production and use of this particular liquid biofuel in the UK relative to conventional fuels from fossil sources *in the longer-term future?*” The results for total greenhouse gas emissions are summarised in Table 6. This shows that there is a significant net increase in total greenhouse gas emissions of 23% due chiefly to the less efficient production of wheat in Australia compared with the UK.

**Table 5: Comparative Reference Systems for Biodiesel Production from OSR in the UK in the Longer-Term Future**

Before		After	
<b>Inputs:</b>		<b>Inputs:</b>	
Land for Wheat (food) in UK	1.0 ha	Land for OSR (fuel) in UK	1.0 ha
Unused Land in Australia	3.3 ha	Land for Wheat (food) in Australia	3.3 ha
<b>Outputs:</b>		<b>Outputs:</b>	
Wheat Grain in UK	6.93 t	Wheat Grain in UK from Australia	6.93 t
Diesel from Crude Oil in UK	40,335 MJ	Biodiesel from OSR in UK	40,335 MJ

**Table 6: Net Greenhouse Gas Emissions Savings for Biodiesel Production from OSR in the UK in the Longer-Term Future**

Situation	Calculation	Total Greenhouse Gas Emissions (kg eq CO <sub>2</sub> )
<b>Before</b>		
Wheat grain (food) production in UK	6.93 t x 201 kg CO <sub>2</sub> /t	1,393
Production and use of diesel from crude oil	40,335 MJ x 0.08755 kg CO <sub>2</sub> /MJ	3,531
<b>Sub-Total</b>		<b>4,924</b>
<b>After</b>		
Wheat grain (food) production in Australia	6.93 t x 480 kg CO <sub>2</sub> /t	3,326
Production of biodiesel from OSR in the UK	40,335 MJ x 0.0680 kg CO <sub>2</sub> /MJ	2,743
<b>Sub-Total</b>		<b>6,069</b>
<b>Grand Total</b>		<b>+ 1,145</b>
<b>Net Savings</b>		<b>+ 23%</b>

From this example, it can be seen that:

1. net greenhouse gas emissions saving can change substantially with circumstances and over time, and;
2. the net savings can transform into net increases.

Obviously, the results depend crucially on the assumptions adopted and the alternative land uses chosen. Ideally, a database of reliable total greenhouse gas emissions for the range of possible alternative land uses is required in order to undertake comprehensive and convincing assessment. However, it should be appreciated that the creation and maintenance of such a database is not a trivial matter. This is because it will have to be very extensive and it will have to contain results that have been derived from fully transparent life cycle assessment studies so that consistency and confidence can be achieved in the analysis. This means that a considerable number of life cycle assessment studies would have to be conducted for many different crops in many different countries. Additionally, database maintenance is required because associated total greenhouse gas emissions can change over time as cultivation practices alter. For example, one important consideration would be improvements in productivity of crops which alter the “land balance” in the comparative reference systems. This could be examined further by considering hypothetical productivity improvements using relevant examples.

### **3.3.1 Set-aside as a reference system**

The reference system for land-use normally used in GHG calculations is set-aside land. Since 1992, in order to reduce 'over-production' there has been a requirement for farmers to 'set-aside' part of their land in order to receive subsidies under the Common Agricultural Policy (CAP). Though the CAP regime has changed in recent years so that crops do not have to be grown in order to receive the payment, there has been an obligation for farmers to set-aside 8% of their land from food crops. This requirement was reduced to 0% for 2007/2008, and is likely to remain at 0% for the foreseeable future. Set-aside land can be in rotational set-aside where it is either left fallow or an industrial (non-food) crop grown on it, or it can be in non-rotational set-aside where it is generally left as permanent grass.

Using set-aside as a reference system may be appropriate where oilseed rape is grown on claimed set-aside land, and on land where if the grower did not grow oilseed rape for biofuel the land would be left uncropped (i.e. the grower would not grow OSR for conventional food oil markets or grow an alternative break crop). However, in practice few growers have chosen to leave productive land un-cropped even when crop prices have been anticipated to be less than the cost of production. If crop prices are reasonable, and the grower stands a good chance of profiting (or at least not losing money) by using land to grow a crop then it is unrealistic to assume that if the biofuel crop was not grown, no crop would be grown. It would appear very unrealistic to many UK farmers to assume that if a wheat crop was not used as a biofuel it would not be grown at all.

Also, as increasingly more crops are used for biofuels it is likely that there will be increasing demand for land, and most compulsory set-aside will revert back to use for biofuels. Government and the EU will also come under increasing pressure to reduce the level of set-aside or remove it altogether.

Much of the set-aside in the UK is non-rotational on poor yielding or logistically challenging land, or used in environmental stewardship schemes. Much of this may only be brought back into production under a favourable economic climate. Under the reformed CAP with the Single Farm Payment (SFP) there is no requirement now for fields to be cropped, so if economic conditions are not suitable then land would be left fallow. This means that the issue of the appropriate reference system may be seen as changing with the prevailing economic conditions. At low crop prices the reference land-use of UK set-aside might be appropriate, however at higher crop prices when all land is in production then cropped land will provide the more appropriate reference standard. The consequences of higher crop prices are likely to result in both intensified production and higher yields (both in UK and elsewhere) and an increase in cropped area on to more marginal lands elsewhere.

Where set-aside is currently used in environmentally useful situations such as headland margins/beetle banks, there will be opposition to a return to cropping if this is perceived to be the result of growing crops for biofuels. Aside from the potential CO<sub>2</sub> implications of mineralised stored soil organic carbon, this could potentially result in a loss of biodiversity.

Using set-aside as a reference system may be misleading. It may be argued that allocating all the crop input GHG costs (especially N fertiliser and N<sub>2</sub>O emissions) to the biofuel crop is unfair, as these costs would be incurred whether the crop is used for biofuel or for food. Using set-aside as a reference system, on paper it may appear as though a switch to biofuels in the UK would result in large increases in GHG emissions from fertiliser and N<sub>2</sub>O emissions, where in fact there is likely to be little change from current emissions from food production within the UK.

### **3.3.2 Food crops as a reference system**

Given that, at least in the initial stages, the grain or oilseeds used for biofuels will be traded on the commodity markets, and that it is possible that growers will grow these crops not being sure whether they are destined for food use or for biofuel use, the most realistic reference system would be to assume that the crop was grown for food use rather than biofuel use. However, adopting this approach causes serious complications. If it is assumed that growing crops for biofuels displaces crops for food, and that the amount of food required globally is relatively stable or increasing, then the next question is how and where is this food going to be produced elsewhere? As food commodities are traded globally this is a very difficult question to answer, as the 'replacement' food could be produced anywhere in the world under any number of different production methods. Considerable concern has been expressed recently over this issue (e.g. Searchinger et al., 2008), for example, in relation to palm oil and destruction of Asian rainforests; even if biofuel crops are explicitly barred from being grown on land that was previously un-cropped, it will not stop previously un-cropped land being used for food crop production.

To generate a reliable estimate of GHG emissions from the reference system would potentially require life-cycle analyses for each food commodity in each production method in each country of the world. Such an analysis has not been attempted to date and such a task would be hugely complex.

These issues are likely to be of increasing importance in the debate over food vs energy. Given that the overall aim is to reduce GHG emissions globally the ultimate solution might be for carbon reporting on all land uses/ end products. Major food processors and retailers (e.g. Walkers, Tesco) are now beginning to think of the GHG costs of their products in order to allow 'carbon labelling' for consumers. Such analyses require the GHG costs of agricultural production to be calculated and allocated in a fair, transparent and consistent manner. It is possible that approaches such as the GHG calculator for biofuels could be used as a forerunner for these uses in allowing calculations to be made and informing grower decisions. Indeed, a methodology for 'carbon footprinting' products, including food products, is being developed by Defra, British Standards Institute and the Carbon Trust as Publicly Available Specification (PAS) 2050<sup>1</sup>.

In conclusion, whilst set-aside is a tolerable reference system for OSR in the short term, it is less so for wheat and will rapidly become inappropriate as the area of biofuel crops expands. As such, it is best to calculate the GHG emissions from cropping on an absolute basis, in terms of kg CO<sub>2</sub> eq/ha and apply any credit for alternative land use subsequently.

### **3.4 Allocation procedures for co-products**

The issue of joint products in life cycle assessment is a major consideration for most biofuels since their production usually involves the generation of one or more co- or by-products. For example, the production of biodiesel from oilseed rape (OSR) results in the simultaneous production of rape straw (mainly treated as a waste product), rape meal (usually sold as animal feed) and glycerine (usually sold as a chemical feedstock). The production of bioethanol from wheat also involves the production of wheat straw (often sold for animal bedding, plant mulch, strawboard manufacture, etc.), bran (possibly recovered and sold to the food industry) and distillers' dark grain solubles (DDGS; usually sold for animal feed). Obviously, the primary energy inputs and greenhouse gas emissions associated with biofuel

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<sup>1</sup> See Defra Project FO0404 "Scenario building to test and inform the development of a BSI method for assessing GHG emissions from food Scenario building to test and inform the development of a BSI method for assessing GHG emissions from food".

production need to be divided amongst such joint products. If this did not occur and all the primary energy inputs and greenhouse gas emissions were allocated to the main product (biodiesel or bioethanol), then misleading consequences arise. In particular, results would suggest that the production and subsequent use of co- and by-products are “free” of any impact on energy resource depletion and global climate change, because their primary energy inputs and greenhouse gas emissions would be recorded as zero. This is clearly incorrect.

The most likely uses for the rapemeal and DDGS co-products from biofuels are as animal feeds, or as biomass for combustion to provide electricity and heat. HGCA Research Review 66 (Cottrill et al., 2007) has examined the potential for rapeseed meal and DDGS to be used as animal feeds in the UK. Their use in ruminant and non-ruminant diets could displace other protein sources such as soyabean meal. In this case the GHG emissions associated with the production and transport of the alternative protein feeds replaced could be discounted.

The selection and application of allocation procedures in life cycle assessment is intended to address the issue of joint production. Unfortunately, there is more than one theoretical way of undertaking allocation. Additionally, there are no firm rules recommended, officially, on how different possible allocation procedures are chosen (Anon 2006). These allocation procedures include the use of substitution credits, and allocation by mass, energy content, etc., and price. In general, the use of substitution credits is favoured by many life cycle assessment practitioners. This is followed, generally, by allocation by a given physical characteristic, such as mass, energy content, etc. Finally, allocation by price is often considered as the last remaining option after all else has failed.

The use of substitution credits has obvious attractions because it usually appears to be logically sound. It involves finding another product which might be displaced by the joint product under consideration. The comparable quantity of this other product is determined and the primary energy inputs and greenhouse gas emissions associated with its production are used to form “credits”. These “credits” represent the primary energy consumption and greenhouse gas emissions that have been avoided by the introduction of the joint product. In effect, the other product has been “displaced” and its production does not occur so that its primary energy inputs and greenhouse gas emissions never arise. However, there are two major problems with this approach to allocation. The first is how to choose the “displaced” product and the second is whether the primary energy inputs and greenhouse gases are actually “avoided”. The difficulties caused by this second problem are examined in more detail by means of examples later.

### ***3.4.1 Mass or energy-based allocation***

These concerns can be sidestepped by deciding to use allocation by a given physical characteristic such as mass, energy content, etc. Again, the attractions of these alternative allocation procedures disappear when the consequences are examined. The fundamental difficulty is which physical characteristic should be chosen from the many featured by joint products. Whilst mass has obvious advantages, its application can produce misleading results from life cycle assessment. For example, relatively large quantities of comparatively low economic value by-products can be derived from biofuel production. Typically, biodiesel production from OSR in the United Kingdom (UK) provides 2.782 tonnes of rape straw, 1.575 tonnes of rape meal and 0.100 tonnes of glycerine, giving a total of 4.457 tonnes of by-products for each tonne of biodiesel (Mortimer et al. 2003). Allocation by mass would, therefore, place the majority of the primary energy inputs and greenhouse gas emissions onto the by-products instead of the biodiesel which is the main output of this activity. The use of energy content as a basis for allocation does not avoid similar problems. Additionally, it can be argued that such a basis should only be used if the joint products are actually burnt. If this is the case, then the use of substitution credits, determined using the “avoided”

primary energy inputs and greenhouse gas emissions of “displaced” fuels and electricity, would seem more appropriate. This then re-introduces the problems of allocation through substitution credits discussed earlier.

Whether substitution or allocation is used can have significant and material impacts on the final GHG ‘answer’, and also on the type of behaviour which is incentivised. For example, where co-products are used for energy by combustion, then allocation by energy content can give much lower benefits to the biofuel than using substitution whereby large credits for displaced electricity are applied to the biofuel. Some people may have the view that what happens to the co-products should be irrelevant to the GHG costs of the biofuel, as would be the case with allocation. This does have material implications because if the benefits of burning co-products are over-stated it is more likely that they will be combusted, so they will be less available as animal feeds, which may actually be a more appropriate use for them. It is arguable whether a biofuel produced with its co-products used as an energy source should be given a better GHG rating than a biofuel produced using other biomass resources (e.g. short rotation coppice) as an energy source (after discounting specific GHG costs of producing/supplying the biomass).

### **3.4.2 Price-based allocation**

Allocation by price is not too complicated provided the quantities and prices of all joint products are known. However, life cycle assessment practitioners are often reluctant to apply this allocation procedure for a number of apparently fundamental reasons. First, not all joint products may have a market price. Instead, they may be transferred within the same production process to be converted into a product which can actually be sold. However, this is not a real problem since it is possible to derive a “shadow” price based on the eventual market price of the joint product and relationship between this and the initial joint product. Second and probably more fundamentally, allocation prices are not attractive because relative market prices change over time and this has the effect of causing the results of life cycle assessment to vary accordingly. Most life cycle assessment practitioners and decision-makers who use their results are uncomfortable with this. In contrast, allocation by physical characteristics produces results which are as fixed as the process under consideration. This only changes if technology changes. Additionally, it is tempting to believe that allocation using substitution credits generates results which are fixed over time. However, a little thought about how the “avoided” primary energy inputs and greenhouse gas emissions of the “displaced” product can change with time should dispel this conclusion.

To those outside life cycle assessment, the recommendation of choosing one allocation procedure from a collection of apparently imperfect options could imply that there is subjectivity and potentially arbitrary bias at the heart of this technique. Hence, a better solution is required than the current approach to allocation. Obviously, this needs to be based on a clear guiding principle which is wholly objective. It is proposed that this principle is that the allocation procedure which is applied produced results from life cycle assessment that reflect reality. Again, as with the issue of reference systems for alternative land use, this comes back to the concept of setting up the calculations involved in life cycle assessment so that they actually answer the specific question under consideration. As with the previous discussion of alternative land use, the framework of “comparative reference systems” may be used to provide a sound basis for resolving the issue of joint product allocation.

As an example, the production of biodiesel from OSR in the UK will be considered (Mortimer 2006b). For illustration, it will be assumed that the by-product of rape meal displaces soya meal which is derived from soya beans grown in Australia. It should be noted that substitution of soya meal is a common assumption in life cycle assessment studies when addressing allocation involving joint products that are sold as animal feeds. However, soya meal is itself a by-product from the production of soya oil. Solving allocation for both rape

meal (and biodiesel) and soya meal (and soya oil) simultaneously is mutually exclusive and illogical. Hence, it is not possible to designate primary energy and greenhouse gas emission credits for soya meal that can be used with rape meal.

Logically, if rape meal actually does displace soya meal as an animal feed, then all the primary energy inputs and greenhouse gas emissions associated with the cultivation, harvesting and processing of soya beans will be allocated to soya oil as the only product. Under such circumstances, soya meal becomes a waste product. For completeness, it would be necessary to evaluate the primary energy inputs and greenhouse gas emissions associated with the disposal of soya meal. If, for simplicity, it is assumed that these are negligible, then there is no overall reduction in total primary energy inputs and greenhouse gas emissions. In effect, this means that the “substitution credits” for soya meal are zero (and not positive and significant values, as usually assumed). In passing, it should be noted that the substitution credits for soya meal that are often cited are derived from allocation by mass or price (see, for example, Beer 2001). Such an approach is hardly consistent.

When establishing comparative reference systems it is essential to ensure that they specify “before” and “after” situations which are completely “balanced”, that is, inputs of fixed natural resources, such as land, and outputs, in the form of required products and services, are equal on both sides. Once this foundation has been established, meaningful evaluation of relative primary energy inputs and greenhouse gas emissions can be attempted. However, the displacement of existing products, such as soya meal, by new by-products, such as rape meal, must be addressed carefully. This is because, weight-for-weight, they are not comparable animal feeds. This issue is normally handled in life cycle assessment by specification of the “functional unit” which describes the characteristic purpose of a product or service. Different animal feeds have different nutritional properties and this has to be taken into account.

Although the comparison of animal feed depends on many factors, including the nature of the animals themselves, the simplest basis is to use metabolisable energy content. Using such data, it can be calculated that 1.0 kilogram of rape meal would displace 0.9 kilograms of soya meal as animal feed. A summary of equivalence ratios, relative to rape meal, for typical animal feeds available in the UK is provided in Table 7 which also contains some estimates for the total greenhouse gas emissions credits of a selection of animal feeds. It should be noted that any differences in methane emissions from animals which consume different feeds are not accounted for here but are becoming the topic of intensive research elsewhere. It will also be seen that the total greenhouse gas emissions credits for straight feeds which are by-products are all recorded as zero, in keeping with the preceding discussion.

It can be seen in Table 7 that there is a considerable range of credits that could be chosen for the evaluation of rape meal as an animal feed by-product of biodiesel production from OSR. Even discounting the values assigned to straight feeds which are themselves by-products, the possible total greenhouse gas emissions credit varies from 287 to 1,387 kg CO<sub>2</sub> eq. per tonne of rape meal. This excludes values for some animal feeds for which results are not currently available. The question that this diversity of possible credits raises is “which value should be chosen and applied in practice?” The simple answer is that it should be the animal feed the by-product rape meal actually displaces. However, substantial market analysis would be needed to determine this and, even if a reliable conclusion could be formed, it would change over time as markets develop and fluctuate. The obvious way of avoiding such complex analysis is to recognise that relative market prices are, despite their imperfections, the best guide to how one product competes with and displaces others. In essence, market prices are regarded as the arbiters of the balance of supply and demand which is exactly what is required for automatically incorporating implicit decisions over the choice of displaced products into allocation.



**Table 7: Typical UK Animal Feeds, Equivalence to Rape Meal and Estimated Total Greenhouse Gas Emissions**

Animal Feed	Total Greenhouse Gas Emissions Credits <sup>(a)</sup> (kg eq CO <sub>2</sub> /t)	Equivalence to Rape Meal (t/t rape meal)	Total Greenhouse Gas Emissions Credits <sup>(a)</sup> (kg eq CO <sub>2</sub> /t rape meal)
<b>Cereals</b>			
- barley (crushed)	514	0.95	488
- barley (pellets)	538	0.95	511
- maize	1,127	0.89	1,003
- oats	287	1.00	287
- wheat	376	0.92	346
<b>Forages</b>			
- forage maize	424	3.27	1,387
- grass silage		4.00	
- hay		1.44	
- pasture		5.22	
- whole crop wheat		2.57	
<b>Roots</b>			
- fodder beet		5.00	
- potatoes	163	4.06	662
<b>Straight Feeds (main products)</b>			
- field beans		0.92	
- lupin seed meal		0.85	
- maize gluten feed		0.96	
- wheat feed		1.09	
<b>Straight Feeds (by-products)</b>			
- corn molasses	0	1.13	0
- distillers' barley grains	0	0.98	0
- distillers' maize grains	0	0.86	0
- distillers' wheat grains	0	0.89	0
- dried citrus pulp	0	0.95	0
- dried molassed sugar beet pulp	0	0.96	0
- palm kernel meal	0	1.05	0
- soya meal	0	0.90	0
- sunflower meal	0	1.20	0

Note

(a) Assuming global warming potentials of 23 kg eq CO<sub>2</sub>/kg CH<sub>4</sub> and 296 kg eq CO<sub>2</sub>/kg N<sub>2</sub>O.

Allocation by price also produces a value for the total greenhouse gas emissions associated with the actual production processes involved in the generation of a by-product. However, more detailed investigation is required before allocation by price could be used as the main or only means for joint product allocation. In particular, it would be necessary to demonstrate that the outcomes of allocation by price coincide with changes in total greenhouse gas emissions that occur in reality. If justified, a consequence of this would be acceptance that the results from life cycle assessment are not fixed but vary over time, or more correctly, depending on circumstances which reflect the question being posed.

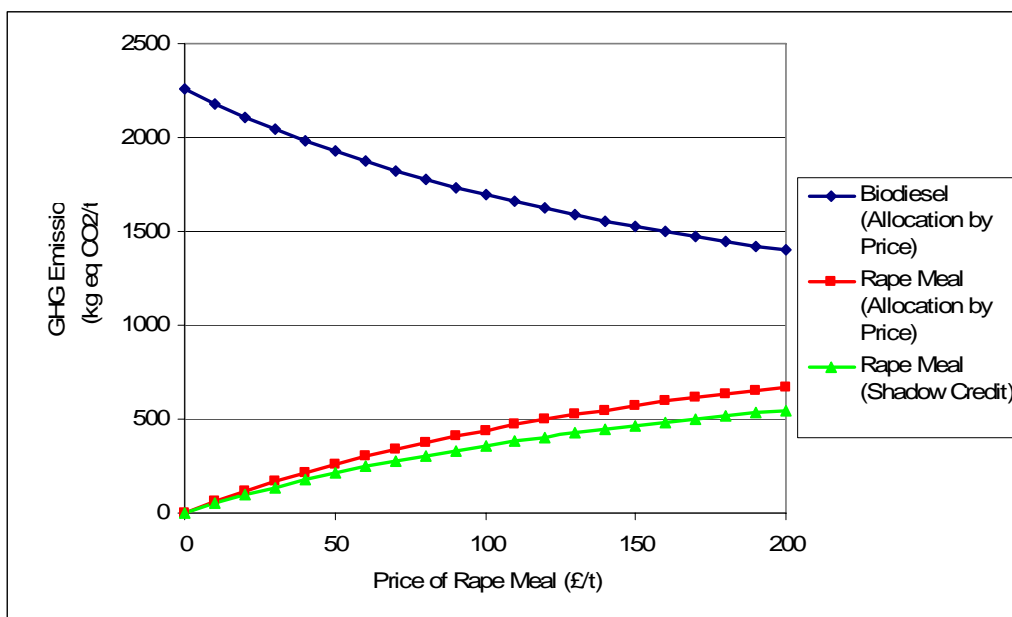
Preliminary investigation of this issue can be attempted by considering the joint production of biodiesel and rape meal from OSR in the UK. It is possible to evaluate the “shadow credit” for rape meal under these circumstances. This is defined as *the value of the substitution credit that would have to be applied with rape meal in order to achieve the same effect on total greenhouse gas emissions when adopting allocation by price*. Mathematically, the “shadow credit” is equal to the total greenhouse gas emissions of biodiesel production from OSR with a given price for rape meal less total greenhouse gas emissions when the price of rape meal is zero, divided by the amount of rape meal produced per tonne of biodiesel. This can be summarised as follows:

$$S_P = \frac{G_O - G_P}{Q}$$

- where  $S_P$  = “shadow credit” of rape meal when price of rape meal equals P (kg eq CO<sub>2</sub>/t rape meal)
- $G_O$  = total greenhouse gas emissions for biodiesel with price of rape meal equal to zero (kg eq CO<sub>2</sub>/t biodiesel)
- $G_P$  = total greenhouse gas emissions for biodiesel with price of rape meal equal to P (kg eq CO<sub>2</sub>/t biodiesel)
- $Q$  = quantity of rape meal produced per unit biodiesel (t rape meal/t biodiesel)

Using existing data (El Sayed et al. 2003), values for the “shadow credit” for rape meal can be derived for different values of the price of rape meal. It is assumed that 1.575 tonnes of rape meal are produced per tonne of biodiesel. The variation in the “shadow credit” with the price of rape meal is illustrated in Figure 1. For comparison, the variations of the total greenhouse gas emissions for biodiesel and rape meal, determined using allocation by price, are also included in Figure 1.

It must be appreciated that the meaning of the “shadow credit” for rape meal is quite different from the total greenhouse gas emissions for rape meal derived using allocation by price. It will also be noted that these values diverge increasingly as the price of rape meal increases. However, this divergence is proportionally small for current values of rape meal prices and within the range of the likely accuracy of estimated total greenhouse gas emissions.



**Figure 1: Variation of the Shadow Credit for Greenhouse Gas Emissions of Rape Meal with the Price of Rape Meal(a)**

Note

- (a) Biodiesel production from OSR in the UK assuming conventional agriculture, processing with heat from a natural gas-fired boiler and imported electricity, joint product allocation by price (rape straw = £25/t, raw rapeseed = £152/t, rape meal – base case = £84/t, crude rapeseed oil = £323/t, biodiesel = £268/t and glycerine = £388/t) and global warming potentials of 23 kg eq CO<sub>2</sub>/kg CH<sub>4</sub> and 296 kg eq CO<sub>2</sub>/kg N<sub>2</sub>O (El Sayed et al. 2003).

The “shadow credit” can be compared with the total greenhouse gas emissions avoided by displacing different animal feeds. The base case price for rape meal of £84/tonne, the resulting greenhouse gas emissions credit for rape meal is 317 kg CO<sub>2</sub> eq /tonne, respectively. With allocation by price, this suggests that the “shadow credit” for rape meal is similar to the avoided greenhouse gas emissions somewhere between the displacement of oats and wheat as animal feeds (see Table 7). If the price of rape meal was zero, this would suggest that it is, in effect, a waste product and that, quite correctly, all the greenhouse gas emission were being allocated to the biodiesel (strictly speaking, to the combination of rape straw, glycerine and biodiesel). Incidentally, a “shadow credit” of zero implies that the rape meal displaces any straight feed which is derived as a by-product. If the price of rape meal rose to double the assumed base case price, then the “shadow credit” of rape meal would be 498 kg eq CO<sub>2</sub>/tonne. This is equivalent to the avoided greenhouse gas emissions for the displacement of crushed barley as an animal feed.

Whether these outcomes are correct would depend on market analysis required to determine which existing animal feeds the rape meal would actually displace. However, for the time being, it is tempting to speculate that allocation by price, in this instance, produces avoided greenhouse gas emissions which are consistent with rape meal displacing a combination of different animal feeds. It should be noted that, in order to achieve “shadow credits” equivalent to maize and forage maize in the UK, considerably higher (possibly unrealistic) rape meal prices would have to apply than those under current circumstances.

Although significant further examination of this issue is required, some tentative conclusions can be drawn from this consideration of joint product allocation procedures and summarised as follows:

- the guiding principle and ultimate test of any allocation procedure is that it should accurately reflect changes that actually happen or are likely to happen in the real world,
- allocation by means of physical characteristics, such as mass, energy content, etc., is rarely appropriate and should only be used, with caution, in very special circumstances,
- allocation using substitution credits is deceptively simple and attractive but can have major drawbacks which become apparent when actual consequences are considered carefully and properly by applying comparative reference systems,
- comparative reference systems must be established to reflect actual changes in the real world and the specific question under consideration by any given life cycle assessment,
- any substitution must be based on appropriate functional units for the products and services under consideration,
- substitution credits for by-products which displace other by-products are zero,
- allocation by price is the most realistic and practical means of addressing joint products in life cycle assessment provided that there is confidence that relative market prices reflect actual changes in the real world,
- it is necessary to examine circumstances where relative market prices do not reflect actual changes in the real world, such as the occurrence of market failures, market distortion, etc.,
- shadow prices, based on careful evaluation, should be adopted for any products and services that are not traded outside the process(es) under consideration, and
- both life cycle assessment practitioners and their audiences need to accept and appreciate that subsequent results can vary over time as both circumstances and the question under consideration change.

### ***3.4.3 Combustion of co-products***

Combustion in combined heat and power plants of co-products from crop production, and from the biofuel process, has been promoted as one of the key ways of improving the GHG balance of biofuels (Mortimer 2006b). As discussed above, the effect that combusting co-products has on the final GHG rating of the biofuel is heavily dependent on whether a substitution or allocation approach is adopted, with GHG benefits to the biofuel being great if a substitution methodology is used.

There may be a number of implications from an approach that incentivises use of co-products for combustion. Firstly, there may be considerable emissions of N<sub>2</sub>O from the combustion of crop residues or co-products, especially if the material has a relatively high N concentration.

It may also be considered that combustion doesn't necessarily represent the 'best' use of a co-product. The full implications for animal feed or alternative uses need to be considered. Where co-products have a high nutrient value, especially protein, it may be 'wasteful' to remove this from the nutrient cycle whereby benefits could be passed on to animals and subsequently returned to the land, this may be especially important for N fertiliser. Calculations currently also ignore the possible land use change implications of using co-products for animal feed or electricity production; the use of such co-products may help avoid land use change elsewhere if they are displacing the need for expanding cropped land or exacerbating deforestation.

As animal feed prices rise it may be considered increasingly inappropriate that relatively high nutrient value co-products such as DDGS and rapemeal are burnt for energy rather than used as animal feed.

If straw is continuously removed to provide energy via combustion there will be a consequent gradual decline in soil fertility and soil organic carbon. It may be necessary to account for these changes in soil organic carbon if the GHG savings from using straw as an energy source were included in biofuel accreditation calculations.

If straw is burnt to provide energy for a biofuels plant there is also an issue of apportionment; should the credit for displaced fossil fuel rest with the grower or with the biofuels processor? In reality it makes no difference in GHG terms whether the straw burnt in a processing plant comes from a biofuel crop or from a food crop. In such a case only a part of the *total* fossil fuel replacement can be said to be specific to biofuel production. Given the bulky nature of straw there would be distinct advantages in sourcing the straw from a closer catchment to the plant than for the grain. It may also make more sense for a different biomass feedstock than straw to be used altogether, especially rape straw. In-any-case, double-counting of credits from burning straw needs to be avoided.

## **4 Technical issues – improving the evidence base**

There are a number of issues of uncertainty that result from a lack of sufficiently detailed understanding, whereby methods of calculation, parameters or default values are uncertain. Of these, the use and fate of nitrogen has the biggest impacts on the overall emissions from cropping, and also over which the uncertainty is largest. These issues are outlined in the following sections.

### **4.1 GHG emissions from N fertiliser manufacture**

The manufacture of nitrogen fertiliser has significant implications for the cultivation of biomass energy crops, such as oilseed rape and wheat, in relation to their primary energy inputs and greenhouse gas emissions. The emissions from the manufacture of fertilisers result from the high energy requirements of the process to synthesise ammonia ( $\text{NH}_3$ ) and from nitrous oxide ( $\text{N}_2\text{O}$ ) emissions during the production of nitric acid ( $\text{HNO}_3$ ). In addition, fossil fuels (generally natural gas, but also heavy fuel oil or coal) are used as the feedstock source of the hydrogen needed to synthesise ammonia with the nitrogen being 'fixed' from the atmosphere. Having extracted the hydrogen, the carbon in these fossil fuels is discarded to the atmosphere as carbon dioxide.

Illustrative contributions from the use of nitrogen to the overall GHG emissions resulting from the production of biodiesel from oilseed rape and bioethanol from wheat in the United Kingdom are provided in Table 8. These contributions can be compared with those due to direct  $\text{N}_2\text{O}$  emissions from soils, as summarised in Table 10. It should be noted that the sum of the  $\text{N}_2\text{O}$  emissions from nitrogen fertiliser and from soils exceeds 100% because there are negative contributions from  $\text{N}_2\text{O}$  emissions from soils in the alternative land use reference system of maintained set-aside. These amount to -19% for  $\text{N}_2\text{O}$  emissions and - 8% for total greenhouse gas emissions in the production of biodiesel from oilseed rape, and -23% for  $\text{N}_2\text{O}$  emissions and -7% for total greenhouse gas emissions for the production of ethanol from wheat grain.

**Table 8: Relative GHG Contributions to Biofuel Production of Nitrogen Fertiliser Manufacture (Mortimer 2006b)**

Biofuel	Relative Contribution of Nitrogen Fertiliser Manufacture <sup>(a)</sup> (%)				
	Primary Energy Input	Carbon Dioxide Emissions	Methane Emissions	Nitrous Oxide Emissions	Total Greenhouse Gas Emissions <sup>(b)</sup>
Biodiesel from Oilseed Rape <sup>(c)</sup>	24	20	34	58	37
Bioethanol from Wheat Grain <sup>(d)</sup>	15	14	21	65	30

**Notes**

- (a) Average manufacture of ammonium nitrate fertiliser production in Western Europe in the 1990's.
- (b) Assuming global warming potentials of 23 kg eq CO<sub>2</sub>/kg CH<sub>4</sub> and 296 kg eq CO<sub>2</sub>/kg N<sub>2</sub>O.
- (c) Production of biodiesel from oilseed rape with conventional agriculture using a nitrogen fertiliser application rate of 196 kg N/ha.a and assuming average EU-15 soil emissions of 3.12 kg N<sub>2</sub>O/ha.a (JEC, 2006), processing using a natural gas-fired boiler and grid electricity, and rape meal used as an animal feed with all allocation by price.
- (d) Production of bioethanol from wheat grain with conventional agriculture using a nitrogen fertiliser application rate of 185 kg N/ha.a and assuming average EU-15 soil emissions of 2.23 kg N<sub>2</sub>O/ha.a (JEC, 2006), processing using a natural gas-fired boiler and grid electricity, and distillers' dark grains (DDGS) used as an animal feed with all allocation by price.

Most evaluations of the primary energy inputs and greenhouse gas emissions associated with the manufacture of nitrogen fertiliser assume that the ammonium nitrate is produced from natural gas. However, assessments have also been made for other forms of nitrogen fertiliser but are not considered further here (Kaltschmitt & Reinhardt 1997; Patyk and Reinhardt 1997). Standard results for ammonium nitrate fertiliser manufacture from natural gas are summarised in Table 9.

**Table 9: Primary Energy Inputs and Greenhouse Gas Emissions Associated with Ammonium Nitrate Manufacture (Elsayed et al. 2007)**

Nitrogen Fertiliser Manufacture	Primary Energy Inputs (MJ/kg N)	Carbon Dioxide Emissions (kg CO <sub>2</sub> /kg N)	Methane Emissions (kg CH <sub>4</sub> /kg N)	Nitrous Oxide Emissions (kg N <sub>2</sub> O/kg N)	Total Greenhouse Gas Emissions (kg eq CO <sub>2</sub> /kg N)
Ammonium Nitrate	40.74 ± 5.43	2.30 ± 0.26	0.012 ± 0.001	0.015	6.93 ± 0.26
Kg CO <sub>2</sub> eq/kg N		2.30	0.28	4.44	

#### **4.1.1 CO<sub>2</sub> emissions from fertiliser manufacture**

Ammonium nitrate fertiliser is often produced from natural gas which provides a source of hydrogen which is reacted with nitrogen to obtain ammonia. Some of this ammonia is converted to nitric acid which in turn is combined with the remaining ammonia to produce ammonium nitrate. The ammonia production stage is the most energy-intensive. Data for production in Western Europe in the 1990's (Konshaug, 1998) indicate that it accounts for 95% of the primary energy inputs of ammonium nitrate manufacture. It is assumed that the primary energy inputs to ammonia production amount to 33.7 ± 4.7 MJ/kg NH<sub>3</sub> (Elsayed et

al. 2007). This can be compared with estimates from Canadian studies of between 29.7 and 37.5 MJ/kg NH<sub>3</sub> (CIPEC, 2004), and benchmarking data of between 28.0 and 53.0 MJ/kg NH<sub>3</sub> (Plant Surveys International, 2005).

The data incorporated into the results quoted in Table 9 are at the low end of observed primary energy inputs. It should also be noted that there is evidence to suggest that the energy consumption of ammonia plants based on the steam reforming of natural gas has approached the theoretical minimum in recent years (IFA, 2007).

#### **4.1.2 N<sub>2</sub>O emissions from fertiliser manufacture**

Whilst ammonia production is the most energy-intensive part of the nitrogen fertiliser manufacturing process, the greatest source of N<sub>2</sub>O emissions and also total greenhouse gas emissions is nitric acid production, accounting for almost 100% and 63%, respectively. This is due to routine leakages of N<sub>2</sub>O to the atmosphere in the nitric acid plants, which are the subject of considerable abatement activity.

GHG emissions from the production of nitric acid can range from around 3 to 12kg CO<sub>2</sub> eq per kg N (Wood & Cowie 2004). In the estimates presented here, the leakage rate is assumed to be 6.67 kg N<sub>2</sub>O/t HNO<sub>3</sub> (Konshaug 1998). It has been suggested that the average emissions per plant in Europe is 6 kg N<sub>2</sub>O/t HNO<sub>3</sub> (IFA 2007). The lowest emissions reported without abatement equate to 5 kg N<sub>2</sub>O/t HNO<sub>3</sub> for atmospheric pressure plants and the best high-pressure plants (Konshaug, 1998). Taking this value as the best currently achievable means that nitrous oxide emissions for ammonium nitrate fertiliser manufacture might be reduced from 0.015 kg N<sub>2</sub>O/kg N to 0.011 kg N<sub>2</sub>O/kg N (from 6.93 ± 0.26 to 5.84 ± 0.26 kg eq CO<sub>2</sub>/kg N). This represents a potential saving of 16%. Nitric acid provides 50% of the nitrogen content of ammonium nitrate.

Some manufacturing plants use N<sub>2</sub>O abatement technologies that could significantly reduce the N<sub>2</sub>O emissions in manufacture, although, due to plant design, this technology is not available to all manufacturing plants (Dawson, 2008). Yara claim that N<sub>2</sub>O abatement technologies installed on their plants reduce N<sub>2</sub>O emissions from nitric acid production by 70 to 90% i.e. to below 2.5 kgN<sub>2</sub>O/kg HNO<sub>3</sub> 'for most plants' (Jenssen, T, 2007). Emissions of 2.5 kg N<sub>2</sub>O/kg HNO<sub>3</sub> represent a 63% reduction from the 6.67 kg N<sub>2</sub>O/t HNO<sub>3</sub> used here. Because all the N<sub>2</sub>O emissions from ammonium nitrate manufacture result from nitric acid production a 63% reduction is also calculated for AN. Thus the N<sub>2</sub>O-derived GHG emissions resulting from AN production could be reduced from 4.44 to 1.67 Kg CO<sub>2</sub> eq/kg N. Overall AN GHG emissions would be reduced from 6.93 to 4.16 Kg CO<sub>2</sub> eq/kg N, by 40% overall.

Given that much of the total emissions of nitrogen fertiliser manufacture are due to N<sub>2</sub>O release, fertiliser products that don't involve nitric acid production tend to have lower associated emissions per unit of N. For example, urea has emissions of 2.9kg eq CO<sub>2</sub>/kg N (Jenssen & Konshaug 2003). There is an urgent need to establish the differences in GHG emissions from fertiliser manufacture across fertiliser types, manufacturers and countries if reliable climate change abatement advice is to be provided.

#### **4.1.3 Dealing with N fertiliser in accreditation schemes**

There are various issues that complicate the calculation of GHG emissions f

rom fertiliser manufacture (e.g. including credits for the export of excess energy; the capture of CO<sub>2</sub> in urea production) and not all LCAs of the process have been fully transparent or used compatible approaches (Wood & Cowie, 2004). These issues have recently been set out by the European Commission Joint Research Council (EC 2006). Overall, for ammonium nitrate most studies have used a total emissions factor of around 7kg CO<sub>2</sub> eq per kg N

(LowCVP 2004; Mortimer et al 2004, Woods et al, 2006). In reality, it is likely that total GHG emissions per kg of N may vary very considerably between fertiliser type (particularly urea versus ammonium nitrate) and the site of manufacture.

There are particular difficulties in calculating GHG emissions from N fertiliser use where a range of fertiliser products are used on farm. Whilst ammonium nitrate (AN) makes up the majority of N fertiliser applications in the UK, few farms in reality will apply the full N requirement of the crop as AN. Many early season applications are compound fertilisers and some farms use urea when the price is favourable. A significant minority of farms apply all their fertilisers in liquid form where the predominant N source is urea ammonium nitrate (UAN).

Given the potentially very large impact on the final GHG balance of small differences in emissions factors used, and the range in emissions between fertilisers used on-farm, it may make sense to allow the use of differing default emission factors within the GHG calculator, based on fertiliser type, country of origin, manufacturing company etc. where this can be satisfactorily demonstrated. Using specific emission factors for individual fertiliser products could potentially incentivise manufacturers to state the GHG emissions associated with their fertiliser products, and help continue the reduction in these emissions.

The choice of which fertiliser a farmer uses could potentially negate any other management factor. There is therefore an urgent need to assess the range in GHG emissions associated from the N fertiliser products available to growers in the UK.

The RTFO draft Carbon Reporting methodology (Department for Transport, 2007b) assumes emission factors for ammonium nitrate of 6.8kg CO<sub>2</sub>eq/kg N and 2.9 kg CO<sub>2</sub>eq/kg N for urea. A biofuel manufacturer might consider sourcing feedstocks solely produced using urea instead of AN due to its reduced GHG emissions, should the financial incentives be sufficiently large. Under these circumstances growers could switch to using urea as their main N fertiliser source.

These differences also penalise countries that predominantly use ammonium nitrate, such as the UK, against countries where urea is predominantly used. Incentivising the use of urea could have serious consequences in increased levels of agricultural ammonia emissions that would threaten the UK's commitment to the National Emissions Ceiling Directive (Defra, 2006), and could increase ammonia emissions globally.

With the current high demand for fertiliser products globally, it can be argued that global average GHG emissions for N manufacture should be used, as the consequences of using one product or another are immaterial in a situation where everything that can be produced will be used. Under this argument, switching between fertiliser products simply means the emissions would be incurred somewhere else, unless it encourages the expansion of more efficient fertiliser production at the expense of more polluting production.

## **4.2 In-field N<sub>2</sub>O emissions**

Of all the values that go into calculating the GHG emissions from biofuel crop production, the values used for N<sub>2</sub>O emissions from soils are perhaps the most uncertain. Because of the high global warming potential of N<sub>2</sub>O (296kg CO<sub>2</sub> eq/kg N<sub>2</sub>O) its emissions can represent over 35% of the total GHG costs of crop growth, equivalent to the GHG emissions associated with N fertiliser manufacture. Together, emissions in-field and from fertiliser manufacture, can cause about 70% of GHG emissions from cropping. Contributions to total N<sub>2</sub>O emissions and total GHG emissions by soils are shown in Table 10 below.



**Table 10: Illustrative Relative Contributions to Biofuel Production of Nitrous Oxide Emissions from Soils (Mortimer 2006b)**

Biofuel	Relative Contribution of Nitrous Oxide Emissions from Soils <sup>(a)</sup> (%)	
	Nitrous Oxide Emissions	Total Greenhouse Gas Emissions <sup>(b)</sup>
Biodiesel from Oilseed Rape <sup>(c)</sup>	61	27
Bioethanol from Wheat Grain <sup>(d)</sup>	53	17

**Notes**

- (a) Based on average EU-15 soil emissions (Edwards et al. 2006).
- (b) Assuming global warming potentials of 23 kg eq CO<sub>2</sub>/kg CH<sub>4</sub> and 296 kg eq CO<sub>2</sub>/kg N<sub>2</sub>O.
- (c) Production of biodiesel from oilseed rape with conventional agriculture using a nitrogen fertiliser application rate of 196 kg N/ha.a and assuming average EU-15 soil emissions of 3.12 kg N<sub>2</sub>O/ha.a (Edwards et al. 2006), processing using a natural gas-fired boiler and grid electricity, and rape meal used as an animal feed with all allocation by price.
- (d) Production of bioethanol from wheat grain with conventional agriculture using a nitrogen fertiliser application rate of 185 kg N/ha.a and assuming average EU-15 soil emissions of 2.23 kg N<sub>2</sub>O/ha.a (Edwards et al. 2006), processing using a natural gas-fired boiler and grid electricity, and distillers' dark grains (DDGS) used as an animal feed with all allocation by price.

Soil-N<sub>2</sub>O emissions are principally produced biologically via nitrification and denitrification. Nitrification is an aerobic process, which is most strongly controlled by the availability of ammonium-N and oxygen (Firestone and Davidson, 1989). Denitrification is an anaerobic process and is more temporally and spatially variable than nitrification (Bouwman et al. 2002). It is most strongly controlled by carbon, nitrate-N and oxygen availability (Firestone and Davidson, 1989). Intense anaerobic conditions may, however, result in decreased N<sub>2</sub>O emissions due to the conversion of N<sub>2</sub>O to N<sub>2</sub>. Both processes are also greatly influenced by temperature, moisture and pH. Whilst denitrification is deemed to be the major source of N<sub>2</sub>O emissions, nitrification can also play a significant role (Martens, 2005; Wrage et al., 2001)

#### **4.2.1 IPCC methodology for quantifying agricultural GHG emissions**

N<sub>2</sub>O emissions from agricultural soils are quantified on a national basis and reported as part of the national emissions inventory. The methodology used for national reporting purposes has been developed by the Intergovernmental Panel on Climate Change (IPCC). This methodology was first set out in 1996, but considerable revisions have recently been made within the 2006 Guidelines. This needs to be borne in mind when considering LCA studies that use the 1996 guidelines e.g. Williams et al., 2006a.

The 2006 guidelines give three 'tiers' of approach that can be used for national reporting; Tier 1, using default emission factors; Tier 2, using country specific emission factors and Tier 3, using a validated modelling approach such as the DeNitrification DeComposition (DNDC) model ([discussed later](#)). To date, although a few countries (e.g. Germany) are using DNDC for reporting purposes, most countries including the UK, are reporting under Tier 1 or Tier 2. This approach quantifies available sources of N (i.e. fertilisers, organic manures, sewage sludge, compost etc, crop residues, mineralisation of soil organic matter (SOM) through changes in land-use or management, grazing animals and organic soils) and uses emission factors to estimate the direct and indirect (losses from subsequent volatilisation or leaching) emissions of nitrous oxide. As this is the method used for official reporting of UK N<sub>2</sub>O emissions, it makes some sense for emissions calculated for biofuels reporting purposes to

be compatible with this. However, the IPCC approach is designed for country-level and not farm-level reporting.

The IPCC 2006 method for calculating N<sub>2</sub>O emissions for arable soils calculates the total amount of N inputs to the soil from inorganic fertilisers, organic N (animal manures, sewage or compost), crop residues and changes in soil organic matter, and multiplies this by an emission factor (0.0125 in 1996 guidelines, revised to 0.01 in 2006 Guidelines) to give 'direct' N<sub>2</sub>O-N emissions. Countries can use the default values (Tier 1 approach), or with sufficient evidence, they can use country specific parameters and emission factors (Tier 2 approach).

In addition, 'indirect' emissions are calculated to account for subsequent N<sub>2</sub>O-N emissions from ammonia volatilised and deposited elsewhere or nitrate leached from the original N inputs of N fertilisers, organic N, crop residues and changes in soil organic matter. To calculate emissions from volatilisation the total N input from fertiliser and manure is multiplied by the fraction that is volatilised (0.1 for inorganic fertiliser, 0.2 from organic N fertilisers) and this is multiplied by a separate emission factor (0.010). Emissions from leaching/run-off are calculated similarly. Total N from all sources is multiplied by the fraction leached (default is 0.3, range is 0.1 – 0.8) and a separate emission factor for N<sub>2</sub>O-N emissions from leached N (0.025 in 1996 guidelines, updated to 0.0075 in 2006 guidelines). Converting from kg N<sub>2</sub>O-N to kg N<sub>2</sub>O simply requires multiplying by 44/28 (the molecular weight of N<sub>2</sub>O as a proportion of the weight of N in N<sub>2</sub>O-N).

**Table 11: Calculating N<sub>2</sub>O emissions for arable soils using the 2006 IPCC Guidelines, with values from 1996 Guidelines for comparison**

	Values		Range		units
	1996	2006	Lower	Upper	
EF1 (Emission factor for N additions (fertilisers, manures, crop residues and changes in SOM))	0.0125	0.01	0.003	0.03	kg N <sub>2</sub> O-N /kg N
EF2 (Emission factor for organic soils (temperate))		8	2	24	kg N <sub>2</sub> O/ha
EF4 (Emission factor from N volatilised and re-deposited)	0.01	0.01	0.002	0.05	kg N <sub>2</sub> O-N/ kg N
EF5 (Emission factor from N leached)	0.025	0.0075	0.0005	0.025	kg N <sub>2</sub> O-N/ kg N
FracGASF (Volatilisation from synthetic fertiliser)	0.1	0.1	0.03	0.3	Proportion
FracGASM (Volatilisation from organic addition)	0.2	0.2	0.05	0.5	Proportion
FracLEACH (fraction of N leached)	0.3	0.3	0.1	0.8	Proportion
<i>Assuming 185kg/ha applied as N fertiliser:</i>					
Direct Emissions from fertiliser	3.63	2.91	0.872	8.721	kg N <sub>2</sub> O/ha
Indirect emissions from volatilization	0.291	0.291	0.017	4.36	kg N <sub>2</sub> O/ha
Indirect emissions from leaching	2.18	0.654	0.015	5.81	kg N <sub>2</sub> O/ha
Total in-field emissions	6.11	3.85	0.904	18.9	kg N <sub>2</sub> O/ha
	1810	1140	280	5600	kg CO <sub>2</sub> eq/ha.a

## **4.2.2 Alternative approaches to in-field N<sub>2</sub>O emissions**

The IPCC methodology has a number of well-recognised short-comings. It has been developed to allow emissions to be calculated on a national basis for each country in the world. As such, the approach, values used and emission factors may not be appropriate for use at the farm or field levels. Also, the approach doesn't fully account for some key factors that are known to affect N<sub>2</sub>O emissions, such as soil type, climate/weather, crop type, type and timing of N application, C:N ratio of crop residues, etc. These issues therefore make it difficult to use the IPCC approach to establish farm-level greenhouse gas emissions accurately.

However, simulation models *can* be used to predict N<sub>2</sub>O emissions from agricultural soils in different regions and under different scenarios. The DNDC (DeNitrification DeComposition) model uses information on soil properties and weather data to calculate the soil moisture and soil temperature profile of a soil. The inclusion of crop growth and N uptake; decomposition, nitrification, ammonia (NH<sub>3</sub>) volatilisation & CO<sub>2</sub> production from organic matter; denitrification of nitrate to nitrite (NO<sub>2</sub><sup>-</sup>), nitric oxide (NO), N<sub>2</sub>O and di-nitrogen (N<sub>2</sub>) and the growth and death of microbial denitrifier populations (Li et al., 1992a,b) allow, site specific daily N<sub>2</sub>O emissions to be simulated.

The DNDC model has been used by the Joint Research Centre (JRC) to produce a database model (Greenhouse Emissions from Agricultural Soils in Europe, termed GREASE) to predict N<sub>2</sub>O emissions from a typical wheat crop in the UK, using soils and climate data from 127 UK regions (see Appendix 2 of LowCVP 2004). LowCVP (2004) produced a 'representative' figure of 4.36 kg N<sub>2</sub>O/ha for an average UK wheat crop which has been included in the GHG balance of subsequent LCA (LowCVP 2004; Mortimer 2004; Woods et al. 2005; Mortimer & El Sayed 2006). This is comparable to the figure using the IPCC method (2006) of 4.08kg N<sub>2</sub>O/ha assuming 196kg N/ha applied, but less than the 6.15kg N<sub>2</sub>O/ha if emissions from crop residues are included. Revised calculations for the EU-15 by CONCAWE (Edwards et al, 2006) give net emissions of 2.23kg N<sub>2</sub>O/ha for wheat and 3.12kg N<sub>2</sub>O/ha for OSR after subtracting simulated equivalent emissions from unfertilised grass as a set-aside reference land use.

The values used by these previous studies fall within the range of emissions from experimental measures from agricultural soils in the UK. Recent work (Smith et al., 2006) gave total emissions from wheat fields between 1.8kg N<sub>2</sub>O/ha and 5.3 kg N<sub>2</sub>O/ha. Using a UK adapted version of DNDC, Brown et al (2002) calculate a figure of 1.6 kg N<sub>2</sub>O/ha for cereals, corrected for background emissions of around 1.7kg N<sub>2</sub>O/ha, giving a total of 3.3kg N<sub>2</sub>O/ha.

The DNDC model used in the previous studies (LowCVP 2004; Edwards et al 2006) is fundamentally based on the DNDC model parameterised for USA conditions but using regional UK and EU soils and climate data. Important differences in the modelling of N<sub>2</sub>O emissions between countries have been demonstrated, particularly with regard to process parameters such as soil carbon (Brown et al. 2002). A version of the DNDC model has been developed by the Institute of Grassland and Environmental Research (IGER) to specifically investigate UK conditions and named UK-DNDC (Brown et al., 2002). There is thus a need to use the UK-DNDC model in order to assess likely N<sub>2</sub>O emissions from wheat/oilseed rape production in the UK.

It is unclear whether the figures produced in the previous studies (LowCVP 2004; Edwards et al., 2006) fully include indirect and direct emissions. Whilst indirect emissions from leaching are accounted for by Edwards et al. (2006), it does not seem that emissions from volatilisation are included. Prior to the 2006 updated IPCC guidelines there has been considerable uncertainty over the emission factor from leached-N, this has now been reduced from 0.025kg N<sub>2</sub>O-N/kg N leached to 0.0075kg N<sub>2</sub>O-N/kg N leached. The IPCC

defaults for fraction of N leached is 0.3; this may be considered high compared to field experience in the UK; Brown et al (2002) calculate mean leaching across UK soils and crops to be 0.14 using UK-DNDC.

In the previous studies using DNDC model, N fertiliser and manure additions are not differentiated by region or soil type, leading to assumed high fertiliser N rates to organic soils which in reality would receive very little fertiliser – in this case on organic soils the proportion of N fertiliser lost as  $N_2O$  is modelled as being very high. This was made workable by introducing limits to the minimum and maximum proportion of N that could be lost as  $N_2O$  (LowCVP 2004), but this may have introduced bias to the values it gives. Both this study and the later CONCAWE study (Edwards et al., 2006) have emphasised the importance of soil organic content (SOC) to the simulated  $N_2O$  emissions. DNDC has been shown to be very sensitive to SOC (Brown et al 2002). Whilst SOC undoubtedly plays an important role in determining  $N_2O$  emissions, it is possible that its importance in describing the real differences between fields has been overestimated in these studies, especially in the UK where soils with high SOC are relatively rare.

Whilst the DNDC model is useful for coming up with a 'realistic' average emission factors for a given region, it would be difficult to use directly in a certification system. Such accounting systems need to concentrate on factors over which the grower has some control by ensuring that are realistically accounted for. The most important factor will be the amount of N added to the land, principally as N fertiliser, so it is entirely appropriate that this is the main driver of the calculation of  $N_2O$  emissions.

The UK-DNDC model needs to be used to assess  $N_2O$  emissions from arable soils in the UK at varying N levels for different soil types, different regions, weather etc. Such an analysis would indicate whether any of these factors are worth using as inputs to a GHG calculator, in order to modify the emission factors used. It should also be possible to use the UK-DNDC model to simulate different levels of N fertiliser on different soils and thus formulate a reasonable response of  $N_2O$  emissions to N-fertiliser over a range of soil available-N. It could thus be used to retrospectively check that the land-based GHG emissions being calculated from a certification system reflect actual net emissions to the atmosphere.

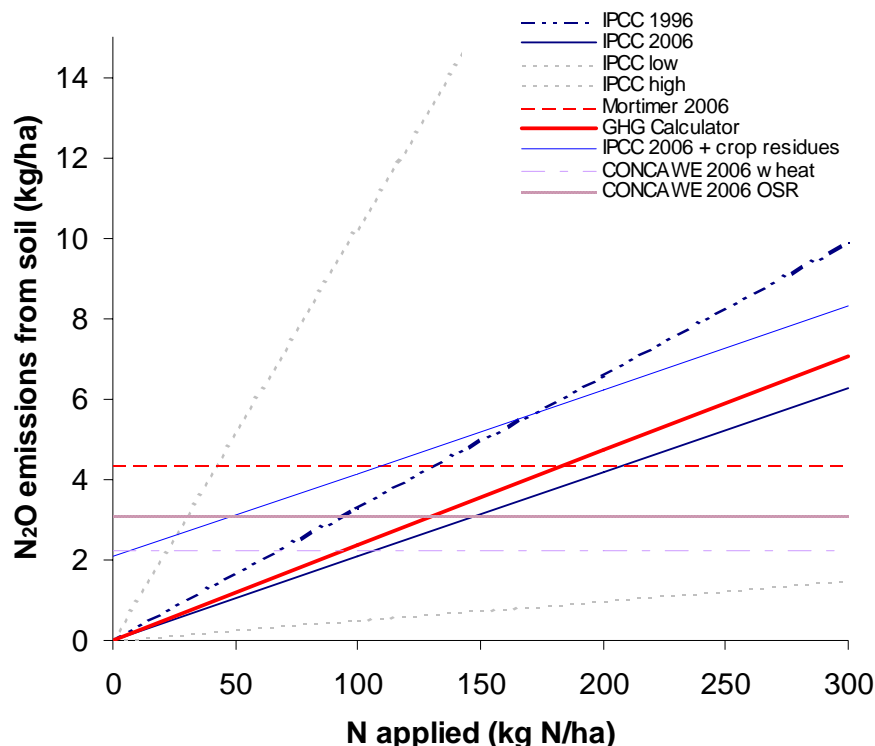


Figure 2: N<sub>2</sub>O emission factors calculated from IPCC guidelines and as used in previous studies (Mortimer et al. 2004; Woods et al. 2005; CONCAWE 2006)

### 4.2.3 Soil N<sub>2</sub>O emissions from N fertiliser and manure

Using the IPCC methodology, total direct and indirect emissions associated with N fertilisers are calculated as 0.021kg N<sub>2</sub>O/kg applied N using 2006 values (see Table 11). Assuming 185kg N fertiliser is applied, total emissions amount to 3.85 kg N<sub>2</sub>O/ha (2006 guidelines). This assumes no addition of organic nitrogen through manures, sewage etc, or from, crop residues. For comparison, if the 185kg N was supplied entirely by manure with no N fertiliser, N<sub>2</sub>O emissions would be 0.0224 kg N<sub>2</sub>O/kg N applied, giving total N<sub>2</sub>O emissions of 4.14 kg N<sub>2</sub>O/ha. The marginally higher figures from manure are due to greater indirect emissions from volatilisation.

Figure 2 shows these emission factors and how they compare to emission factors used in previous LCA studies (discussed in following section). The possible highest and lowest N<sub>2</sub>O emission factors from N fertiliser using the uncertainty range in default values listed for each of the parameters (IPCC 2006 Guidelines and presented in Table 11) are also plotted, these show graphically the very large range in uncertainty of N<sub>2</sub>O emissions. The calculations only deal with emissions from N fertilisers, emissions from crop residues and soil organic matter are dealt with in Sections 4.2.4 and 4.2.5 respectively, though the effect of including emissions from crop residues is also plotted in Figure 2.

#### 4.2.3.1 Relationship between N fertiliser and N<sub>2</sub>O emissions.

Given the importance of N fertiliser to grain yield, grain quality, input costs and given that it is probably the variable over which the grower has most control, it is important that the relationship used between N fertiliser and N<sub>2</sub>O emissions is realistic. It is possible that

growers could 'optimise' their N inputs to deliver an improved GHG balance (Kindred et al., 2007a).

In the current GHG calculator, to allow the soil N<sub>2</sub>O consequences of different N rates to be looked at, Woods et al. (2005) divided the figure of 4.36kg N<sub>2</sub>O/ha derived from DNDC by the average N application of 185kg N/ha to give an emission factor of 0.0236 kg N<sub>2</sub>O/kg N applied. However, this gives the somewhat unrealistic implication that with zero applied N fertiliser there will be zero N<sub>2</sub>O emissions. Using such a relationship in optimizing N rates for reducing GHG emissions from biofuels could be misleading, potentially overestimating the GHG consequences of reducing N applications.

The common approach taken in experimental N<sub>2</sub>O studies has been to assume linear emission factors for N<sub>2</sub>O emissions from fertilisers, based only on emissions at two N rates; nil-N and perceived optimum-N. However, recent results suggest that N<sub>2</sub>O emissions rise non-linearly with fertiliser applications (McSwiney and Robertson, 2005; Grant et al 2006;). This is being investigated for both grass and arable crops in the UK by a Defra project conducted jointly by ADAS and IGER – "An improved inventory of greenhouse gases from agriculture" (Defra Project AC0101).

It has been hypothesised that emission factors will increase as the amount of N applied increases to above that which the crop can use; i.e. emissions are likely to be proportionally greater at N rates above the crop optimum than below it, possibly rising exponentially (Kachanoski et al. 2003; Izaurralde et al. 2004; Bouwman et al. 2002). Grant et al. (2006) suggest that larger N<sub>2</sub>O emission factors should be used for N rates above the crops uptake capacity.

#### **4.2.3.2 N fertiliser decision making**

The optimum N for a crop will be dependent both on the crop requirement (yield potential) and the supply of N from the soil (Soil N Supply (SNS)). The amount of N available in the soil to begin with will depend on previous manure applications, N fertiliser applications to previous crops, crop residues, soil type and mineralisation of soil organic matter. Soil N supplies will also be high on highly organic soils.

In this case, the most important objective of carbon accreditation might be to ensure that growers are not over-fertilising. If a grower is applying 200kg N fertiliser to a very low SNS site with good yield potential it may be unfair for them to be penalised to the same extent as a grower applying 200kg/ha N fertiliser on a well manured site with an SNS of 200kg/ha and a lower yield potential. Potentially, a system that uses only N fertiliser as an input figure for N<sub>2</sub>O emissions could discriminate against low SNS/low organic matter soils as these would require more N fertiliser but may not necessarily have greater emissions at the optimum.

The difficulty is that N available in the soil is not routinely measured on every field. It can be estimated from previous cropping, soil type and excess winter rainfall as set out in RB209 (MAFF 2000) and this is considered sufficient for N recommendation purposes and for Nitrate Vulnerable Zone (NVZ) compliance. However, the accuracy of predicting SNS from previous cropping may not be considered reliable enough to use in calculating 'additional' GHG emissions for accreditation purposes.

Therefore, it is suggested that the IPCC relationship should be used for quantifying N<sub>2</sub>O emissions from N fertiliser i.e. 0.021kg N<sub>2</sub>O/kg applied N. This should be reviewed as new research is gathered, and disaggregated emission factors for different soil types, regions etc should be considered, possibly through studies using UK-DNDC.

In order to avoid the complex issues of estimating all N inputs, it may be possible to estimate N<sub>2</sub>O emissions using a measure of crop N offtake. This would give an assessment of how much N was available to the crop and it could be argued that it doesn't matter too much whether this N was from artificial fertilisers, manures, or from mineralisation etc. However,

this approach would also need an estimate of how much N has been applied and lost to the soil through over-fertilisation.

It is possible that a measure of grain N% could be used for this purpose. RB209 (MAFF 2000) and current HGCA work (HGCA Project 3084) seems to confirm that grain N% is relatively constant at around 2% at the optimum fertiliser rates. If grain N% is higher than about 2.3% it is indicative that the crop has received more fertiliser than is needed for yield. It might be possible to combine grain N% data with grain yield and N input data to give a fair estimate of N<sub>2</sub>O emissions, possibly utilising additional information such as soil type. Such a speculative approach would obviously need developing and validating with real data and peer review before it could be used in an accreditation system.

#### **4.2.3.3 N<sub>2</sub>O emissions from manures**

Ideally, it is necessary to account for all the N in the system that is susceptible to N-loss in considering N<sub>2</sub>O emissions. As well as N fertiliser, this means that an approach is required for dealing with manure/compost/sewage sludge application, N mineralised from organic soils, residual soil mineral nitrogen from the previous crop, and crop residues from the biofuel crop.

It can be argued that emissions from manures would occur anyway as if they are not spread on the biofuel crop they will be spread on another crop in the rotation. The amount of manure available for application nationally is finite and is dependent on supply from livestock production, not demands from crop production. Emissions from manure could therefore be considered as part of the livestock system, not part of the biofuel system.

However, the issue might become more complex if there is an alternative use for the manure, such as in anaerobic digestion or combustion. In this case some of the emissions may be avoided by the alternative use, but not by use for the biofuel crop; it would therefore be correct to include the emissions in the calculations for the biofuel crop. It is not clear that substantial N<sub>2</sub>O emissions would actually be avoided by anaerobic digestion, as the nitrogen is still present at the end of the process in the digestate, and this will be returned to the land where N<sub>2</sub>O emissions would still occur (Amon et al 2004a,b; Clemens, 2004).

Emissions from manures/compost/sewage sludge etc could be calculated reasonably simply using the IPCC approach. The emissions from these sources would be calculated from *total* N applied. However, only a proportion of the N from these sources is actually available to the crop to which it is applied, typically 5-25% (MAFF 2000 – RB209). Therefore, if emissions from manures are fully accounted for using the IPCC approach, growers using manures to replace some of their N fertiliser input would show considerably higher calculated N<sub>2</sub>O emissions than growers not using manures. This could potentially lead to the unintended consequence of farmers deciding *not* to use organic manures on crops for biofuels. Given that the manure/compost/sludge will still be applied somewhere, and that such additions can be beneficial to the soil, this change in behaviour would not make sense in overall GHG emissions terms and would not promote the efficient use of N fertilisers.

A workable compromise to the situation might be to include N<sub>2</sub>O emissions from manures, composts etc, but to calculate this on the basis of *readily available* nitrogen. A similar approach was taken by Thorman et al., (2006) when new research data on measured N<sub>2</sub>O emissions following the land spreading of a range of farm manure types was collated and incorporated into an enhanced and updated version of the MANNER decision support system (MANNER-NPK). MANNER was developed to enable farmers to estimate the manure contribution to crop available nitrogen supply and to estimate environmental losses (Chambers et al., 1999). Loss of N<sub>2</sub>O was expressed as a percentage of the manure *readily available* nitrogen content *remaining after* nitrogen loss via *ammonia volatilisation*. The mean loss of N<sub>2</sub>O was 1.96% of *readily available* nitrogen *remaining after ammonia volatilisation* and no difference was observed in emission factors between manure types (Thorman et al.,

2006). Such an approach could help avoid both scenarios of reduced fertiliser use due to manure application being unfairly advantageous in GHG terms and the use of manure being discouraged by full accounting of its N<sub>2</sub>O costs.

#### 4.2.4 N<sub>2</sub>O from Crop Residues

The above calculations only deal with N<sub>2</sub>O emissions related to applying N fertiliser. The IPCC calculations also account for both direct and indirect emissions from N in crop residues and N that results from mineralisation of SOM. Crop management and N fertiliser can alter the amount of N in crop residues in the soil and soil organic matter (e.g. crop yield, straw removal/incorporation) and can also potentially affect the amount of mineralisation of soil organic matter. Changes in SOM caused by cropping are discussed in the next section.

In terms of N<sub>2</sub>O emissions resulting from crop residues, using the IPCC methodology and defaults for an 8t/ha crop of wheat gives total N<sub>2</sub>O emissions of 2.07 kg/ha (1.69kg direct emissions, 0.38kg indirect emissions via subsequent leaching) see Table 12. Using the same approach but values that might be considered more appropriate for UK wheat crops gives a slightly reduced figure of 1.70 kg N<sub>2</sub>O/ha. Either way, a considerable 'baseline' emission in addition to the emissions associated with N fertiliser is derived, contributing a further 500-600 kg CO<sub>2</sub> eq/ha.

**Table 12: Calculation of N<sub>2</sub>O emissions from crop residues**

	Wheat (using IPCC defaults)	Wheat ADAS estimate	OSR (IPCC using grain default)	OSR (IPCC using IGER figures)	OSR ADAS Estimate
DM Crop yield <i>t DM/ha</i>	7.12	6.80	2.75	2.75	2.75
Harvest Index	0.387	0.500	0.408	0.455	0.350
Above ground residues <i>t DM/ha</i>	11.27	6.80	3.98	3.29	5.10
N% of above ground residues	0.6%	0.9%	0.6%	1.5%	1.2%
N in above ground residues <i>kg N/ha</i>	67.6	61.2	23.9	49.4	61.2
ratio of below ground residues to above ground biomass	0.24	0.22	0.22	0.22	0.22
Below ground biomass <i>t DM/ha</i>	4.41	2.99	1.48	1.33	1.73
N% below ground residues	0.9%	0.9%	0.9%	0.9%	0.9%
N in below ground residues <i>kg N/ha</i>	39.7	26.9	13.3	12.0	15.5
Total N in crop residues <i>kg N/ha</i>	107.4	88.1	37.2	61.4	76.7
Direct N <sub>2</sub> O emissions <i>kg N<sub>2</sub>O/ha</i>	1.69	1.38	0.58	0.96	1.21
Indirect N <sub>2</sub> O emissions <i>kg N<sub>2</sub>O/ha</i>	0.38	0.31	0.13	0.22	0.27
Total N <sub>2</sub> O emissions <i>kg N<sub>2</sub>O/ha</i>	2.07	1.70	0.72	1.18	1.48



In reality the amount and N concentration of crop residues will be heavily influenced by N fertiliser applications (Powlson et al. 1992), but there is not a straightforward consistent relationship that could be used. In terms of certification, if N<sub>2</sub>O from crop residues is to be accounted for, then it is likely that standard default factors for each crop will be applied.

With organic inputs such as slowly decomposable manure, and especially crop residues such as straw, only a small fraction of the N will be converted to nitrate within the first growing season after application. However, the IPCC approach works on the basis that the N contained in these inputs will be at risk of denitrification, even if not until several years in this future. Thus, verifying the default emission factors used will be difficult as measurements would need to continue over several years.

Factors other than the N content of straw are crucial for influencing likely N<sub>2</sub>O emissions. For example, straw can alter soil physical conditions and this will greatly affect the risk of N<sub>2</sub>O emissions from denitrification. In general it would be expected that straw incorporation would improve physical conditions, with a decreased risk of anaerobic zones forming and lower N<sub>2</sub>O emissions a likely long-term impact.

Using the IPCC approach for calculating N<sub>2</sub>O emissions from crop residues for an 8 t/ha wheat crop gives about 100kg N returned to the soil which is treated in the same way as N fertiliser for direct emissions, giving 1.7kg N<sub>2</sub>O/ha, and a further 0.4kg N<sub>2</sub>O/ha from indirect emissions via leaching. The inclusion of the calculation of N<sub>2</sub>O emissions from crop residues gives a baseline figure that may overestimate 'real' emissions of N<sub>2</sub>O from unfertilised arable soils (e.g. Defra 2001; Project CC0224).

However, it is possible to account for straw removal. If crop residues are removed then, assuming 2/3 removal of above ground dry matter (R Sylvester-Bradley pers. comm.), using the IPCC calculations would reduce total N<sub>2</sub>O emissions to 1.2 kg N<sub>2</sub>O/ha, giving a GHG saving of around 250kg CO<sub>2</sub> eq/ha.

In reality, evidence that removing crop residues reduces subsequent N<sub>2</sub>O emissions in UK arable conditions by this extent is more equivocal; the N<sub>2</sub>O emissions from incorporating crop residues can depend on the carbon:nitrogen ratio (N<sub>2</sub>O emissions decreasing with increasing C:N ratio) of the residue, timing of incorporation (whether it is wet & warm or cold & dry), type of incorporation, soil type and N uptake from the following crop (Defra 2000 Project CC0235; Defra 2001 Project CC0224). Straw removal is not an appropriate mitigation tool for reducing N<sub>2</sub>O emissions, as it has the benefit of maintaining soil organic matter content, as discussed in more detail in section 4.3.

There is also some question as to whether it is logical or fair to consider N cycling through crops and their residues to give N<sub>2</sub>O emissions but not to consider the N cycling continuously occurring in soils. There is a wider question of whether the potential for N<sub>2</sub>O emissions from a short period of high nitrate concentrations (as is the case after fertiliser application) is equivalent to that from a long period of low nitrate concentrations (e.g. after incorporating residues).

In the long term, N<sub>2</sub>O emissions will result from incorporating crop residues. However, incorporation of straw back into soils is considered good practice for maintaining soil organic matter levels. Therefore it may currently be appropriate to include a small default value in an accreditation system for emissions from crop residues, whether or not straw is removed.

#### ***4.2.5 N<sub>2</sub>O from soil organic matter changes and organic soils***

The mineralisation of soil organic matter can result in nitrate becoming available in the soil and thus being at risk of denitrification. In the UK, SOM in long-term arable land remains at a relatively stable equilibrium (Webb et al. 2003) so N<sub>2</sub>O emissions from mineralisation of SOM are likely to be small on the majority of UK arable soils. However, emissions might be

significant where non-cultivated soil with high SOM, such as grassland, is brought into cultivation for the purpose of biofuel cropping. This may be regarded as a consequence of land-use change rather than biofuel cropping per se. Changes in SOM will be discussed in more detail in section 4.3.

IPCC accounts for emissions from soils with a very high organic matter separately. Organic soils (or histosols) are defined, as soils where the topsoil contains at least 20% organic matter (see IPCC 2006 for full definition). Care is needed here as alternative soil definitions (such as RB209; MAFF 2000) define soils with between 6% and 20% organic matter as being 'organic', and those with more than 20% organic matter as being 'peaty'. Organic (or peaty) soils would include much of the arable/vegetable producing land in the Fens of Cambridgeshire, Lincolnshire and Norfolk totalling 39,000ha (IGER GHG inventory).

The IPCC emission factor used for temperate organic crop and grassland soils is 8kg N<sub>2</sub>O-N/ha (uncertainty range 2-24kg/ha), or 12.57kg N<sub>2</sub>O/ha equating to 3720kg CO<sub>2</sub> eq/ha. Given the size of the emissions from organic soils it may seem inappropriate to use organic soils for biofuel cropping.

There is also a broader question of apportionment to consider. The fact that organic soils are emitting N<sub>2</sub>O (and also CO<sub>2</sub>) is generally the result of these fields being drained and converted to cropping many years ago. It might therefore be considered that these emissions should not be associated with the growing of a biofuel crop, as they will be cropped anyway and emissions will occur whether or not a biofuel crop is grown. The counter argument however would suggest that given sufficient incentive, these soils could be managed in a way so that GHG emissions are avoided (e.g. reversion to wetland, forest etc) and so the full GHG costs of continued cropping should be accounted for. Also, the high level of soil mineral nitrogen in organic soils means that fertiliser requirements are very low. If N<sub>2</sub>O emissions from organic soils are not accounted for, the low fertiliser inputs and reasonably high yields could translate into very favourable total GHG emissions/ha, potentially inappropriately incentivising the use of these soils for biofuel cropping.

The IPCC approach only specifically accounts for soils with greater than 20% organic matter. However, significant emissions may still be expected from soils with between 6 and 20% OM content. In theory these could be accounted for by the yearly change in organic matter, but this would not be easy in practice. It is possible that it would be appropriate to include a default figure for these soils as well as for organic soils. There is little experimental data on N<sub>2</sub>O emissions from organic soils in the UK to test the suitability of the IPCC default values for UK conditions. Experimental validation of these emissions might be required before implementing such high values in an accreditation scheme.

An estimate of N<sub>2</sub>O emissions from soils classed as organic (>20% SOM) should be included in the GHG calculator and certification schemes, and in future an estimate of emissions from soils with 6-20% SOM should also be included.

#### **4.2.6 Appropriate baseline N<sub>2</sub>O emissions**

Another uncertain issue is that of baseline emissions; there would still be some N<sub>2</sub>O emissions even on set-aside land if no fertiliser was added. This would depend on the amount of mineral N and N in soil organic matter and crop residues remaining in the soil. However, due to a lack of clear data and uncertainty over the appropriateness of this figure the LowCVP (2004) study and the current GHG calculator do not include a reference value for set-aside land. Mortimer & El Sayed (2006) did use a figure of 0.95 kg N<sub>2</sub>O/ha for mown fallow rotational set-aside. Brown et al. (2002) calculate that, overall for the UK, 'background' emissions amount to 43% of total emissions.

The issue of appropriate background N<sub>2</sub>O emissions to assume is not straight-forward as the IPCC methodologies aim to deal with anthropogenic N<sub>2</sub>O only.

Research experiments commonly use unfertilised plots to account for background emissions. However, these background emissions cannot be considered completely unaffected by agriculture as they are influenced by past land-use and agricultural practice of the site (Brown et al., 2002). More generally, difficulties can also arise as experimental results often don't cover a full year, and given the large temporal variability (both day to day and season to season) in N<sub>2</sub>O emissions it can be difficult to extrapolate to annual figures. There is a need to separate out the specific emissions from biofuels cropping from emissions that would be incurred anyway. Essentially, the question is, would N<sub>2</sub>O emissions from cropped land without fertiliser be different from uncropped rotational set-aside, or indeed from ungrazed, unfertilised permanent grass? Dobbie & Smith (2003a) suggest that background N<sub>2</sub>O emissions from arable crops (i.e. without fertiliser) tend to be higher than from grassland with emissions of 0.3kg N<sub>2</sub>O-N/ha for unfertilised spring barley compared to 0.1kg N<sub>2</sub>O-N for unfertilised grass (Clayton et al 1997; McTaggart et al 1997). However, data on background N<sub>2</sub>O emissions is limited and conclusions drawn from experiments on Scottish sites do not always translate to the UK as a whole. Recent work across the UK has shown background emissions in arable crops to be very variable across sites, ranging between 0.23 and 1.52 kg N<sub>2</sub>O-N/ha (Defra 2006; Project NT26).

There is a need to more rigorously assess how large these background emissions are, and how much of it can be apportioned to biofuel cropping. Until there is better understanding of this it may be best to only include emissions directly related to N fertiliser (and manure) inputs in accreditation calculations at this stage.

In terms of background emissions not associated with fertiliser it is possible that calculated emissions from crop residues using the IPCC approach could be used as a surrogate figure.

#### ***4.2.7 Other potentially important factors affecting N<sub>2</sub>O emissions***

A range of additional factors to those outlined above may also have an influence on the rate and scale of N<sub>2</sub>O emissions from the feedstock production for biofuels and other products. These are discussed below.

##### ***4.2.7.1 Weather and soil type***

Denitrification is favoured by wet and warm conditions. Soils that hold more water, such as clays, may remain wetter for longer so generally denitrify more. However, the systems responsible for N<sub>2</sub>O emissions are very complex and so such statements may not always hold true. Differences between soil types in N<sub>2</sub>O emissions are often not consistent as N<sub>2</sub>O emissions are dependent on a full range of soil parameters, and it hasn't proved possible to clearly classify soils for their likely N<sub>2</sub>O emissions (Defra 2003 – CC0248).

The parameter which has been found to be most useful in explaining N<sub>2</sub>O emissions is the percentage of water-filled pore spaces (WFPS) (Dobbie et al 1999). This is a function of both soil moisture (i.e. rainfall) and soil properties.

When soil conditions are very dry or very cold then N<sub>2</sub>O emissions are usually very low, irrespective of form or quantity of N applied. Conversely, when soils are much wetter (WFPS>70-80%) and warmer, emissions can be very high, if nitrate is readily available in the soil (Defra 2006 (NT26)). The period immediately after inorganic N fertiliser application is especially vulnerable due to the high nitrate concentration at the time, before very much N has been taken up by the crop. The weather in the three weeks after N application for OSR and wheat has been shown to be particularly important in determining N<sub>2</sub>O emissions from denitrification (e.g. Addiscott and Powlson, 1992; Macdonald et al., 1997).

Whilst there may well be accountable differences in N<sub>2</sub>O emissions from the extremes of soil type (e.g. clay soils in the wetter West vs sandy soils in the drier East), it is less obvious that

it would be possible to distinguish fairly between more medium soil types. Organic soils should be treated as a different class where the potential for N<sub>2</sub>O emissions is substantially greater.

In reality, the quantity of N<sub>2</sub>O evolved from soil in any situation will be very dependent on weather conditions (especially rainfall and temperature). The same system of crop management, including fertiliser or manure applications, could give low emissions in one year and high emissions in another – the actual emission depending on weather. Whilst calculating emissions for a range of cropping scenarios each year to reflect differing weather conditions, might be possible using a model such as DNDC, this is unlikely to be practical for accreditation purposes: it would be resource intensive, would give unmanageable uncertainty to the carbon intensity figures year on year, and would be very difficult to implement given that accreditation and data collection on-farm would likely occur after the crop growing season and therefore after much of the crop would already have been bought and used by the biofuel processor and hence data for carbon reporting already required.

Any simple scheme must therefore reflect the main factors increasing the *risk* of N<sub>2</sub>O emissions, averaged over weather conditions. How such averaging should occur is also problematic; it is likely that (other factors staying the same), emissions will be fairly low in many years but with some periods of very high rates for fairly short periods in some years, for example when a period of warm temperatures and high rainfall follows fertiliser application or a period of high soil nitrate content following mineralization of soil organic N or added manure.

#### **4.2.7.2 Fertiliser type and timing**

The form of N fertiliser can influence subsequent N<sub>2</sub>O emissions, being influenced also by soil and weather conditions (Harrison & Webb, 2001). In wet soils, N<sub>2</sub>O emissions can be substantially greater from nitrate than from ammonium or urea, as denitrification predominates. However, ammonia emissions from urea can be significantly higher than from ammonium nitrate, giving the potential for much increased indirect N<sub>2</sub>O emissions. Overall, recent Defra funded work has concluded that there is no convincing difference in total N<sub>2</sub>O emissions between ammonium nitrate and urea, the two most common fertilisers used in the UK (Defra 2006 (NT2603)). However, the work did show some benefit in using nitrification inhibitors or slow-release fertilisers to help mitigate emissions.

Rainfall immediately before and after N fertiliser addition has been shown to be a key determinant factor in resulting denitrification and N<sub>2</sub>O emissions (Dobbie and Smith 2003b). Seeing as the main applications for N fertiliser in OSR and wheat are in March and April the weather in the months of March, April and May is of key importance. There could potentially be some benefit in altering N inputs in order to avoid high emissions, by making more of the application in the cooler months February/March than in April/May. This may well also be an appropriate strategy for fertilising wheat crops to get low grain protein content for good alcohol yields. However, wetter soil conditions in February/March often preclude applying much fertiliser at this time as it is not possible to get on to the land, and there would be no guarantees that reduced N<sub>2</sub>O emissions would result.

The practice of applying autumn-N to rape could lead to potentially high N<sub>2</sub>O emissions, if temperatures are warm. Seeing as yield benefits from autumn application are inconsistent, this could potentially be worth incorporating into the calculations in accreditation to ensure that the full effects are accounted for, and these applications may be limited if appropriate.

The timing of manure applications is of importance with regard to nitrogen loss and N<sub>2</sub>O emissions. Autumn/winter applications can be associated with greater N losses. Recent research has shown that following livestock slurry application on free draining, grassland soils both direct and indirect N<sub>2</sub>O losses were greater from autumn/winter slurry application timings than from spring timings (Thorman et al., 2007). Manure applications at a time when

the crop is actively growing and taking up nitrogen i.e. spring will incur less N loss than if there is no actively growing crop. Current Nitrate Vulnerable Zone (NVZ) rules preclude manure applications at certain times of year. Recent work has shown a degree of 'pollution swapping' between ammonia, leaching and N<sub>2</sub>O for different manure management strategies (Williams et al 2006b).

Further work is required before any certification scheme could incorporate information on N<sub>2</sub>O emissions related to fertiliser type or timing of application.

#### **4.2.7.3 Crop type**

Different crops tend to receive differing quantities of N fertiliser, so N<sub>2</sub>O emissions per ha will vary between crops. However, there is little evidence that wheat and OSR would differ much in N<sub>2</sub>O emissions per unit of N applied (although it is possible that greater N<sub>2</sub>O emissions could be associated with autumn timing of N fertiliser to rape). There is some evidence that sugar beet may give greater N<sub>2</sub>O emissions per unit of N applied than combinable crops (Flynn et al. 2006). In general however, there is insufficient data to assign different emission factors to different crops.

Under the 1996 IPCC guidelines nitrogen fixation by legume crops was assumed to result in N<sub>2</sub>O emissions to the same extent as N additions to the crop. However, this has changed under the revised 2006 guidelines, whereby no emissions are assumed to come from biological fixation. N<sub>2</sub>O emissions only come from incorporating the crop residues. No leguminous biofuel crops are grown in the UK currently, but this does affect how the carbon intensity of biodiesel from oilseed rape compares with that from soy: as soy is a legume requiring no fertiliser-N inputs, and responsible for no N<sub>2</sub>O emissions (if crop residues are not accounted for) soy has a very favourable carbon intensity (1182kg CO<sub>2</sub>/t biodiesel compared to 2286kg CO<sub>2</sub>/t biodiesel for OSR; Department for Transport, 2007).

#### **4.2.7.4 Cultivations and residue incorporation**

Generally, the effects of different cultivation strategies on N<sub>2</sub>O emissions are small and can interact substantially with soil type, weather conditions, timing and the residues or manure that are incorporated. However, zero-tillage (also known as direct drilling or no-till) has been associated with very large N<sub>2</sub>O emissions following N applications (Aulakh et al. 1984; Ball 1999, Thorman 2002), as the frequency of anaerobic conditions can be increased. As zero tillage can be used to increase soil organic matter it has been advocated as a possible climate change abatement strategy through carbon sequestration. Clearly, if the potential sequestration benefits of zero-tillage were to be quantified in an accreditation scheme it would also be necessary to account for potential negative consequences of increased N<sub>2</sub>O emissions. Whilst values have been used to model the effect of zero-tillage on N<sub>2</sub>O emissions (Mackenzie et al. 1998; King et al. 2004) there is considerable uncertainty over this so further work would be required to establish an appropriate emission factor for this purpose.

In practice zero-tillage is currently used on only a small proportion of UK arable land (~7%; Anon, 2005), largely due to problems with slugs, weed control and compaction. However, *minimum*-tillage (also known as non-inversion tillage) has increased in popularity over recent years and now accounts for the main cultivation strategy on a high proportion of arable land (~43%), as opposed to the traditional strategy of ploughing followed by cultivations (~50%). Whilst differences in N<sub>2</sub>O emissions between ploughing and minimum tillage have been noted in research, at this stage consistent trends are not obvious; ploughing gives highest N<sub>2</sub>O emissions in some circumstances whilst minimum tillage gives high emissions in others (Defra 2001 (CC0224)). A recent report (Defra 2007 (SP0561)) has cautioned that 'increased N<sub>2</sub>O emissions may completely offset the amount of carbon

potentially sequestered through changing from conventional to reduced tillage practices', but that 'the evidence is not clear'.

The N<sub>2</sub>O emissions that result from crop residues can depend on the overall N content and C:N ratio of the residue; residues with low C:N ratios (such as sugar beet) tend to give higher N<sub>2</sub>O emissions than those with higher C:N ratios (such as cereal straw) (Defra 2001 (CC0224)). The influence of the C:N ratio is not accounted for in the IPCC approach to calculations of N<sub>2</sub>O emissions from crop residues.

Further experimental evidence is required on the effects of cultivation on N<sub>2</sub>O emissions to increase the understanding in this area, and before effects could be quantified in accreditation schemes.

#### ***4.2.8 Modelling N<sub>2</sub>O emissions on a geographic basis***

It has been suggested that climate and soil factors could be used on a geographical basis to give N<sub>2</sub>O emissions using DNDC model outputs (Bauen et al., 2006). This could be a useful approach for selecting different emission factors for application to N input data, however, it is unlikely to give fair results if individual values were assigned simply to different regions. Potentially, it could be possible to use actual weather data from each individual year in the simulation of N<sub>2</sub>O emissions. There may however be a question of fairness in doing this, with farmers potentially being penalised for weather conditions that they obviously have no control over. There could be possible issues of the nature of a post-code lottery if soil type and climate data are taken from look-up tables on a post-code basis, potentially with disputes for crops grown on or near boundaries. The nutrient management software 'Planet' does use a system whereby farmers can access rainfall data by inputting their telephone dialling code, or can use their own records if available ([www.planet4farmers.co.uk](http://www.planet4farmers.co.uk)).

Importantly, there is a need to recognise that the ultimate source of N<sub>2</sub>O emissions is from N inputs (wherever these are from). Ultimately, therefore any system should encourage the efficient use of N and discourage the over-use of N. This cannot however be arbitrarily assigned to a given appropriate rate of N fertiliser; in a situation with low soil SMN, low N from crop residues, no N from animal manures etc and good yield potential 250 kg/ha fertiliser N may be an appropriate rate. On the other hand, an application of 150 kg/ha N could be excessive where the soil has a high SMN and considerable N is available from crop residues and previous manure applications and could potentially be associated with higher N<sub>2</sub>O emissions.

Any basis for producing geographically based emissions could face potentially large interactions with weather and soil conditions, particularly regarding rainfall, temperature and water-filled pore space around the time of N application. At present it probably isn't possible to predict emissions in such a way with sufficient accuracy or consistency for use in reporting purposes. As this level of detail isn't yet used in reporting for IPCC purposes then it is probably not necessary for accreditation for biofuels at this stage.

The absolute extent of N<sub>2</sub>O emissions from biofuel is of great consequence to the biofuel industry, as it has a large bearing on the overall GHG savings calculated for biofuels, and hence will effect the public perception of biofuels and their economic value as the RTFO becomes based on a 'carbon saving' basis. The differences in calculated N<sub>2</sub>O emissions are therefore of utmost importance to the viability of a UK biofuels industry. Using the IPCC approach to calculate N<sub>2</sub>O emissions, and accounting for emissions from inorganic and organic fertilisers, crop residues and organic soils would give emissions significantly in excess of 6.2kg/ha N<sub>2</sub>O at normal N rates of 200kg/ha N. This is considerably higher than the estimates generated using DNDC of between 2.2 and 4.36kg/ha N<sub>2</sub>O (LowCVP, 2004; Woods et al., 2005; Edwards et al., 2006). However, recent work by Crutzen et al. (2007) examining the increase in atmospheric N<sub>2</sub>O concentrations has suggested that emissions

from agricultural land are much greater than calculated even by the IPCC approach. There is clearly a need to better understand and quantify the magnitude N<sub>2</sub>O emissions resulting from cropping, not just for quantifying the carbon intensity of biofuels but to inform climate science and climate change mitigation strategies on land-based production more generally.

#### **4.2.9 Conclusion on N<sub>2</sub>O emissions**

The methodology for calculating N<sub>2</sub>O emissions needs to be transparent and defensible. If conducted on a farm by farm basis, it should help to influence the factors by which N<sub>2</sub>O emissions are truly affected and over which growers have some control. In terms of keeping emissions calculations simple and practical there is a case for just using a simple linear relationship between N fertiliser and N<sub>2</sub>O emissions ignoring other N sources. Indeed, this is the approach proposed for the RTFO Carbon Reporting (Department for Transport, 2007).

However, to avoid the calculations being misrepresentative, it will also be necessary to account for emissions from organic soils, from crop residues (perhaps as a baseline emission) and from the use of manures/compost/sewage sludge.

It may be simple to account for emissions from organic soils by using the IPCC default of 12.6 kg N<sub>2</sub>O/ha, though this should be validated for UK conditions. It would also be possible to account for N<sub>2</sub>O emissions from applied manures etc by using the IPCC approach, however, this may give unintended implications for growers. There is a need to resolve how N<sub>2</sub>O emissions from manures etc should best be dealt with, giving full consideration to potential impacts on farm practice. A compromise solution of using N<sub>2</sub>O emissions calculated on *available* nitrogen may be the most appropriate and practical approach. The approach to use for N<sub>2</sub>O emissions from crop residues requires further consideration. Accounting for crop residues gives a convenient baseline emission with no fertiliser. However, consideration needs to be given to dealing with straw removal, where it may be best to use the same number whether or not straw is removed.

Given the complexity of the system leading to N<sub>2</sub>O emissions and the very large interactions with weather and soil type, it may not be possible to reliably estimate *actual* emissions in a given year. Despite this, the main factors increasing *risk* of N<sub>2</sub>O emissions must be accounted for, averaged over long-term weather conditions.

There is a problem in accounting for the N available to the crop from the soil, and how this influences appropriate fertiliser rates. There is a need to better understand the relationship between N fertiliser (or really N supply) and N<sub>2</sub>O emissions. If it is the case that super-optimal applications give proportionately larger emissions than applications at or below the optimum, it would be right for this to be reflected in the calculations. Further work is required to consider how such an approach could be developed.

In including emissions from different sources in the calculation of N<sub>2</sub>O, regard must be paid to the reference system being used, and it will be important that emissions in any reference system are fully accounted for. As the growing of crops for bioenergy is unlikely to differ markedly than growing crops conventionally (other than potentially lower N fertiliser use, see section 3.2), it should be recognised that switching to biofuels crops will not substantially increase the N<sub>2</sub>O emissions from the UK; these same emissions are currently occurring from conventional arable crops.

There is uncertainty around all of the emission factors used in the IPCC approach to calculating N<sub>2</sub>O emissions, and in many instances specific emission factors for the UK (or for regions, or management practices) may be more appropriate than the default emission factors. Work is needed to validate and improve the range of emission factors used for UK conditions. This work is occurring and it is inevitable that any emissions factors established now will change with improved understanding and base levels of data. Thus accounting and

certification systems must be flexible enough to allow the incorporation of such new knowledge as it emerges.

There is a need for future work to define 'realistic' N<sub>2</sub>O emissions from arable land (with and without fertiliser) across UK regions and soil types, including reference systems. As well as drawing on past and present experimental work, UK-DNDC may be a useful tool in this regard.

In future other influencing factors such as climate, soil factors and cultivations could be included if robust relationships can be proven, perhaps by using refined disaggregated emission factors for calculation of the different parts of the N<sub>2</sub>O emissions, as suggested in the IPCC guidelines 2006 Tier 2 approach. Disaggregated emission factors could be potentially envisaged for different soil types, climate, fertiliser type etc (Bouwman 2002; Stehfest & Bouwman 2006). The values for such emission factors may come from experimentation, or also potentially from testing scenarios using models such as DNDC.

When using the final calculated figures for the GHGs arising from biofuel production that include N<sub>2</sub>O emissions from land, it is important to recognize that these are not 'new' emissions that are the result of growing the crop for biofuel; they would very likely have occurred anyway given that the crop would likely still have been grown, but for food use. This soon leads to the inescapable conclusion that it makes little sense accounting for GHGs from biofuel crops without accounting for GHGs from all land uses and products from the land. It is also important that comparisons of land use or biofuels from different parts of the world use a common approach to calculating N<sub>2</sub>O and land-use change emissions.

### **4.3 Effects of cropping on soil organic matter**

The soil can act as either a sink or a source for CO<sub>2</sub>, dependent on various soil factors and agricultural practices. Soils may deplete in soil organic matter (SOM, or soil organic carbon (SOC)) so lose CO<sub>2</sub> (and N with some N<sub>2</sub>O loss) to the atmosphere, thus adding to global warming. Arable soils in the UK have generally declined in SOC since permanent grasslands were ploughed up post World War II, but the rate of decline seems to have reduced to be close to equilibrium in SOC (Webb et al. 2003; King et al. 2005). With some suitable agricultural practices it can be possible to maintain or increase SOM over a number of years (Lal 2004; King et al. 2004; Smith et al. 2007), thus potentially sequestering CO<sub>2</sub>.

The simple act of growing higher yielding crops (through better varieties or nutrition) tends to increase soil organic matter to some extent through the return of greater quantities of crop residues (Smith et al. 2007). Cover crops may be considered another potential way of adding carbon to the soil, as well as potentially reducing nitrogen losses. Adding organic manures, sewage sludge or compost to the soil can also increase SOM.

Estimates of the potential carbon sequestration from management practices in the UK calculated in two studies (Smith et al. 2000a,b; King et al. 2004) are given in Table 13. There is a considerable range in the potential savings from some of these management practices, both within and between the studies.

It would however seem from this analysis that incorporating cereal straw or adopting a zero-tillage cultivation strategy could give substantial CO<sub>2</sub> savings. The biggest contention may be over the value of zero tillage. Current Defra research is investigating whether carbon sequestration by changed land management practice is both scientifically plausible and measurable; "A critical review of the extent to which reduced tillage practices and organic matter returns will increase the carbon content of arable soils under English and Welsh conditions. (Project Defra 2007 (SP0561))". This is likely to report that the average effect of zero-till from UK experiments gives a total saving of 310kg Carbon/ha.yr (SE 180kg C/ha.yr), equating to 1240 kg CO<sub>2</sub> (A. Bhogal, pers comm.) Recent UK experimental data for tillage



effects on SOC, especially regarding minimum tillage, is lacking. There is currently very little certainty in estimates of sequestration from minimum tillage in UK soils, but the best estimate currently is half that for zero tillage (A Bhogal, pers comm.)

**Table 13: Potential for CO<sub>2</sub> mitigation from different practices from data from King et al (2004) and derived from Smith et al (2000a,b) using potential accumulation rates given and assumed SOC content of 81t C/ha. Values are expressed as kg CO<sub>2</sub> saved through S**

Management practice	Smith et al. 2000a,b	King et al. 2004
Animal manure	454-2109	200-832
Sewage sludge	1588	2440
Cereal straw	1361-3891	2128-2868
No-till	2365	580-940
Min-till		160

However, changes to SOC are finite as new equilibria will be reached after a number of years. There is little agreement as to when significant changes to SOC from a change in management will stop, but King et al. (2004) suggest a period of 10 years as being prudent for calculating mitigation potential. Furthermore, any change in management strategy that reverts back to conventional arable would subsequently cause a fairly rapid decline in soil organic matter, and release of 'stored' CO<sub>2</sub> back to the atmosphere. This is especially important with regard to minimum tillage as ploughing every 3 to 4 years is normal practice to alleviate weed, disease and compaction problems, and ploughing is likely to result in much of the previously stored carbon being released. It is therefore not appropriate to consider this sequestration as being annually cumulative (A Bhogal, pers comm.).

It is very questionable whether accounting for effects on SOM of incorporating animal manures, compost or sewage sludge within the calculations for biofuel crops would be appropriate. The amount of each of these sources that can be applied each year is determined by how much is produced, not by agricultural demand (Powlson et al 2008), and so might be considered outside the biofuel crop production system. Their use may be beneficial to the soil both in terms of providing nutrients and increased SOM, and it is important that their efficient use is encouraged. There is a danger that including their effects on SOM in accreditation schemes could distort practices.

Carbon returned to the soil in farmyard manure (FYM) is largely in the form of straw that will have previously come off the land elsewhere. The issues of incorporating straw and FYM therefore need to be considered together.

In terms of including calculations of SOM in accreditation schemes there are a number of difficult issues, not least because CO<sub>2</sub> sequestration/emissions will be affected by past and future management. It would be difficult to measure or verify the calculated changes in SOM and consequent CO<sub>2</sub> savings/emissions for an individual farm. As the practices to increase SOM are required on a long-term basis, are finite and are reversible on reversion back to conventional practice, it would be difficult to allocate savings to an individual biofuel crop year, and to ensure that savings were not lost in subsequent years.

Consideration of SOM in the UK may be less important than in other parts of the world (due to having mature soils/ temperate climate) or if comparisons are made against perennial crops, which can accumulate more SOM (Powlson et al. 2005). There may therefore be a case for including calculations to allow fair comparisons with other global crop systems.

If SOM is to be used in GHG calculations then a practical approach for accounting for changes and dealing with the issues will need to be developed. To ensure that benefits are not over-represented, accounting for CO<sub>2</sub> sequestration from straw incorporation or zero-tillage would require that any negative consequences on N<sub>2</sub>O emissions are also appropriately accounted for. A pragmatic first solution may be to count a CO<sub>2</sub> credit for all straw that is not burned, assuming that if the straw isn't burned it will find its way back to the land at some point through FYM.

Whilst it may not make sense to directly quantify management effects on SOM for individual biofuel crops this may not mean that it should be abandoned from the calculations altogether. For example if minimum tillage is maintained on all crops throughout a rotation then there may be positive effects on SOM that should be encouraged. This would require however a 'whole-farm' approach to accreditation.

The above discussion relates to SOM changes within the biofuel growing system; these are minor compared to changes in SOM that result from land use change due either directly or indirectly from biofuels (e.g. Searchinger et al, 2008) see Section 1.1.2 and below.

#### **4.3.1 CO<sub>2</sub> released from organic soils**

SOM changes due to land use change, i.e. converting from grassland to arable, can give very large emissions of CO<sub>2</sub> over a period of years (>3500 kg CO<sub>2</sub>/ha.yr). Even with very efficient biofuel production it could take decades before sufficient GHG savings are made from the use of biofuels to give a truly net positive GHG saving (Edwards et al. 2006).

The long-term nature of these effects makes them very difficult to deal with in a quantitative way in accreditation systems. It is also difficult to attribute such land-use change directly to cropping for biofuels; if grassland is converted to arable it is likely that the crops for food as well as for fuels will be grown in the rotation. Further problems arise if land-use change results indirectly from cropping for biofuels, i.e. when biofuel crops displace food crops so more land is required to produce food. In such a case it doesn't really matter whether biofuel crops or food crops are grown on the previously uncropped land, the change is, partially, due to the introduction of biofuels and the consequent GHG emissions remain the same. Because of these difficulties, the CO<sub>2</sub> consequences of land use change have generally been excluded from assurance and certification systems. It has been suggested that such issues of land use change are more appropriately dealt with through government legislation. Recent analysis by Searchinger et al. (2008) suggests the scale of these indirect effects may be too large to ignore. This is currently being assessed for the UK in the 'Gallagher Review' on indirect effects on biofuels.

#### **4.4 Lime acidification and CO<sub>2</sub> release**

Lime is applied to land to compensate for lime loss and to neutralise soil acidification. IPCC coefficients for emissions of CO<sub>2</sub> from lime are 0.12 and 0.13 tonnes C per tonne crushed limestone and dolomite<sup>2</sup> applied. These are the mass fractions of C in these rocks. However, this approach clearly misrepresents losses of CO<sub>2</sub> from the fossil C in arable soils. There are three main issues:

- A large proportion of UK arable land is naturally calcareous, being developed from underlying limestones or chalks e.g. the Wolds, the Downs, the Chilterns and East Anglian boulder clays. These soils are never limed, but nevertheless lose C as CO<sub>2</sub> due

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<sup>2</sup> See [http://www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/4\\_Volume4/V4\\_11\\_Ch11\\_N2O&CO2.pdf](http://www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/4_Volume4/V4_11_Ch11_N2O&CO2.pdf)

to acidifying processes. Such emissions of fossil C are overlooked in current IPCC calculations.

- A proportion of the lime applied to non-calcareous soils is to replace leaching of carbonate. Thus West & McBride (2005) estimate that only about 50% of the C in crushed limestone and dolomite is released as CO<sub>2</sub> to the atmosphere, the rest being precipitated at sea.
- The principal soil processes causing soil acidification are associated with fertiliser application (especially nitrate, sulphate and chloride containing products), atmospheric deposition of sulphur dioxide and ammonia, mineralisation of soil organic matter, nutrient transformations in soil (e.g. conversion of ammonium to nitrate by soil organisms) and nutrient uptake by roots (Archer 1985). It is estimation of these processes that will give the best assessment of CO<sub>2</sub> emissions from fossil soil C.

The main body of information in the literature concerns lime losses from soils and lime requirements to maintain a particular pH. There do not appear to have been studies on CO<sub>2</sub> emissions from fossil soil C. It is therefore difficult to make quantitative estimates or specify the factors causing most variation. For example, it is not clear whether nitrogen fixed biologically has a net acidifying effect equivalent to the same amount of N applied as fertiliser. Suffice it to say that the stoichiometric quantities of CO<sub>2</sub> released through neutralisation of a normal application of 200 kg/ha fertiliser N (as ammonium nitrate or urea) would be 200-300 kg CO<sub>2</sub>. This is likely to be an underestimate of fossil C release from the soils to be used for biofuel production in the UK. On the other-hand, recent work by Hamilton et al. (2007) suggests that agricultural liming can actually cause *sequestration* of atmospheric CO<sub>2</sub>, by reactions with carbonic acid.

There is clearly a need for clarification of these issues. We recommend that these issues are communicated to IPCC, and that estimation of fossil C release from soils (and liming materials) should be changed from a lime-based approach to a fertiliser-based approach. There is a need for further research to relate fossil C release from soils to agronomic practices.

## 4.5 Grain drying

Wheat grain is generally traded at 15% moisture content (mc), oilseed rape seed at 8% mc, and the grain needs to be kept at or below this moisture content for safe storage (see HGCA Grain Storage Guide). In the field, grain can be harvested at moisture contents between ~12% and 25%mc for wheat, and ~5% to 13% mc for oilseed rape. The amount of grain drying on a typical tonne of grain will vary considerably between fields, farms and years. Grain from many individual fields may not be dried at all, whilst for other fields the majority of the grain may be dried. Whether or not grain is dried will be dependent on weather conditions, harvesting capacity of the farm.

Whether or not grain from an individual field is dried (and how much it is dried) will be fairly arbitrary, depending on weather conditions at the time that the field is harvested. If grain drying of specific biofuels crops is to be reported, it is possible that growers could choose to avoid drying biofuels crops by prioritising these fields for harvest when conditions are dry. However, the total amount of drying on the farm may be unaffected by this, if it simply means that a greater proportion of the non-biofuel crops are dried. In terms of minimising GHG emissions, it makes no difference whether the energy used in drying was used to dry a crop for biofuels or for food.

It is likely that there will be a geographical divide in the amount of grain drying with more in the North where harvest is later and there are fewer 'dry' combining days than in the South and East. The harvesting capacity in relation to the area to be harvested will also be

important; to spread the large capital costs of combine harvesters farmers may maximise the area to be cut per machine. To do this it is often necessary to ensure that the combine is operating at all times when the crop will thresh, even if the grain has a fairly high moisture content, for example early in morning and late into the evening. In order to harvest in these conditions the farm must have sufficient capacity to deal with and dry the large quantities of moist grain. Farmers with more limited drying facilities, and ample harvesting capacity, are likely to be more reluctant to combine when the grain is above 15% moisture content.

There is potential for biofuels producers and merchants to not overly penalise grain with moisture contents between 15-16% to avoid potentially unnecessary energy costs of over-drying. It may be that mixing of high and low moisture content grain by merchants could avoid an amount of grain drying, and it may make sense for this to be incentivised.

There is also a wide range in grain drying equipment, and the relative energy requirements and consequent GHG emissions of different drying systems varies considerably, ranging from around 2MJ/kg of water removed for a bulk drying system to potentially over 5 MJ/kg for high temperature dryers (Mclean, 1989 – page 27-30).

Previous LCA studies (LowCVP 2004; Mortimer 2004; Woods et al. 2006), including the GHG calculator, have incorrectly assumed that grain was dried to 3% moisture content. This has led to inflated GHG costs associated with grain drying in these studies. Care is therefore needed in comparisons with these studies.

In terms of calculating emissions from drying for a biofuel crop there are a number of factors to be considered:

- The proportion of grain that is dried
- The average amount of water removed (%mc reduction)
- The energy requirement/GHG emissions of the drying system per %mc removed/t grain.

In principle it could be relatively straight-forward to calculate emissions from drying per tonne of grain for an individual farm. As growers may be unlikely to know the exact energy specifications of their drying equipment it would be possible to have a range of default figures for different drying systems. It may be difficult for growers to give exact figures for the amount of grain drying on their farm, as this is not usually routinely recorded; these figures are likely to be informed estimates. Such figures would be difficult to verify.

There are not currently good data available on the amount of grain drying, or types of grain dryers used on UK farms. There is a need to survey farm practice to estimate the size and importance of the issues around grain drying, and to develop sensible and realistic default values. There may be an opportunity to begin this survey process through the HGCA farm audit questionnaires.

Although the evaluation of such emissions can be complex, given all the major parameters and options which have to be accounted for, it is possible to accommodate this within LCA calculations. This has been achieved with DEFRA Project NF0434 "Bioenergy Technologies Environmental Impact Assessment Tool" which is currently being completed for the Department of Environment, Food and Rural Affairs by AEA Technology plc and North Energy Associates Ltd. This work includes the preparation of workbooks for the calculation of primary energy inputs and CO<sub>2</sub>, CH<sub>4</sub> and N<sub>2</sub>O emissions associated with 110 different combinations of biomass energy technologies currently relevant to the UK. Within these technologies, conventional production of biodiesel from oilseed rape and bioethanol from wheat are covered.

This work extends and updates the Biomass Environmental Assessment Tool which was developed by AEA Technology plc and North Energy Associates Ltd for the Environment Agency in 2005. One particularly relevant improvement is the incorporation of a number of

drying, cooling and storage options which can accommodate changes in input and output moisture content, drying methods and storage times. It should be noted that, with the addition of this expanded capability, it has been necessary to account for the influence of the moisture of the oilseed rape and wheat on the subsequent yield of biodiesel and bioethanol, respectively. The drying options incorporated into the workbooks consist of natural drying, bulk drying (with electric fans), batch drying (with diesel fuel) and continuous drying (with diesel fuel) and cooling (with electric fans). All options, apart from natural drying, also include cooling by electric fans in storage.

## 4.6 Farm Diesel usage

The GHG costs from farm diesel use generally make up a relatively small proportion of the total GHG costs from cropping, around 5-10% using the GHG calculator.

There are different approaches possible for estimating farm diesel used in tractors etc for the cultivation, agrochemical and fertiliser application and harvesting involved in growing biofuels crops. Ideally it would be best to work from direct knowledge of the amount of diesel used per ha. This may be possible on some farms with good records, but it is likely to be difficult and complicated on the majority of farms, where it will not be simple to allocate the diesel used to different fields or crop types.

Another approach would be to use knowledge of individual tractors, in terms of horse power/diesel usage and tractor hours to calculate total emissions. Again, this may not be simple in terms of available data or getting to a figure for an individual ha. Given that farm diesel normally accounts for less than 10% of the total emissions from cropping, it may not be worthwhile collecting highly detailed farm data.

The simplest approach to calculating farm diesel use may be from knowledge of the field operations undertaken. There is an inherent energy requirement for different cultivations that is generally independent of tractor power, size or implement width (P Metcalfe, pers comm.). Energy requirements of different field operations have been collated by Williams et al. (2006a) and could be used with type and number of field operations to give diesel use for individual farms. However, there is a wide range in energy requirements for cultivations with coefficient of variation typically around 40%. Much of this variation will be due to differences in soil type, with light loamy soils being easier to work than heavy clays. It may be possible to adjust the energy requirements according to soil type using adjustment factors such as described in Williams et al. (2006a).

Various combinations of cultivations used in modern reduced-tillage cultivation equipment may not be well described in terms of energy requirements. Whilst, energy requirements of cultivation operations are relatively independent of tractor power and working width, this may be less so for surface operations such as spraying or harvesting. However, these operations tend to need less energy than cultivation operations.

The biggest difference in diesel use that farmers have control over is cultivation strategy; ploughing and power harrowing is likely to use more diesel per ha than minimal cultivation techniques. This gross difference would be relatively easily accounted for in a GHG calculator or accreditation scheme. There may be more subtle differences in diesel usage between cultivation types or strategies that may be worth accounting for. It is also possible that differences or improvements in tractor fuel efficiency could be worth accounting for. There is a need to assess more fully the differences that exist in GHG emissions between different farms and practices, and to consider whether new 'greener' technologies or practices may be sought with sufficient incentives.

On some farms there can be considerable post harvest movement of grain by road pulled by agricultural tractor. Haulage by agricultural tractor is less efficient than by lorry so there may be justification for including this as a separate input to fuel used in cultivation.

There is a question over whether machinery wear & tear and depreciation costs should be included in calculations. Whilst this is included in some LCA (e.g. Williams et al. 2006a; Mortimer & Elsayed 2006) it is not included in others (e.g. LowCVP 2004, Woods et al., 2006). The current GHG calculator does not include machinery wear & tear costs.

## **4.7 Straw incorporation or removal**

A considerable effect of straw removal on fertiliser requirements of subsequent crops has been assumed in many LCA studies (LowCVP, 2004; Mortimer, 2004; Woods et al., 2006). In previous versions of the GHG calculator, removing straw resulted in an increased requirement in fertilisers from 46kg/ha to 164kg/ha for K, from 41kg/ha to 53kg/ha for P, and from 185kg/ha to 253 kg/ha for N.

In practice, decisions on fertiliser applications are generally made independently of straw removal. Whilst nutrient content from straw is deemed important for P and K and may be accounted for to some extent in maintenance dressings, the difference of 118kg/ha for K seems to be excessive. Increasing N applications for N fertiliser after cereal straw removal however, doesn't reflect commercial practice. Indeed, the high C:N ratio of cereal straw can actually result in immobilisation of soil mineral nitrogen, and N applications are sometimes actually *increased* when straw is incorporated. There is therefore little justification for such a strong relationship between straw removal and N fertiliser requirement. Straw incorporation can maintain or increase soil organic matter levels which have been discussed in section 4.3.

## 5 Conclusions and Future Research Requirements

In order to maximise the potential benefits of the biofuels industry, and in particular to maximise GHG savings, there is a need to promote farm-level reporting of GHG emissions to allow value to get down to growers and to incentivise improvement in GHG intensity of biofuel crop production. The continued parallel development of the science-base and the practical tools necessary to implement farm-level GHG auditing is required.

This work has shown that whilst there are a range of important issues that remain to be resolved before farm level carbon reporting can become basic farming practice, these issues are not insurmountable. The Farm audit trials and development of the calculator show that it is possible to take data from farms to get individual GHG intensities. The reporting phase of the RTFO with conservative default values should encourage the use of such farm audits. Issues of approach such as allocation procedures have implications on the final carbon intensities and potentially on behaviour, though ultimately an approach should be adopted that reflects reality. There is a need for consensus across stakeholders and the LCA community in the approaches adopted. Continued development of the farm audits is necessary to demonstrate to the farming and biofuel production communities that the collection, compilation and evaluation of farm-level data are both practical and accurate.

### 5.1.1 N<sub>2</sub>O emissions

The biggest uncertainty surrounding GHG intensity concerns N<sub>2</sub>O emissions. The IPCC approach advocated in the proposed RTFO Carbon reporting methodology (DfT 2007) provides the simplest, most transparent and defensible basis for quantifying N<sub>2</sub>O emissions and may be suitable in the first instance. It is appropriate that emissions are driven by N fertiliser emissions. However, the approach currently ignores emissions from organic N sources (manures/compost/sludge), organic soils and crop residues. Going forward, it will be necessary to account for N<sub>2</sub>O emissions from these sources using a version (or adaptation) of the IPCC approach. Regard will have to be given to any potential consequences, intended or not, on on-farm practices that could result from this approach: Ignoring 'borrowed' nitrogen may mean that crops receiving N from organic soils, animal manures or first wheats may receive reduced fertiliser N and so have more favourable carbon intensity, whilst fertiliser use across the farm won't have changed. These issues will need to be reviewed before economic incentives derive from low carbon intensities, to avoid encouraging inappropriate practices.

In terms of producing a least-worst methodology for dealing with N<sub>2</sub>O emissions, it is recommended here that:

- organic additions are accounted for using the IPCC approach on the basis of *available* N content rather than *total* N content.
- Crop residues are accounted for using the IPCC approach assuming a modest N addition that is included irrespective of yield, N fertiliser and whether or not straw is removed.
- That appropriately large emissions should be assumed for cropping on organic and humose soils.

There is a need to reconcile the IPCC approach to N<sub>2</sub>O emissions, DNDC outputs and findings from recent work (Crutzen et al. 2007). Whilst the work of Crutzen et al. (2007) suggests that real N<sub>2</sub>O emissions from cropping may be higher than the IPCC approach,

there is considerable evidence from field experimentation and modelling with DNDC that the IPCC approach may significantly *overestimate* the real N<sub>2</sub>O emissions from cropping in the UK. In this case, biofuel production in the UK could be unfairly penalised. Given the markedly different conditions and climates in different countries of production there is a need to evaluate whether using the same IPCC default emission factors for all countries is appropriate, or even regions within the UK. It would be possible to advocate a regional approach to N<sub>2</sub>O emissions, using DNDC to calculate emissions from crop types in specific regions for specific soil types assuming certain N fertiliser and manure inputs. However, such an approach would mean that GHG emissions from farms producing crops with lower N inputs and reduced N<sub>2</sub>O emissions would not be fairly accounted for, and thus activities to reduce N<sub>2</sub>O emissions would not be properly incentivised.

The most promising approach for the future for quantifying N<sub>2</sub>O emissions on a farm by farm basis will be to use different emission factors for different scenarios, e.g. soil types, climates, regions, etc, as per Tier 2 of the IPCC methodology. Such emission factors could be derived using UK-DNDC in combination with experimental and field validation.

Generally, it will be important that changes to the approach used for quantifying N<sub>2</sub>O emissions in carbon reporting methodologies can be made as more accurate approaches and emission factors are developed.

### **5.1.2 Other issues of uncertainty**

There is significant uncertainty over the emission factors used for N fertiliser manufacture. The different emission factors assumed in the RTFO draft Carbon Reporting methodology (Department for Transport, 2007b) give substantially higher emission factors for ammonium nitrate over urea (6.8 vs 2.9 kg CO<sub>2</sub>eq/kg N respectively). Such a difference potentially penalises the UK (where AN is predominantly used for N fertiliser) against other parts of the world. Given that many of the N fertiliser manufacturing plants in the UK and Western Europe have N<sub>2</sub>O abatement technologies, there is a need to assess the difference in GHG costs of different N fertiliser products, to ensure that appropriate emission factors are used for the UK situation. It is also important that if use of urea is effectively incentivised by carbon reporting methodologies, full consideration is given to the likely impacts on national and global ammonia emissions.

It has become clear through this research project that there may be significant additional CO<sub>2</sub> emissions associated with cropping through the acidification of lime and chalk that have hitherto been ignored. The IPCC methods assume that CO<sub>2</sub> release only occurs from applied materials, and not from chalky soils. It is suggested here that it may be appropriate to base calculations of CO<sub>2</sub> emissions on emission factors related to the acidifying nature of the nutrients applied. There is a need for further work to clarify this issue and determine what emission factors, if any, should be used.

There is a good deal of uncertainty over the most appropriate default values to use for grain drying. Different methods of grain drying may have very different energy requirements which ought to be accounted for. There is also uncertainty surrounding the diesel used in farm cultivations, with the true benefits of minimal cultivation techniques on fuel use being unclear and difficult to quantify in a carbon reporting approach.

## **5.2 Identified research requirements**

There are two broad areas of research needed with regard to developing the direct quantification of farm-level biofuel-based GHG balances. They can be split into issues that are solely relevant to biofuels and those that are required to understand the GHG impacts of agricultural production systems in general.



1. Establish direct coupling between the farm audit questionnaire and the GHG calculator. The main areas to be resolved are:
  - i. Derive robust land-use change indicators (direct and indirect)
  - ii. Adequately quantifying actual energy use in cultivations
  - iii. Develop methodologies for estimating energy use in grain drying
  - iv. Fertiliser requirements and plant-available nutrient estimates throughout a rotation
  - v. Develop new combined audit and calculator

The following issues are relevant to biofuels but also to any agricultural production system.

2. Fertiliser management (mainly nitrogen) and impact assessments:
  - i. Provide detailed analyses of in-field N<sub>2</sub>O emissions. Evaluate the appropriateness of the IPCC emission factors for N<sub>2</sub>O emissions from UK arable biofuel cropping. Approaches for dealing with organic manures, crop residues, organic soils and baseline emissions from non-cropped land need to be developed and evaluated. Given the relative paucity of published data on N<sub>2</sub>O emissions from arable soils, and the large expense of experimental N<sub>2</sub>O measurement, the UK-DNDC model will be useful in answering these questions.
  - ii. There is a need to evaluate the most appropriate emission factors for fertiliser manufacture for ammonium nitrate and other N fertiliser products in the UK. The variation in manufacturing emissions between products, manufacturing plants and country of origin needs to be assessed.
  - iii. Exploration of how N fertiliser rates could be optimised for GHG savings could be very instructive for the agricultural and biofuels industry. The N fertiliser rates that maximise GHG savings should be determined, and the economic costs of optimising GHG savings should be assessed.
  - iv. The potential for using grain N% (or grain protein) as a 'signature' for GHG emissions from nitrogen needs to be evaluated.
3. Quantify the CO<sub>2</sub> emissions resulting from the acidification of lime or calcareous soils. The current understanding in the literature needs to be reviewed, and there may be a need for experimentation.
4. Develop globally agreed standardised allocation procedures for co-products
5. Develop and employ standardised comparative reference systems – requires the development of a global land use inventory
6. Explore the role of increasing crop productivity in meeting the feedstock demands from the growing biofuels industry, and in delivering improved GHG intensities per tonne.

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