

October 2015



## **Project Report No. 548**

### **Minimising nitrous oxide intensities of arable crop products (MIN-NO)**

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This is the final report of a 60 month project (RD-2008-3474) which started in July 2009. The work was sponsored through the Defra Sustainable Arable LINK programme (LK09128), including a contract for £300,000 from AHDB Cereals & Oilseeds.

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## 1. Abstract

The MIN-NO project (2009 to 2014) used multi-site industry data, field experiments and modelling to improve estimates of nitrous oxide (N<sub>2</sub>O) emissions associated with major UK arable crops and their products. Of 24 field experiments conducted in widely contrasting rainfall, soil and crop conditions, 21 showed direct N<sub>2</sub>O emissions due to fertiliser nitrogen (N) to be less than the 1% default emission factor (EF) assumed by the Intergovernmental Panel on Climate Change. A simple model summarising these emissions predicted a 30-year average EF for arable land across the UK of only 0.46% of N applied.

A set of 'smart' EFs was devised for consideration by UK stakeholders, based on the MIN-NO model, other MIN-NO results and associated evidence<sup>1</sup>. The smart EF for fertiliser N predicted a decrease in emissions of almost 10% of the previously estimated total N<sub>2</sub>O-N emission from UK agriculture (which excludes fertiliser manufacture). The greenhouse gas (GHG) intensity estimated with the MIN-NO smart EFs (which include reduced GHG from fertiliser manufacture) expressed as emissions per tonne of UK feed wheat was 20% less than the 'benchmark' GHG intensity using a current default methodology. Smart EFs also gave reduced GHG intensities for harvested rapeseed, similar intensities for sugar beet and increased intensities for vining peas. Thus most UK arable food products are likely to have smaller GHG intensities than are being estimated at present. Also, biofuels made from N-fertilised crops could be considered more effective in reducing GHG emissions than is currently assumed.

However, prospects for mitigation of N<sub>2</sub>O emissions associated with UK arable cropping are less than was thought previously. Farmers already using abated N fertilisers and following good practice lack any easy means of further mitigation. Feasible approaches tend to have economic costs, so further mitigation depends on the arable industry finding ways of capturing financially some of the value. Four feasible options were identified and, if all of these were aggregated, a combined GHG emissions mitigation potential of around -30% was estimated for the harvested produce of most crops, and from -5% to -25% for their food or fuel products. The best mitigation options appeared to lie in employing more sophisticated crop nutrient supply systems, and / or growing more N-efficient crops through better-informed selection of species and varieties. Other options, such as cultivation strategies to improve soil conditions, cannot be advocated without further research.

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<sup>1</sup> Note that final EFs adopted for use in the UK agricultural GHG inventory and for GHG accounting by other stakeholders may differ from these values. For example, data from additional field trials are being taken into account in the [Defra-funded GHG Research and Development Platform](#).

## 2. Abbreviations

AEA	<a href="#">Ricardo-AEA</a> was formed in 2012 when Ricardo acquired AEA Technology Plc., the privatised offshoot of the United Kingdom Atomic Energy Authority.
AFOLU	The 'agriculture, forestry and other land use' sector, as defined by IPCC.
AG	Above ground
AHDB	<a href="#">Agriculture and Horticulture Development Board</a>
AIC	<a href="#">Agricultural Industries Confederation</a>
AN	Ammonium nitrate
BAT	Best Available Technologies; applied to fertiliser manufacture.
BER	Break even ratio: the price of fertiliser N divided by the price of the same weight of grain (or other crop produce)
BG	Below ground
BNF	Biological nitrogen fixation
C	Carbon
CALM tool	Carbon Accounting for Land Managers: footprinting tool operated by CLA
CLA	<a href="#">Country Land and Business Association</a>
CO <sub>2</sub>	Carbon dioxide
CO <sub>2</sub> e	Carbon dioxide equivalents: the standard unit adopted for quantification of all GHGs weighted for their respective global warming potentials.
CT	Carbon Trust
DM	Dry matter
DNDC	'DeNitrificationDeComposition': the process-based model most widely used internationally to estimate nitrous oxide and other GHG emissions from agricultural systems. See <a href="http://www.dndc.sr.unh.edu/">http://www.dndc.sr.unh.edu/</a>
EC	European Commission
EF	Emission factor. As employed here, EFs are coefficients for which values are set within schemes designed to enable estimation of GHG emissions. These 'EFs' include coefficients that only indirectly enable estimation of GHG emissions, such as the fractions of applied N emitted as NH <sub>3</sub> or leached as NO <sub>3</sub> .
EU	European Union
GHG	Greenhouse gas
ha	Hectare
HI	Harvest index
ILUC	Indirect land-use change: a change in land use in one place attributable indirectly to actions elsewhere
IPCC	Intergovernmental Panel on Climate Change
LEAF	<a href="#">Linking Environment and Farming</a>
LCA	Life-cycle assessment

LpE	Linear plus exponential; function to describe how yield responds to fertiliser N
LSD	Least significant difference
LTA	Long-term average
MIN-NO	This Project's acronym: MINimising Nitrous Oxide intensities of arable crop products.
N	Nitrogen
NFU	<a href="#">National Farmers Union</a>
NH <sub>4</sub>	Ammonium
N <sub>2</sub> O	Nitrous oxide; having a global warming potential of 298 kg CO <sub>2</sub> per kg
N <sub>2</sub> O-N	Nitrous oxide nitrogen: N <sub>2</sub> O contains 28 / 44 = 63.64% N.
NO <sub>3</sub>	Nitrate
N <sub>minGHG</sub>	The rate of applied N that minimises the GHG intensity of crop produce.
N <sub>opt</sub>	Optimum N; the rate of applied N that maximises economic returns from a crop, determined as the N rate at which the incremental yield response to N equals the BER
NUTS2	Nomenclature of Units for Territorial Statistics, Level 2. NUTS2 codes of the UK refer to 37 Level 2 regions for statistical accounting.
ORC	<a href="#">Organic Research Centre</a>
OSR	Oilseed rape
P	Probability
PAS2050	Publicly Available Specification for the assessment of the life cycle greenhouse gas emissions of goods and services (British Standards Institution, 2011).
RED	Renewable Energy Directive
SD	Standard deviation
SE	Standard error
SMN	Soil mineral nitrogen: the sum of nitrate N and ammonium N.
SNS	Soil nitrogen supply, the quantity of N acquired by a crop receiving nil applied N.
SOM	Soil organic matter
t	Tonne
TIAN	Total GHG Intensity of Added N: used to quantify GHG impacts of crop N delivery systems, including manufacturing emissions, direct emissions from soil, and indirect emissions due to NH <sub>3</sub> emission and re-deposition, and (if related to added N) leached NO <sub>3</sub> .
WFPS	Water Filled Pore Space: an indicator of soil oxygen status.
Y <sub>opt</sub>	Economic optimum crop yield; i.e. yield at N <sub>opt</sub> .

### 3. Executive summary

#### ***Approach and the 'State of Play'***

A collaborative project was conducted from 2009 to 2014 involving 23 partners from government, industry and academia. The project aimed to improve estimates of N<sub>2</sub>O emissions associated with production of major UK arable crops (cereals, sugar beet, oilseeds and pulses) and their products, so as to help improve estimates of greenhouse gas (GHG) emissions reported in the UK's inventory and in commercial GHG accounting (carbon footprinting) procedures. It provided evidence of direct nitrous oxide (N<sub>2</sub>O) emissions due to major UK arable crops and suggested better means of estimating and mitigating these and other GHG emissions associated with arable crop production and products. The project involved (i) sensitivity analysis of existing GHG accounting procedures for crops and their products, (ii) field experiments to measure N<sub>2</sub>O emissions associated with manufactured nitrogen (N) fertiliser use, cultivation of pulses and soil incorporation of crop residues, and (iii) modelling to generalise the findings for all UK crops and crop products, and for the National GHG Inventory.

Based on national statistics, 'Benchmark' crops of winter wheat (both for animal feed and breadmaking), winter oilseed rape (OSR), sugar beet and vining peas were defined with typical yields and input levels. GHG emissions for the N fertilised benchmark crops ranged from 2.5 to 3.5 t CO<sub>2</sub>e ha<sup>-1</sup>, as estimated using 'Standard' GHG accounting procedures based on emission factors (EFs) defined by the Intergovernmental Panel on Climate Change (IPCC, 2006). Estimated emission intensities of harvested produce were 443, 518, 974, 45 and 294 kg CO<sub>2</sub>e tonne<sup>-1</sup> respectively, of which from 43% to 79% was related to fertiliser N use or N<sub>2</sub>O emissions.

An extensive sensitivity analysis used farm-level yield and husbandry data provided by industry relating to 880 fields of wheat, 350 of OSR, 510 of sugar beet and 34 of vining peas; this showed wide ranges in GHG intensities. The main causes of variability were fertiliser N rate and yield, but fertiliser choice, soil organic matter (SOM) levels and crop residue removals were also important. Further industry data showed contributions of crop-related emissions to the GHG footprints of food products (bread, chicken meat, cooking oil, frozen peas, sugar and whisky) estimated with the methodology set out under Publicly Available Specification 2050 (PAS2050; British Standards Institution, 2011) and fuel products (bioethanol from wheat and sugar beet, and biodiesel from OSR) estimated with the Renewable Energy Directive approach (RED; European Commission, 2009) ranged from 20% to >80%. Variability in GHG due to crop production thus affected GHG intensities for crop products accordingly.

## **Experimentation**

Twenty four experiments tested the effects on crop yield and direct N<sub>2</sub>O emissions of five rates of manufactured fertiliser N from nil to 160% (or more) of recommended levels over three seasons, involving feed and breadmaking wheat varieties, spring and winter barley, winter OSR, and sugar beet. Three further experiments compared crop yields and N<sub>2</sub>O emissions associated with winter and spring beans, vining and dry-harvested peas, with wheat having no N applied as the control. The same experiments also tested effects on N<sub>2</sub>O emissions of removing crop residues of all these pulse crops as well as fertilised wheat, OSR, and sugar beet.

The fertiliser experiments largely supported the assumption that, excluding a background emission, direct soil N<sub>2</sub>O emissions relate directly to fertiliser N rate; they did not generally support the hypothesis that annual cumulative direct soil N<sub>2</sub>O emissions might relate better to the N-balance (N applied less N uptake). About half of the responses in emissions ha<sup>-1</sup> were non-linear, and about half the responses in emissions t<sup>-1</sup> (intensity of crop produce) showed minima as N rate increased. However, soil mineral N data and N<sub>2</sub>O emissions measured through the weeks following N applications showed these to be largely an artefact of some N rates being confounded with different N timings – small N rates were applied on one date, whereas large N rates were applied on two dates.

The N<sub>2</sub>O emissions associated with pulses were small during crop growth but larger emissions occurred, probably due to premature death of root nodules e.g. after harvest of immature peas for vining. Removal of 'dead' crop residues (<2%N) after harvest of cereals, OSR or pulses caused no significant effects on N<sub>2</sub>O emissions; however, removal of green residues (>2%N), e.g. sugar beet tops, reduced emissions in two of three experiments, on average by 1.2% of the N removed.

## **Modelling**

The simulation model, DNDC, did not predict measured emissions satisfactorily so a statistical model ('the MIN-NO model') was developed that related the natural logarithm of observed total annual direct soil N<sub>2</sub>O emissions to the fertiliser N applied, annual rainfall and soil clay content. Crop type or SOM effects were not significant. When extrapolated nationally (using 5 km grid scale activity data), this model predicted (i) background emissions ranging from 0.2 to >1.5 kg N<sub>2</sub>O-N ha<sup>-1</sup>, (ii) a weighted UK average EF for N<sub>2</sub>O-N from applied fertiliser N of 0.46% (SD 0.07%), and (iii) a total annual UK fertiliser N-related direct N<sub>2</sub>O emission from all arable land of 1.7 Mt CO<sub>2e</sub>.

Taking into account other recent UK research on N<sub>2</sub>O emissions and GHG accounting, a set of 'smart' EFs was proposed for UK Tier 2 GHG accounting and life cycle assessment (LCA) to best represent key causes of arable N<sub>2</sub>O emissions, and the most obvious opportunities for mitigation. These included EFs relating applied fertiliser N due to its manufacture (3.52 kg CO<sub>2e</sub> kg<sup>-1</sup> N) and to

direct soil N<sub>2</sub>O emissions (the UK weighted average being 0.46% of N applied), a ‘background’ emission (UK weighted average, 0.69 kg N<sub>2</sub>O N ha<sup>-1</sup> year<sup>-1</sup>) in place of emissions previously related to some crop residues (those considered to be ‘dead’), and an indirect emission from leached nitrate that was crop-type-related rather than related to applied N, as by IPCC. Compared to using standard EFs, smart EFs predicted reduced GHG intensities for the harvested produce of wheat and OSR, similar intensities for sugar beet and increased intensities for vining peas.

### **Key conclusions**

- Based on multiple robust measurements, a simple statistical model (the ‘MIN-NO model’), and comprehensive data on annual cropping in the UK, it is clear that N<sub>2</sub>O emissions averaged across arable land in the UK are less than are predicted by IPCC guidelines: the new estimate for just direct soil emissions is 1.7 Mt CO<sub>2</sub>e smaller than that previously estimated by IPCC EFs (Table 46).
- Compared to the default IPCC EF of 1%, direct N<sub>2</sub>O emissions from soil due to fertiliser use on arable crops across the UK were estimated to average at 0.46% of the N applied. This was unaffected by crop type but subject to interacting effects of rainfall and soil type (% clay), such that fertiliser-induced emissions could be larger than the default IPCC EF in the wetter regions of the UK.
- Compared to the default IPCC EF of 1%, direct N<sub>2</sub>O emissions from soil due to returned and incorporated crop residue N (from straw, haulm and leaves) were negligible over the first 12 months, except where these residues contained more than 2% N (e.g. sugar beet leaves).
- All arable land emitted significant additional N<sub>2</sub>O, unrelated to recent N additions as fertiliser or crop residues. These N<sub>2</sub>O emissions were estimated to range from 0.2 kg ha<sup>-1</sup> N in the drier East and South to 1.5 kg ha<sup>-1</sup> N in the wetter West and North; they are likely to arise from SOM and be influenced by many factors, including levels of organic N inputs over recent years and soil cultivations that cause soil N to mineralise. After comparison with smaller ‘background’ emissions from unfertilised, undisturbed land e.g. grassland, it is suggested that these emissions could be attributed in GHG accounting schemes to arable land, perhaps best defined as ‘cultivable land that annually has a period without crop cover’.
- The abatement of N<sub>2</sub>O emissions from manufacture of N fertilisers used in the UK has reduced GHG intensities of arable food and biofuel products substantially: for bread by 7%, bioethanol from wheat by 15%, and biodiesel from OSR by 16%.
- Compared with the ‘benchmark’ GHG intensity of 445 kg CO<sub>2</sub>e t<sup>-1</sup>, estimated for grain from an average UK crop of feed wheat using PAS 2050 methodology, the equivalent GHG intensity based on smart EFs was ~350 kg CO<sub>2</sub>e t<sup>-1</sup> – a reduction of over 20%.

### ***Messages for industry and policy***

- Most arable food products have significantly smaller GHG footprints than are being estimated by or on behalf of industry at present.
- Biofuels made from N-fertilised crops grown in the UK are more effective in reducing GHG than was previously thought. The impact of this finding will be enhanced further if the UK defines NUTS2 regional emission estimates for biofuels in a similar way to that suggested by the MIN-NO model e.g. depending on regional rainfall.
- Mitigation of arable GHG emissions by reduced use of fertiliser N was estimated to be largely ineffective if indirect effects on land uses elsewhere were acknowledged.
- As proposed in recent UK reviews, many potential GHG mitigation methods may be applicable to arable crops; these can be classed into four distinct themes,
  - i. Fertiliser systems (methods of manufacture, formulation, application and timing) with low GHG emissions per kg nutrient. (These should probably include the use of chemical inhibitors of soil processes, but exclude 'clever' fertiliser timing because this was judged to be impractical.)
  - ii. Selection of species, varieties and / or fertiliser systems that convert soil and fertiliser N more efficiently into harvestable biomass.
  - iii. Sourcing of crop produce from regions with low rainfall and light soils hence low N<sub>2</sub>O emissions. (Whilst having benefits to individual businesses, global benefits of this approach might be near-neutral, due to displacement effects.)
  - iv. Removal of crop residues, if green; this applies to a minority of crops.Individually these approaches were estimated to have maximum mitigation potentials (on GHG intensities of crop produce) of -25%, -23%, -23% and approximately -16%.
- The maximum GHG mitigation potential derived by aggregating all four mitigation approaches was around -30% for the harvested produce of most crops (grain, seed or root), hence from -5% to -25% for their food or fuel products, depending on the contribution of crop produce to total GHG footprint of the product.
- Thus there are opportunities for industry to help further mitigate the GHG footprints of arable products through improved fertiliser systems (better regarded as 'crop nutrient supply systems') e.g. incorporating chemical inhibitors within fertiliser products, but their exploitation will depend on finding means of capturing some of the value e.g. through economic incentives offered by the supply chain.
- Any improvements that the plant breeding industry can make in the N Use Efficiency of crop varieties will prove beneficial to GHG mitigation, but the scope will be modest, especially if further progress is made in fertiliser technology, because mitigation is multiplicative, not additive.
- The main opportunities for farmers to mitigate N<sub>2</sub>O emissions lie in selecting crop species and fertiliser systems. Unfortunately farmers using abated N fertilisers and following best

practice have few other means of effective N<sub>2</sub>O mitigation at present (at least that could affect calculated GHG emissions). Even under-fertilising with N is counter-balanced by GHG effects through indirect land use change (ILUC).

- Thus the scope for the UK arable industry to further mitigate GHG intensities of its products is less than previously estimated, and GHG mitigation maxima could only be achieved if adequate and sustained incentives became available to support development and use of all the appropriate technologies.

### ***Project achievements and highlights***

- Through a combination of design and luck, the MIN-NO project has quantified N<sub>2</sub>O emissions across extreme contrasts in growing conditions, particularly rainfall, so in predicting emissions across the UK it largely proved possible to generalise by interpolation rather than extrapolation. The main conditions untested here are the combination of high rainfall and light soil.
- The hypothesis that direct N<sub>2</sub>O emissions should have a non-linear relationship with N applied was not universally upheld; many relationships were linear, and occurrence of non-linear relationships was more easily explained by these arising as artefacts of N rate treatments being confounded with N timing differences, than by any biological explanation.
- However, frequent measures of soil mineral N (SMN) soon after N applications have revealed large perturbations in available N that may help to explore the causes of the high variation commonly seen in recovery of applied N by crops.
- This Project did not test nitrification inhibitors. However, Defra Project AC0213 ('Potential for nitrification inhibitors and fertiliser nitrogen application timing strategies to reduce direct and indirect N<sub>2</sub>O emissions from UK agriculture') has shown positive results and with this knowledge, we were able to explore ('theoretical') mitigation practices that might halve direct N<sub>2</sub>O emissions from soil.
- The LCA review showed that most elements of a footprint have similarly large uncertainties and that sensitivities of estimated GHG intensities to uncertainties are largely predictable from knowing crop contributions to the full product footprint. The LCA review confirmed the importance of fertiliser N, and quantified the extent to which fertiliser N mitigations have already reduced GHG footprints (through manufacturing improvements) and might reduce these further. The LCA review also revealed boundary problems with some of the larger contributors to the footprint e.g. current classification of soils according to their organic matter content is clearly inadequately crude; the soils classes of mineral, organic or peaty need to be replaced by a continuous scale relating emissions to SOM content. Note that there are multiple technical benefits to farmers of knowing topsoil organic matter contents, so this information should become commonly available.

- The simulation model DNDC proved unfit for the purposes intended in this project. This finding should have positive influences on both science (because the limits to understanding have been thrown into closer focus) and practice (because the adoption of inaccurate GHG estimates has been avoided). A more generic benefit is that simulation models should attract closer scrutiny before being adopted for use in practice.
- The consortium successfully brought together scientists and practitioners concerned with both GHG inventory reporting and carbon footprinting of food and fuel products. Members of the consortium have engaged comprehensively and actively with the work of the project. Attendance at six-monthly steering meetings has been excellent, with additional meetings arranged to satisfy their wish to engage with the technical findings. Consortium members have engaged actively with the preparation of this report.

### ***Opportunities for further progress through research***

Whilst this Project successfully improved UK estimates of N<sub>2</sub>O emissions associated with arable crop production, and its products, its conclusions were reached with varying levels of certainty. Thus further research on various aspects of arable N<sub>2</sub>O emissions would be beneficial. The most important opportunities for progress are listed at the end of this report (Section 11). In summary, research funders could benefit from better understanding of soil N dynamics; this could lead to improved management of cultivations, crop residues and irrigation according to climate and soil characteristics (especially SOM and texture) so as to maximise crop productivity whilst reducing both direct and background N<sub>2</sub>O emissions.

## 4. Introduction

### 4.1. The problem

The atmospheric abundance of the greenhouse gas (GHG) nitrous oxide (N<sub>2</sub>O) has increased by 20% above pre-industrial levels with food production contributing to 80% of this increase (Stocker *et al.*, 2013). Emissions of N<sub>2</sub>O are particularly important, as it is a powerful GHG with a global warming potential 298 times greater than carbon dioxide (CO<sub>2</sub>) (IPCC, 2007). By ratifying the Kyoto Protocol the UK agreed to achieve a 12.5% reduction in GHG emissions of 1990 levels by the 2008 to 2012 reporting period. Further, more challenging GHG emission reductions are required from agriculture (in common with all other sectors) in order to meet the reduction targets set by the UK Climate Change Act 2008, as detailed in the UK government's Low Carbon Transition Plan (Anon, 2009). A 34% reduction in GHG emissions below 1990 levels is required by 2020 and an 80% reduction by 2050 (Anon, 2009). Agriculture, therefore, has a significant role to play in helping to achieve these GHG emission reduction targets. GHG emissions from anthropogenic sources are reported annually in the UK GHG inventory, which is calculated using the Intergovernmental Panel on Climate Change (IPCC) methodologies (IPCC, 1997, IPCC 2006).

The current UK GHG emissions inventory (Webb *et al.*, 2014, inventory year 2012) estimates that c.85% of total N<sub>2</sub>O production is from agriculture, amounting to 96.2 Kt N<sub>2</sub>O (equivalent to 29.8 Mt CO<sub>2</sub>) of which more than 90% originates from soil. N<sub>2</sub>O emitted *directly* from agricultural soils accounts for c.35% of total GHG emissions from UK agriculture on a carbon dioxide equivalent basis (CO<sub>2</sub>e). The most important sources of *direct* N<sub>2</sub>O emissions from soil are estimated to arise through fertiliser nitrogen (N) applications, from grazing (urine) returns, after the incorporation of crop residues and following the application of livestock manures. A further c.20% of UK agricultural GHG emissions are emitted *indirectly* either following N loss via ammonia (NH<sub>3</sub>) volatilisation and NO<sub>x</sub> emission, or after leaching of nitrate (NO<sub>3</sub><sup>-</sup>). Part of the N *indirectly* lost from agricultural soils is assumed to be subsequently converted to N<sub>2</sub>O in the wider environment. Furthermore, although not included in agricultural emissions at the national level, N<sub>2</sub>O (and other GHG) emissions associated with manufacturing of fertiliser N are included within commercial GHG accounting procedures applied to arable food, feed and fuel products.

There is an urgent global challenge in providing sufficient primary production to sustain a rapidly expanding global population with increasing demands for food, feed and fuel without exacerbating climate change and other environmental impacts of agriculture (Tilman *et al.*, 2001; Reay *et al.*, 2012). Temperate cropping systems, such as those in the UK, will have an increasingly vital role in food production, but most crops receive significant amounts of manufactured N fertilisers, which can be associated with large losses of N<sub>2</sub>O. It is anticipated that farming practices will need to be modified in order to contribute to the government's ambitious targets for mitigation of GHG

emissions (Rees *et al.*, 2013), yet without compromising food production. New N fertiliser strategies are likely to be a vital part of this, but these must be based on the most accurate estimates of N<sub>2</sub>O emissions and the factors affecting them.

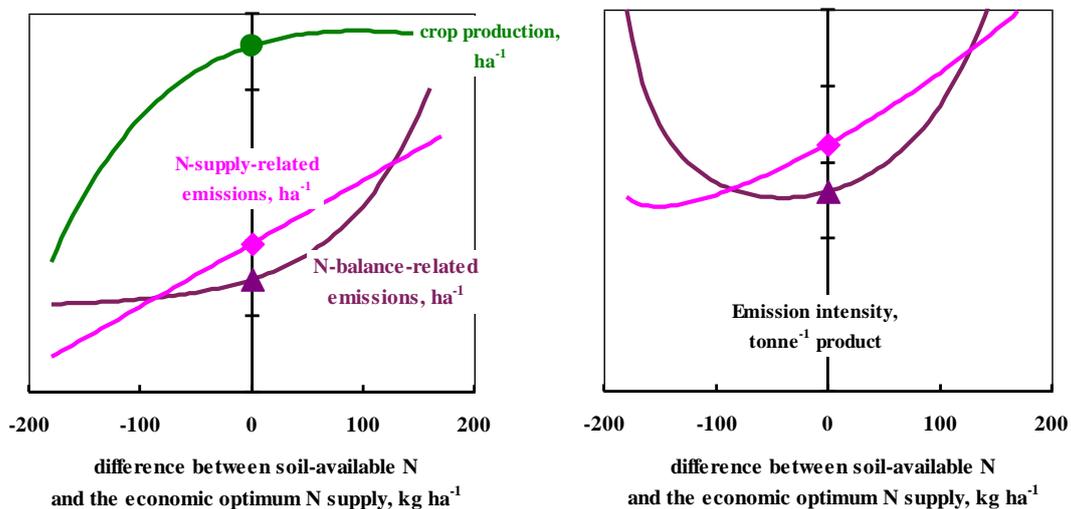
According to estimates agreed by the IPCC, GHG emissions associated with most crop products are dominated by N<sub>2</sub>O (Reay *et al.*, 2012). In national GHG inventories where *direct* N<sub>2</sub>O emissions from soil are calculated using the standard Tier 1 IPCC methodology (IPCC, 2006) and in current commercial GHG accounting procedures, *direct* N<sub>2</sub>O emissions from soil are assumed to be linearly related to N inputs (Figure 1a) and emissions are predicted to be the same for all crops and conditions (Bouwman *et al.*, 2002). This generic and globally applicable method states that, irrespective of source, 1.0% of total-N applied is emitted as N<sub>2</sub>O-N (IPCC, 2006). As this approach is based on *direct* N<sub>2</sub>O emissions measurements from areas including tropical, sub-tropical and other non-temperate climates where soils and agricultural systems are likely to differ substantially from those in the UK, it should now prove valuable to develop N<sub>2</sub>O inventories and mitigation strategies that are more obviously tailored to UK cropping and conditions.

Irrespective of the scope for N<sub>2</sub>O mitigation, there is also an urgent need for better information to support GHG accounting at a field or farm scale. The agricultural industry in England is working together to reduce emissions (without compromising domestic production) through the Agriculture Industry GHG Action Plan (Industry Delivery Partners Group, 2011). The Action Plan aims to help farmers, growers and land managers to adapt to changing climatic conditions alongside reducing GHG emissions, and to understand and reduce their own carbon footprint. Estimating emissions of cropping and other agricultural activities is important to help identify appropriate mitigation options and a range of tools are available to calculate on farm emissions (e.g. the CALM tool; the AHDB Cereals & Oilseeds GHG Calculator; etc.). Estimating GHG emissions of products also allows consumers to make informed purchasing decisions and offers positive marketing opportunities for producers and retailers, especially where carbon footprints are shown to improve over time. Guidelines to support GHG accounting of UK products have been set out in PAS2050 in an attempt to ensure a common approach to GHG accounting (BSI, 2008). The biofuels industry is in the vanguard of GHG accounting for arable products. In Europe this industry operates in a policy framework with the primary aim of reducing GHG emissions; it is crucial for producers to ensure and show that GHG savings from biofuels exceed 40% over fossil fuels. The Renewable Energy Directive (RED) (European Parliament, Council, 2009) prescribes the way that GHG emissions resulting from growing and processing biofuel crop feedstock must be calculated and, for this purpose, GHG emission intensities are expressed per unit of product (all emissions divided by all saleable outputs), hence cropping practices, especially those associated with fertiliser inputs, need to be optimised in relation to emission intensities (e.g. Kindred *et al.*, 2007). Output per hectare (yield) is important in diluting GHG emissions per tonne, but yield is also important through its

indirect effect on land-use-change (Kindred *et al.*, 2008), so it is vital that studies of fertiliser use and N<sub>2</sub>O emissions are referenced against crop yields i.e. they must be yield-scaled.

## 4.2. The hypothesis

The IPCC assumption of linearity between *direct* N<sub>2</sub>O emissions and fertiliser N rate implies that drastic reductions in N fertiliser use and crop productivity would be required to minimise N<sub>2</sub>O emission intensities of crop products (kg N<sub>2</sub>O-N per kg product). Supported by limited UK evidence (Defra Project AC0101; Cardenas *et al.*, 2010) and recently reinforced through a review on linear and non-linear dependency of N<sub>2</sub>O on fertiliser N rate (Kim *et al.*, 2013b), we hypothesised at the outset of the project (in 2009) that the response of annual N<sub>2</sub>O emissions to fertiliser N supply must, to some extent, be related to the surplus of N supply over crop N uptake (i.e. it must be balance related) rather than being linearly related to fertiliser N supply itself (supply-related) (Figure 1a). If this hypothesis were upheld, fertiliser N application strategies to minimise N<sub>2</sub>O emission intensities of arable crop products might be significantly changed, with less severe implications for crop productivity (Figure 1b). The hypothesis illustrated in Figure 1 aligns with a similar analysis by van Groenigen *et al.* (2010). If N<sub>2</sub>O emissions were *entirely* N-balance related, N amounts to minimise N<sub>2</sub>O intensities might be rather similar to current use, with little effect on crop productivity. Whilst N<sub>2</sub>O emissions which occur soon after fertiliser application are likely to relate to the amount of N just applied, later N<sub>2</sub>O emissions are likely to relate more to the balance between N supply and crop N uptake.



**Figure 1. (a) Effects of N supply on crop production (circle), and on N<sub>2</sub>O emissions if related directly to N supply (diamond; as estimated by the IPCC Tier 1 approach) or to the balance between N supply and crop N uptake (triangle; as hypothesised here). (b) Consequent contrasting effects of N supply on N<sub>2</sub>O emission-intensities of crop products for the IPCC (diamond) and our hypothesised (triangle) scenarios.**

### 4.3. Current understanding

It is understood that *direct* N<sub>2</sub>O emissions from agricultural soil are predominately produced via the microbially mediated processes of nitrification and denitrification which are aerobic and anaerobic respectively (Firestone and Davidson, 1989; Baggs and Philippot, 2010). The most important factors controlling the magnitude of N<sub>2</sub>O emissions include the substrates, nitrate and ammonium, often considered together as 'soil mineral N', the energy sources, organic carbon, the oxygen status, primarily governed and best represented by soil moisture, and topsoil temperature (Dobbie and Smith, 2001; Dobbie and Smith, 2003; Smith *et al.*, 2012). Most N<sub>2</sub>O emissions are associated with topsoils (Conen and Smith, 2000), since atmospheric exchange predominates here, and almost all N fertilisers are surface applied. Following fertiliser application, N<sub>2</sub>O emission from topsoil is unlikely to be limited by the mineral N content, therefore temperature and moisture are likely to be more critical in influencing the magnitude of loss. Fertilised warm and wet topsoils have the potential to generate large N<sub>2</sub>O emissions (Clayton *et al.*, 1997; Smith *et al.*, 2012), although the response of N<sub>2</sub>O to soil moisture content is typically non-linear (e.g. Clayton *et al.*, 1997; Davidson, 1991; Smith *et al.*, 1998a). To indicate its influence through soil oxygen status, soil moisture is commonly described in terms of water-filled pore space (WFPS), defined as the ratio of volumetric soil water content to total soil porosity<sup>2</sup>. This is controlled by water additions from rainfall and irrigation, and by water removals through evapotranspiration and drainage. Consequently, the amount and distribution of rainfall is frequently considered to be a strong driver of N<sub>2</sub>O loss. Production of N<sub>2</sub>O is understood to maximise at a WFPS at which both nitrification and denitrification (from NO<sub>3</sub><sup>-</sup> to N<sub>2</sub>O) can occur but at which the soil is insufficiently anoxic for much reduction of N<sub>2</sub>O to N<sub>2</sub>. Davidson (1991) suggested this to be at 60%, whilst Granli and Bøckman (1994) suggested 70-80%; conditions for maximum emission most probably vary with soil type and conditions (Ball, 2013). UK studies have indicated that maximum N<sub>2</sub>O emissions frequently occur at a WFPS >60% (e.g. Dobbie *et al.* 1999; Dobbie and Smith, 2001; Dobbie and Smith, 2003); in a laboratory study using arable soil, Dobbie and Smith (2001) measured an approximate 30 fold increase in N<sub>2</sub>O emissions as the WFPS increased from 60 to 80%. Thus N<sub>2</sub>O emissions with increasing WFPS are likely to be predominantly from denitrification, after short-term wetting events that cause anaerobic soil conditions.

Whilst the fundamental microbial processes that lead to N<sub>2</sub>O emissions are known, quantifying and predicting emissions in the field is challenging because soil conditions, especially temperature, moisture, organic carbon and available N are very variable spatially and temporally, and their effects interact strongly (Brown *et al.* 2000) causing emissions to be spatially and temporally

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<sup>2</sup> A soil's WFPS is determined simply from its bulk density and moisture content, because particle and water densities are stable at ~2.6 and 1 kg/l respectively. Thus if the density of the dry soil bulk is 1.3 kg/l, half of the soil volume is pore space, of which the water filled portion is indicated by weight loss on drying. Note that, although soil aeration status is almost entirely reported in the literature in terms of WFPS, the converse (soil pore air content) might more directly address the concerns here.

episodic (Dobbie and Smith 2001). Cumulative annual emissions over fields or farms are therefore difficult to measure with certainty; the internationally accepted conclusion is that they are best estimated through a combination of intensive, targeted measurements and interpolation by modelling, and this was also the approach taken here.

DNDC (DeNitrificationDeComposition) is the process-based model most widely used in research to estimate N<sub>2</sub>O and other GHG emissions from agricultural systems (Li *et al.* 1992). DNDC has previously been tested and applied to UK conditions (Brown *et al.* 2002; Rees *et al.*, 2008, Topp and Rees 2008) but mainly in grassland systems. At the outset of this project, this model still required validation for use with arable crops under UK conditions, especially with respect to the main factors predicted to affect emissions (i.e. fertiliser applications and crop residue returns to land).

#### **4.4. Objectives and work packages**

The overall aim of the research described here was to improve estimates of N<sub>2</sub>O emissions associated with production of major UK arable crops (cereals, sugar beet, oilseeds, pulses) and their products, so as to help improve estimates of GHG emissions reported in the UK's inventory and in commercial GHG accounting (carbon foot-printing) procedures. In anticipation of expanding interest in GHG accounting at the field and farm scale, the project was also tasked with identifying the major uncertainties relating to N<sub>2</sub>O emissions (soil type, fertiliser N applications, and crop residue returns to land) and with suggesting and prioritising strategies (including incentivisation) to mitigate N<sub>2</sub>O emissions. The project was initiated before inception of the Defra-funded GHG Platform Programme (2011-2016; [www.ghgplatform.org.uk](http://www.ghgplatform.org.uk)). Thus intentions in the project proposal to study effects of nitrification inhibitors were transferred elsewhere in the Defra programme on GHG mitigation (Defra project AC0213; Misslebrook *et al.*, 2014), and freed-up resources enabled better estimation of emissions from removable crop residues in this project. Nevertheless, mitigation effects equivalent to those of nitrification inhibitors were explored in Work-Package 5 (Section 10).

Partners in the MIN-NO project represented all groups with an interest in crop GHG emissions, from the national and policy level (Defra, Scottish Government), through farmers' representatives (AHDB Cereals & Oilseeds, CLA, NFU), the grain trade (Frontier), the supply trade (AIC, Bayer, CF Fertilisers UK, Yara) and food (British Sugar, Warburtons, SWRI) industries, biofuel producers (Vivergo, REA), retailers (Co-op) and research organisations (ADAS, Hill Court Farm Research, SRUC, Rothamsted – North Wyke, North Energy, PGRO, Soilessentials). The project work was organised in five work packages (WP):

1. Desk studies to gauge the importance of and variability in N<sub>2</sub>O emissions associated with crop products at the 'farm gate', as retailed, and according to alternative commonly-used life-cycle assessment (LCA) approaches. Five crops and nine crop products were studied.
2. Experimentation to quantify N<sub>2</sub>O emissions associated with UK arable cropping, as affected by crop species and growing conditions (e.g. regional climate, soil type), and fertiliser use, crop yield and crop residue management. Two sets of experiments were undertaken with treatments in WP2A comparing fertiliser rates, and in WP2B comparing species, including pulses, and the fate of their removable residues (straw, etc.).
3. Desk studies using data from previous research and WP2A to validate the DNDC model of N<sub>2</sub>O emissions under conditions typical of arable cropping in the UK. After testing DNDC, this WP also included some statistical modelling of annual N<sub>2</sub>O emissions found in WP2A.
4. A desk study to develop and recommend appropriate approaches and 'smart' N<sub>2</sub>O EFs for use in GHG accounting procedures involving crop products, national IPCC Tier 2 methodology (relating to arable cropping) and the UK GHG inventory, and finally,
5. A quantitative review of all project findings, to develop financially optimised strategies for minimising N<sub>2</sub>O emissions from UK field cropping, to help minimise the GHG intensities of UK crop products, and to disseminate results from the project.

Each of these WPs will now be described in turn, with a main section devoted to each.

## **5. Sensitivity of greenhouse gas intensities of arable crop products to N<sub>2</sub>O emissions – WP1**

A full report of Work-Package 1 was prepared during the first three years of the MIN-NO Project and is available from the authors, so just a summary of the collated data and the main findings from this work are provided here.

Quantifying GHG emission intensities is important to the agricultural and food industries for a range of reasons. At a national scale the government has committed to reducing GHG emissions by 80% by 2050, and agriculture is expected to play its part in meeting these targets. The industry, represented by ADAS, AEA, AHDB, AIC, CLA, LEAF, NFU, NIAB-TAG and ORC, is working together to reduce emissions through the Agriculture Industry GHG Action Plan (Industry Delivery Partners group, 2011). Quantifying emissions associated with cropping and other agricultural activities is important to help identify appropriate mitigation options and a range of tools has been developed to calculate on-farm emissions (e.g. CALM tool, the AHDB Cereals & Oilseeds GHG Calculator; AHDB 2013). Quantifying GHG emission intensities of products also allows consumers to make informed purchasing decisions and offers positive marketing opportunities for producers and retailers, especially where carbon footprints are shown to improve over time. Guidelines for calculating carbon footprints of products have been set out in PAS2050 (BSI, 2011) in an attempt to ensure a common approach across products. The European biofuels industry operates in a policy framework that aims to reduce GHG emission intensities associated with biofuels; a requirement is to ensure and prove that GHG savings from biofuels exceed 40% (over fossil fuels), and the appropriate method for estimating the GHG emission intensities resulting from growing and processing biofuel feedstocks is prescribed in the European Commission's RED (EC, 2009). It should be noted that N<sub>2</sub>O emissions associated with fertiliser manufacture are excluded from national inventories of agricultural GHG emissions (since they are classed as industrial emissions); however, these emissions are included within the PAS2050-based carbon footprints of agricultural products.

GHG emissions from crops and agricultural products are usually dominated by non-CO<sub>2</sub> emission sources; specifically methane (CH<sub>4</sub>) with respect to non-ruminants and manures, and N<sub>2</sub>O with respect to N fertiliser, manures and crop residues. The MIN-NO project has investigated the levels of soil N<sub>2</sub>O emissions as they relate to N fertiliser application and crop residues, and ultimately has considered how they may be reduced. This summary report from WP1 arises from work with the industry partners to assess the importance of 'N-related emissions' in the GHG emission intensities of various crops and crop products. 'N-related emissions' are taken here to include all of the GHG associated with use of N fertilisers, including CO<sub>2</sub> and N<sub>2</sub>O emitted during fertiliser manufacture, direct soil emissions of N<sub>2</sub>O following fertiliser application, and also due to crop residue

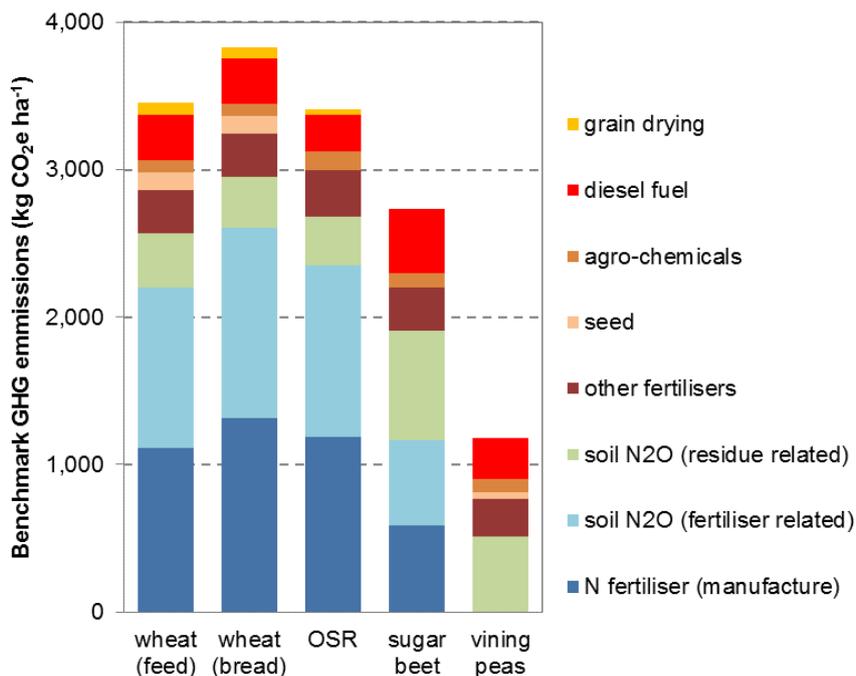
incorporation. The report initially sets up benchmarks for GHG intensities of arable products (Section 5.1); it then explores effects on these that arise from differences in GHG accounting methodologies, and from uncertainties in the emission factors (EFs) which these GHG accounting methodologies employ (Section 5.2). And finally it seeks to understand how variability in estimates of GHG emission intensities is affected by variability in on-farm practices and yields (Section 5.3).

## 5.1. Benchmark GHG and N<sub>2</sub>O intensities

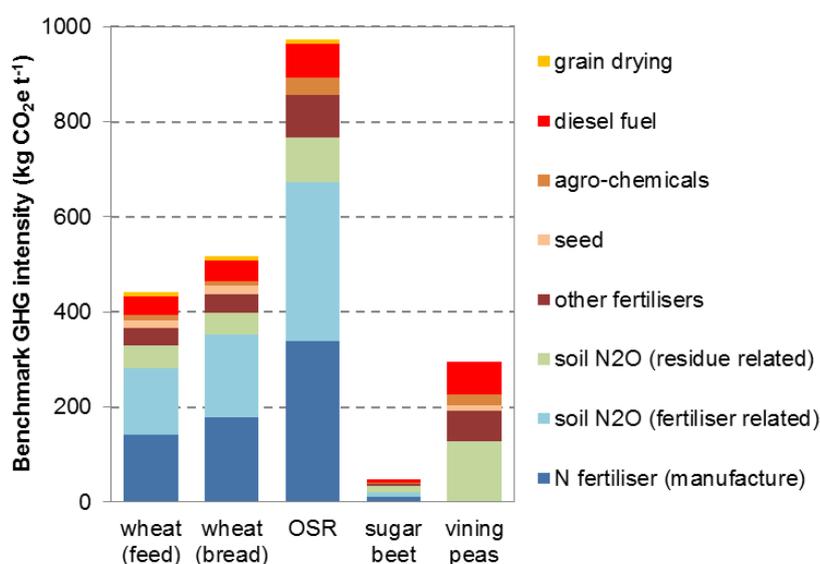
A context for the results generated from farmer's datasets in the MIN-NO project can be provided by deriving a benchmark estimate for each crop using national average data. These average data are summarised in Table 1 and then the derived benchmark GHG emissions (in kg CO<sub>2</sub>e) per hectare (ha) and per tonne (t) of product using the PAS2050 methodology (BSi, 2011) are shown in Figure 2 and Figure 3 respectively. These estimates are based on Defra UK average yields for years 2008-2012 (Defra, 2013) and average N and sulphur (S) fertiliser application rates from the British Survey Fertiliser Practice for 2008-2012 (Defra, 2012). Feed wheat was assumed to yield 0.4 t ha<sup>-1</sup> more than wheat for breadmaking and to require 32 kg ha<sup>-1</sup> less fertiliser N. All N fertiliser was assumed to be ammonium nitrate (AN). To account fairly for applications of P and K fertiliser, which may be applied on a rotational basis for the benefit of all crops, maintenance dressing rates for Index 2 soils were assumed for each crop as recommended in the Fertiliser Manual (RB209; Defra, 2010). A standard lime application of 3.7 t ha<sup>-1</sup> applied annually to 10% of the tillage crop area was assumed (BSFP 2013) giving equal emissions to each of the crops. Carbon dioxide (CO<sub>2</sub>) emissions from the hydrolysis of applied lime (CaCO<sub>3</sub>) were included as per IPCC 2006 guidelines. In actual fact, the requirement for liming is related to acidification of soils by fertiliser and nitrification of ammonium to nitrate, so it could be argued that a higher burden of CaCO<sub>3</sub> provision and hydrolysis should be attributed to crops with higher N applications. Hydrolysis impacts could also be held to occur on calcareous soils not receiving added lime due to their inherent CaCO<sub>3</sub> content (Kindred *et al.*, 2008). However, these impacts are relatively small and the necessarily more sophisticated analysis was not undertaken in deriving these crop benchmarks. Emissions from diesel use for the benchmark crops were estimated assuming a 'typical' set of field operations for each crop with associated diesel fuel use taken from the AHDB Cereals & Oilseeds GHG calculator (AHDB, 2013). Diesel use estimates for sugar beet harvest were provided by British Sugar. Diesel use figures for pea vining were unavailable so figures for combine harvesting were used, although this will have under-estimated true emissions from pea vining. Emissions for agrochemicals are taken from data agreed with Bayer CropScience for each crop.

**Table 1. Crop yields, fertiliser application rates and emissions factors used to derive benchmark GHG emissions estimates.**

		Wheat (feed)	Wheat (bread)	OSR	Sugar beet	Vining peas	EF (kg CO <sub>2</sub> e kg <sup>-1</sup> )
Yield	<i>t ha<sup>-1</sup></i>	7.8	7.4	3.5	57.1	4.0	-
N application rate	<i>kg ha<sup>-1</sup></i>	176	208	188	93	0	6.310
P <sub>2</sub> O <sub>5</sub> application rate	<i>kg ha<sup>-1</sup></i>	60	60	50	50	40	0.731
K <sub>2</sub> O application rate	<i>kg ha<sup>-1</sup></i>	45	45	40	100	40	0.501
SO <sub>3</sub> application rate	<i>kg ha<sup>-1</sup></i>	23	23	57	0	0	0.840
Lime application rate	<i>kg ha<sup>-1</sup></i>	350	350	350	350	350	0.590



**Figure 2. Benchmark estimates of GHG emissions (kg CO<sub>2</sub>e per ha) for selected crops.**



**Figure 3. Benchmark estimates of GHG emission intensities (kg CO<sub>2</sub>e per t) for selected crops.**

On an area basis, benchmark estimates of total GHG emissions associated with wheat for feed and for breadmaking, OSR and sugar beet were relatively similar at 3,845, 3,469, 3,411 and 2,768 kg CO<sub>2</sub>e ha<sup>-1</sup> respectively, but emissions from vining peas were much less at only 1,251 kg CO<sub>2</sub>e ha<sup>-1</sup>. The differences between crops is largely attributable to differences in N fertiliser input and associated N<sub>2</sub>O emissions, being highest for breadmaking wheat and absent for peas. Whilst sugar beet had lower fertiliser N related emissions per ha, this was countered by much greater diesel fuel emissions, largely from the energy intensive harvest and carting operations. Typical total machinery hours were 5.77 h ha<sup>-1</sup> for feed wheat, 6.75 h ha<sup>-1</sup> for OSR, and 21.54 h ha<sup>-1</sup> for sugar beet (of which harvesting accounted for 10.77 h ha<sup>-1</sup>). PAS2050 guidelines do not require GHG emissions associated with the manufacture of capital goods (i.e. machinery) to be included unless their impact on the total footprint is found to be 'material'; however, if these emissions are known, they should be included. Estimates were not available for all crops here; they would be expected to add about 86 or 101 kg CO<sub>2</sub>e ha<sup>-1</sup> for wheat and OSR respectively. In practice, most GHG emission calculations for agricultural products omit emissions from manufacture of machinery.

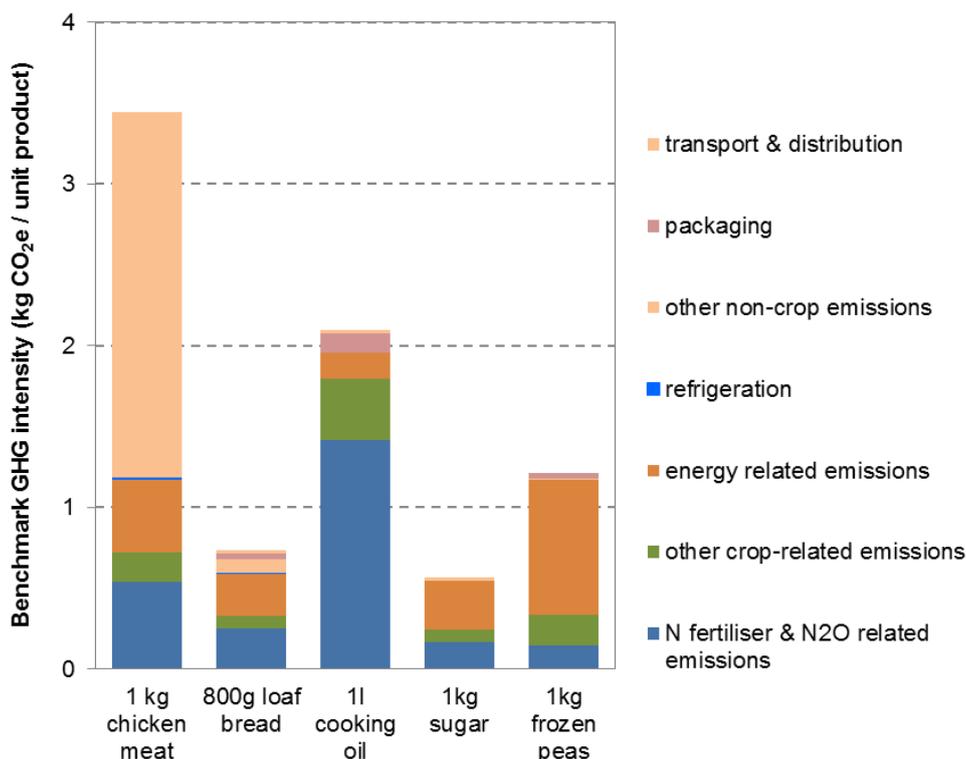
OSR had similar fertiliser-related GHG emissions to wheat, but estimated soil emissions of N<sub>2</sub>O from crop residues were less due to its smaller yield. Overall, the proportion of GHG emissions related to fertiliser N manufacture plus N<sub>2</sub>O soil emissions was 74-77% for wheat and OSR, 65% for sugar beet and 46% for vining peas.

Figure 3 shows benchmark estimates of GHG emission intensities (t<sup>-1</sup>) for each crop (assuming residue incorporation and complete allocation to the harvested product). The benchmark estimate for wheat grown for bread was 16% greater than for feed wheat (518 versus 443 kg CO<sub>2</sub>e t<sup>-1</sup>, respectively) due both to its greater N fertiliser use and its lower yield. The low yield of OSR resulted in it having the highest benchmark estimate at 992 kg CO<sub>2</sub>e per t. The very high yield of sugar beet (57.1 t ha<sup>-1</sup>) resulted in the lowest benchmark estimate of 52 kg CO<sub>2</sub>e t<sup>-1</sup>. However, note that approximately 75% of sugar beet is water, whereas moisture only makes up 15% of wheat grain and 9% of OSR seed. Despite their relatively low yield, vining peas provided a relatively low benchmark estimate of 347 kg CO<sub>2</sub>e per t because no N fertiliser is used.

Benchmark estimates of GHG emission intensities associated with crop-derived food products are shown in Figure 4. Each product was of roughly comparable weight (around 1kg) but GHG emission intensities differed substantially, being greatest for chicken meat due to the large feed requirement of livestock. GHG emissions due to wheat constituted 21% of total GHG emission intensities for chicken meat production, though additional GHG emissions from other crop-derived ingredients (especially high protein pulses etc.) contributed more. Other products showed proportions of total GHG emission intensities due to cropping ranging from 20% for whisky from wheat grain to 86% for cooking oil from OSR. For bread, 45% of total GHG emission intensity was

crop-related, the next largest contributor being energy use for milling and baking at 35%. The majority of total GHG emission intensities for cooking oil were crop-related, with only 7% due to energy required for crushing and processing. In sugar production 47% of total GHG emission intensities were crop-related whilst 50% were due to beet processing. The majority of total GHG emission intensities related to frozen peas were due to the freezing process, accounting for 65% of the total intensities. The ‘N-related’ portions of total GHG emissions were 16% for chicken, 34% for bread, 67% for cooking oil, 30% for sugar, 14% for whisky and 11% for frozen peas.

Furthermore, Figure 5 shows the benchmark GHG emission intensities per litre of biofuels derived from wheat, OSR or sugar beet using the RED methodology. Each of these includes credits for electricity generation in processing the biofuel so that, despite processing being an energy intensive process, GHG emissions attributed to biofuel processing were relatively small compared to crop-related emissions. For wheat bioethanol and OSR biodiesel, cropping made up the majority of total GHG emission intensities (83% and 75%, respectively) whereas for sugar beet bioethanol the proportion due to cropping is less (45%). The proportions of emission intensities due to fertiliser manufacture and direct fertiliser-related N<sub>2</sub>O emissions from soil were 62% for bioethanol from wheat, 58% for biodiesel from OSR and 29% for bioethanol from sugar beet.



**Figure 4. Benchmark estimates of GHG emission intensities of crop-derived food products. ('Other non-crop' refers to other raw materials for bread, and other raw agricultural materials such as feedstocks and energy in housing chickens).**

## 5.2. Sensitivity analyses of benchmark estimates

To assess the impact of changes in GHG EFs on these benchmark estimates of GHG emissions for crops and their products, e.g. reductions in N fertiliser manufacturing (Table 2), a sensitivity analysis was conducted for each factor, with all other parameters being kept constant. Results are summarised in Table 3 to Table 5.

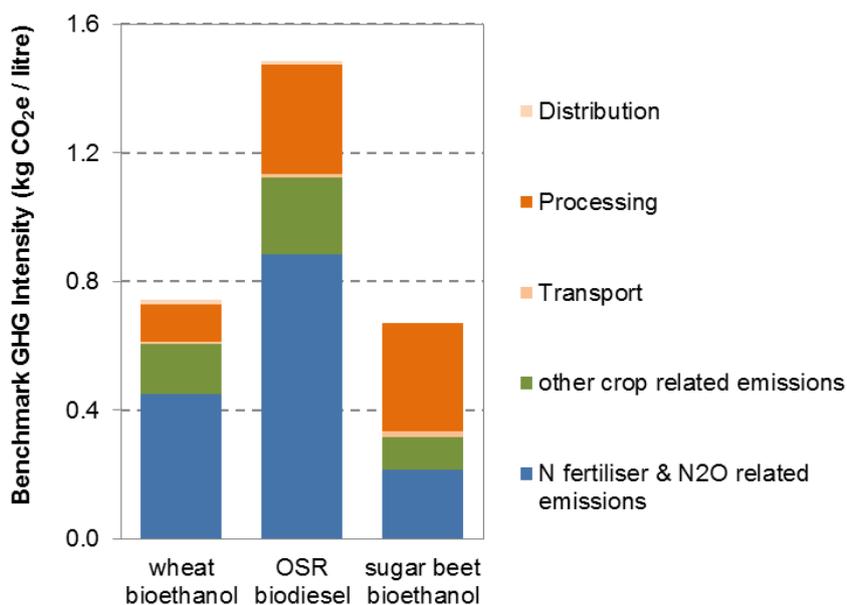


Figure 5. Benchmark estimates of GHG emission intensities from crop-derived biofuels.

### 5.2.1. N fertiliser manufacturing emissions

Values for GHG emissions factors for N fertiliser manufacture used in the sensitivity analyses of benchmark estimates were provided by AIC, CF Fertilisers UK and Yara, and constituted an updated version of values published by Brentrup and Palliere (2008). The production of ammonia from atmospheric N by the Haber-Bosch process is energy intensive and uses natural gas as a hydrogen feedstock, as well as a fuel. Manufacturing plants have been designed to maximise the efficiency of this process and many now approach the stoichiometric limits (Brentrup and Palliere, 2008). The production of nitric acid for AN fertiliser is associated with losses of N<sub>2</sub>O to the atmosphere; these emissions can be abated and most European AN plants now have some degree of abatement. The production of urea fertiliser does not incur N<sub>2</sub>O losses, and it is commonly credited with fixing CO<sub>2</sub> from exhaust gases into the urea molecule. This CO<sub>2</sub> is however released back into the atmosphere when urea is applied to soils and hydrolysed.

Most N fertiliser supplied in the UK is AN, produced either in the UK or sourced in Europe. CF Fertilisers UK operate the only two N fertiliser factories in the UK, at Ince and Billingham. Substantial investments have abated N<sub>2</sub>O emissions from their nitric acid production, so N<sub>2</sub>O (and other GHG) emissions relating to production of UK product have recently been reduced by 40%.

Table 2 gives GHG EFs per kg N for AN and urea. Note that the CO<sub>2</sub> from hydrolysis of urea (equating to 1.587 kg CO<sub>2</sub>e kg<sup>-1</sup> N) has been included within this table to allow fair comparisons. The bulk of this report uses Brentrup and Palliere's (2008) estimates for emissions from N fertiliser manufacture; these relate to the fertiliser industry in 2003. However, AN fertiliser plants in Europe now use processes akin to Brentrup and Palliere's 'Best Available Technologies' (BAT; Hoxha *et al.*, 2011). Work has been on-going by AIC and the European Fertiliser Manufacturer Association (EFMA), now 'Fertilizers Europe', during the course of the MIN-NO project to update the estimates of emissions associated with European fertilisers, to reflect the improved production and abatement technologies now used. Fertilizers Europe (2014) have recently estimated that average manufacturing emissions for European AN in 2011 were almost half of those in 2003.

**Table 2 EFs for manufacture of N fertilisers.**

<b>Fertiliser source</b>	<b>GHG emission factors (kg CO<sub>2</sub>e kg<sup>-1</sup> N)</b>	<b>Reference<sup>#</sup></b>
EU standard AN 2006	6.31	Brentrup & Palliere, 2008
Best Available Technology (BAT) AN	2.77	Brentrup & Palliere, 2008
EU standard urea 2006	3.18*	Brentrup & Palliere, 2008
Best Available Technology (BAT) urea	2.74*	Brentrup & Palliere, 2008
UK AN (abated)	3.40	GrowHow, 2013
Best estimate EU AN 2011	3.52	Fertilizers Europe, 2014
Best estimate EU urea 2011	3.57*	Fertilizers Europe, 2014

\* GHG estimates for urea manufacturing omit CO<sub>2</sub> fixed and subsequently lost through hydrolysis.

# References cited do not use the same methodologies. Only GrowHow 2013 complies with PAS2050.

**Table 3. Effects of fertiliser choice on (a) the contribution of N fertiliser manufacture and N<sub>2</sub>O soil emissions (including crop residues) to total GHG intensities of arable crop produce and (b) the total GHG intensity of arable crop produce.**

	<b>Feed wheat</b>	<b>Bread wheat</b>	<b>OSR</b>	<b>Sugar beet</b>	<b>Vining peas</b>
<i>% emissions which are N or N<sub>2</sub>O related:</i>					
Standard EU AN (2008)	74%	77%	79%	67%	40%
Best Available Technology (2008)	69%	71%	73%	63%	40%
UK AN (abated)	70%	73%	75%	64%	40%
EU updated AN (2014)	70%	73%	75%	64%	40%
Urea (2014)	70%	73%	75%	64%	40%
<i>Change in GHG intensities of produce, compared to standard EU AN (2008)</i>					
Best Available Technology (2008)	18%	20%	20%	12%	0%
UK AN (abated)	15%	16%	16%	10%	0%
EU updated AN (2014)	14%	15%	15%	9%	0%
Urea (2014)	14%	15%	15%	9%	0%

**Table 4. Effects of fertiliser choice on (a) the contribution of N fertiliser manufacture and N<sub>2</sub>O soil emissions (including from crop residues) to total GHG intensities of arable food products and (b) the total GHG intensity of arable food products.**

	<b>Chicke n</b>	<b>Bread</b>	<b>Cookin g oil</b>	<b>Sugar</b>	<b>Frozen peas</b>	<b>Whisky</b>
<i>% emissions which are N or N<sub>2</sub>O related:</i>						
Standard EU AN (2008)	16%	35%	68%	30%	12%	14%
BAT (2008)	12%	28%	61%	26%	12%	10%
UK AN (abated)	13%	30%	62%	27%	12%	10%
EU updated AN (2014)	13%	30%	63%	27%	12%	10%
Urea (2014)	13%	30%	63%	27%	12%	10%
<i>Change in GHG intensities of produce, compared to standard EU AN (2008)</i>						
BAT (2008)	4%	9%	17%	5%	0%	4%
UK AN (abated)	3%	7%	14%	4%	0%	3%
EU updated AN (2014)	3%	7%	13%	4%	0%	3%
Urea (2014)	3%	7%	13%	4%	0%	3%

**Table 5. Effects of fertiliser choice on (a) the contribution of N fertiliser manufacture and soil N<sub>2</sub>O emissions (including crop residues) to total GHG intensities of arable biofuel products and (b) the total GHG intensity of arable biofuel products.**

	<b>Wheat bioethanol</b>	<b>OSR biodiesel</b>	<b>Sugar beet bioethanol</b>
<i>% emissions which N or N<sub>2</sub>O related:</i>			
Standard EU AN (2006)	61%	62%	30%
Best Available Technology (2006)	55%	54%	26%
GrowHow Nitram AN (abated)	56%	56%	27%
EU updated AN (2014)	56%	56%	27%
Urea (2014)	56%	56%	27%
<i>Change in GHG emission intensities of total product</i>			
Best Available Technology (2006)	15%	16%	5%
GrowHow Nitram AN (abated))	12%	13%	4%
EU updated AN (2014)	12%	12%	4%
Urea (2014)	11%	12%	4%

### 5.2.2. N fertiliser-related N<sub>2</sub>O emissions

The calculation of N<sub>2</sub>O soil emissions from N fertiliser using IPCC (2006) guidelines involves a number of steps to quantify both direct and indirect emissions. Direct emissions are taken to be linearly related to fertiliser N, at 1% N<sub>2</sub>O-N (=0.01\*(44/28) = 0.0157 kg N<sub>2</sub>O kg<sup>-1</sup> N or 4.68 kg CO<sub>2</sub>e kg<sup>-1</sup> N). Indirect emissions are assumed to come from the 40% of N that is not taken up by the crop which is assumed to be lost either by volatilisation of ammonia or leaching of nitrate. The fraction of fertiliser N volatilised is assumed to be 10% and the fraction leached 30%. Of the volatilised N, 1% is assumed to be lost as N<sub>2</sub>O-N, and for leached N, 0.75% is assumed to be lost as N<sub>2</sub>O-N. Overall, indirect emissions from volatilisation of fertiliser are 0.00157 kg N<sub>2</sub>O kg<sup>-1</sup> N applied (0.1 \* 0.01 \* 44/28), or 0.468 kg CO<sub>2</sub>e kg<sup>-1</sup> N. Indirect emissions from leaching are 0.00225 kg N<sub>2</sub>O kg<sup>-1</sup> N

applied ( $0.3 \times 0.0075 \times 44/28$ ) or  $1.05 \text{ kg CO}_2\text{e kg}^{-1} \text{ N}$ . Direct and indirect  $\text{N}_2\text{O}$  emissions in relation to fertiliser N applied sum to  $0.0208 \text{ kg N}_2\text{O kg}^{-1} \text{ N}$ , or  $6.205 \text{ kg CO}_2\text{e kg}^{-1} \text{ N}$ .

Each of the EFs used in the calculation of direct and indirect  $\text{N}_2\text{O}$  emissions have associated uncertainty ranges, as shown in Table 6. A crude indication of the total uncertainty in  $\text{N}_2\text{O}$  emissions can be made by calculating emissions using all the lower estimates and all the upper estimates. This has been used to show the possible extremes in the contribution of  $\text{N}_2\text{O}$  in crops and crop derived products (the extreme high and extreme low uncertainty limits). Another scenario has also been used to indicate a more plausible lower estimate to  $\text{N}_2\text{O}$  emissions for UK arable crops, with direct emissions assumed to be 0.005 rather than 0.01  $\text{kg N}_2\text{O-N kg}^{-1} \text{ N}$ , fraction volatilised assumed to be 0.03 rather than 0.1 (this is more appropriate for AN fertiliser; Chambers and Dampney, 2009) and fraction leached 0.15 rather than 0.3 (perhaps more appropriate for heavier soils in drier parts of the UK (Lord *et al.*, 2006). This combination of EFs gave total  $\text{N}_2\text{O}$  emissions (excluding manufacturing emissions) of  $2.88 \text{ kg CO}_2\text{e kg}^{-1} \text{ N}$  and will be referred to as the 'preliminary plausible (PP-) scenario'.

**Table 6. Direct and indirect EFs from IPCC Guidelines in 1996 and 2006 with the uncertainty range quoted by IPCC (2006).**

Emission factor	IPCC Code	Units	1996	2006	Uncertainty range	
					Lower	Upper
Direct from applied N	EF <sub>1</sub>	$\text{kg N}_2\text{O-N kg}^{-1} \text{ N}$	0.0125	0.01	0.0030	0.0300
Applied N volatilized	FracGASF	Fraction	0.1	0.1	0.0300	0.3000
Applied N leached	FracLEAC H	Fraction	0.3	0.3	0.1000	0.8000
From volatilised N	EF <sub>4</sub>	$\text{kg N}_2\text{O-N kg}^{-1} \text{ N}$	0.01	0.01	0.002	0.05
From leached N	EF <sub>5</sub>	$\text{kg N}_2\text{O-N kg}^{-1} \text{ N}$	0.025	0.0075	0.0005	0.025
Direct emissions		$\text{kg N}_2\text{O kg}^{-1} \text{ N}$	0.01964	0.01571	0.00471	0.04714
Indirect emissions (volatilization)		$\text{kg N}_2\text{O kg}^{-1} \text{ N}$	0.00157	0.00157	0.00009	0.02357
Indirect emissions (leaching)		$\text{kg N}_2\text{O kg}^{-1} \text{ N}$	0.01179	0.00354	0.00008	0.03143
Direct emissions		$\text{kg CO}_2\text{e kg}^{-1} \text{ N}$	5.85	4.68	1.40	14.05
Indirect emissions (volatilization)		$\text{kg CO}_2\text{e kg}^{-1} \text{ N}$	0.47	0.47	0.03	7.02
Indirect emissions leaching		$\text{kg CO}_2\text{e kg}^{-1} \text{ N}$	3.51	1.05	0.02	9.37
Total emissions		$\text{kg CO}_2\text{e kg}^{-1} \text{ N}$	9.83	6.20	1.46	30.44

Table 7 shows the impacts of different  $\text{N}_2\text{O}$  EFs on benchmark estimates of GHG emission intensities for various crops. The extreme low  $\text{N}_2\text{O}$  uncertainty limit reduces benchmark estimates of GHG emissions by around 25% for wheat and OSR, and by 12% for sugar beet. By contrast, the extreme high  $\text{N}_2\text{O}$  uncertainty limit increases the benchmark estimates of GHG emissions for wheat and OSR by over 120%, and for sugar beet by 62%. The proportion of emissions related to  $\text{N}_2\text{O}$  increases to over 90% for wheat and OSR. The 'PP-scenario' shows reductions in benchmark estimates of GHG emissions of more than 16% for wheat and OSR, and of 9% for sugar beet.

When these calculations were taken through to crop-derived food products the impacts were still substantial; the PP-scenario reduced benchmark estimates of GHG emission intensities for chicken, sugar and whisky by 4%, bread by 8% and cooking oil by 16%, as indicated in Table 8. Similarly, benchmark estimates of GHG emission intensities from wheat bioethanol and OSR biodiesel are reduced by around 15%, and sugar beet bioethanol by 4%, as illustrated in Table 9.

**Table 7. Effects of using different scenarios for calculating soil N<sub>2</sub>O emissions on (a) the proportion of total GHGs related to N fertilisers and (b) estimates of total GHG emissions.**

	<b>Feed wheat</b>	<b>Breadmaking wheat</b>	<b>OSR</b>	<b>Sugar beet</b>	<b>Vining peas</b>
<i>% emissions N related</i>					
Standard N <sub>2</sub> O EFs	74%	77%	79%	67%	40%
Extreme low uncertainty limit	66%	69%	71%	61%	40%
Extreme high uncertainty limit	89%	90%	91%	82%	40%
PP-scenario	69%	72%	74%	63%	40%
<i>Changes in total GHG intensities (kg CO<sub>2</sub>e t<sup>-1</sup>), compared to standard EFs</i>					
Extreme low uncertainty limit	-24%	-26%	-26%	-16%	0%
Extreme high uncertainty limit	123%	131%	134%	81%	0%
PP-scenario	-17%	-18%	-18%	-11%	0%

**Table 8. Effects of using different scenarios for calculating N<sub>2</sub>O soil emissions on benchmark estimates of GHG emission intensities for crop-derived food products and the proportion related to N fertilisers.**

	<b>Chicken</b>	<b>Bread</b>	<b>Cooking oil</b>	<b>Sugar</b>	<b>Frozen peas</b>	<b>Whisky</b>
<i>% emissions N related</i>						
Standard N <sub>2</sub> O EFs	16%	35%	68%	30%	12%	14%
Extreme low uncertainty limit	11%	26%	58%	25%	12%	5%
Extreme high uncertainty limit	33%	59%	85%	49%	12%	27%
PP-scenario	13%	29%	62%	27%	12%	7%
<i>Changes in total GHG intensities (kg CO<sub>2</sub>e t<sup>-1</sup>), compared to standard EFs</i>						
Extreme low uncertainty limit	-5%	-12%	-22%	-7%	0%	-5%
Extreme high uncertainty limit	26%	59%	115%	37%	0%	24%
PP-scenario	-4%	-8%	-16%	-5%	0%	-3%

**Table 9. Effects of scenarios for calculating N<sub>2</sub>O soil emissions on benchmark estimates of GHG emission intensities for crop-derived biofuels and the proportion related to N fertilisers.**

	<b>Wheat bioethanol</b>	<b>OSR biodiesel</b>	<b>Sugar beet bioethanol</b>
<i>% emissions N related</i>			
Standard N <sub>2</sub> O emissions factors	61%	62%	30%
Extreme low uncertainty limit	52%	52%	25%
Extreme high uncertainty limit	81%	81%	49%
PP-scenario	55%	55%	26%
<i>GHG effect for total product</i>			
Extreme low uncertainty limit	-20%	-20%	-7%
Extreme high uncertainty limit	101%	104%	36%
PP-scenario	-14%	-14%	-5%

### 5.2.3. N<sub>2</sub>O emissions from crop residues

Within the IPCC 2006 guidelines for reporting national inventories of GHG emissions the N in crop residues (both above ground as straw and stubble and below ground as roots) are deemed to cause N<sub>2</sub>O emissions when they break down after soil-incorporation. Most LCA protocols (excluding the RED) take the same approach. However, many issues and uncertainties surround the N<sub>2</sub>O emissions that may arise from crop residues, some philosophical, others technical. Many of these issues have been investigated in Work Package 2 of the MIN-NO project and so are explained and fully discussed in Section 7 of this report. However, in scoping the extent of possible crop residue-related effects here, it should be recognised that residues may have opposing effects through C-sequestration as well as through N<sub>2</sub>O emission, and some approaches choose to exclude crop residue effects from their calculations (e.g. Kindred *et al.*, 2008). This is in keeping with the supposition that N<sub>2</sub>O emissions from crop residues remain unchanged, whether or not the removable portion is removed, and that N<sub>2</sub>O emissions from crop residues constitute a background emission merely associated with cropping, hence avoiding the creation of any perverse incentives concerning crop residue management.

Here we assess the impacts on crop product GHG intensities of different current approaches to quantifying N<sub>2</sub>O emissions from crop residues; these are irrespective of the outcomes of Work-Package 2. Then Section 7 describes Work-Package 2, in which N<sub>2</sub>O emissions from crop residues were measured, and it proposes and discusses new approaches that might be more accurate and appropriate for estimation of N<sub>2</sub>O emissions associated with crop residues. Then Section 10 explores the impacts on crop product GHG intensities of feasible new approaches.

#### **Calculating N<sub>2</sub>O from crop residues**

The IPCC (2006) gives guidance for calculating the N in above and below ground crop residues in order to estimate direct and indirect N<sub>2</sub>O emissions for national GHG inventories. Default values and relationships are provided by the IPCC for a range of crop types (or specific crops) including grains, pulses, tubers, root crops, N-fixing forages, non-N fixing forages, maize, wheat, rice and barley. Relationships with grain yield, adjusted for dry matter content, are used to give estimates of above-ground dry matter of residues. The IPCC uses regression relationships (slopes and intercepts, differing between crops) to enable estimation of crop residue biomass from yield activity data. Estimates of N concentration then allow calculation of total N in above ground residues. Estimates are also given of the ratio of below ground to above ground biomass, to allow calculation of below ground biomass from the above ground estimate, and separate N concentration estimates are given for below ground residues to allow calculation of N in the below ground residues. Direct N<sub>2</sub>O emissions are calculated with the standard EF of 0.01 kg N<sub>2</sub>O-N kg<sup>-1</sup> N returned. Indirect emissions are also calculated assuming leaching of 30% of crop residue N and an associated EF

of 0.0075 kg N<sub>2</sub>O-N kg<sup>-1</sup> N leached. (Note that IPCC assumes no ammonia emissions from crop residues.)

There are a number of difficulties with the application of this methodology (i.e. IPCC, 2006), as follows:

- Appropriate dry matter contents of crops are often not readily known by LCA practitioners.
- Using the slope and intercept from regression analysis and various ratios in order to calculate above ground residues from crop yield is a complex and confusing approach. It doesn't allow for values for specific crops or countries to be easily ascertained. Using a harvest index approach would be much simpler and specific crop data are more readily available.
- More crop specific data are required for AG residue N concentration of different crops, beyond the defaults given. These should be readily available from standard feed references or the like, or they could be obtained at modest cost by laboratory analysis.
- The root crop default is based on groundnuts so is not relevant for root crops in the UK.
- Rooting data are very sparse for all crops, and it is questionable whether use of different root:shoot ratios for different crops can be justified. Recent data for wheat indicates that root biomass is normally around 1 t ha<sup>-1</sup> to 90cm depth for UK crops (Gregory *et al.*, 2005; ADAS unpublished data). Assuming significant in-season turnover of root DM, perhaps this is similar to the 23% of total above ground biomass assumed by IPCC, but given sparse data, 20% is assumed here for all crops.
- There is very little data for N% concentrations of root tissues. The IPCC default root N% for pulses is *less* than that for other crops, which is perplexing given that N fixation occurs in the root system of legumes. Again, given sparse data, a uniform value of 0.9% is assumed here (Pederson *et al.*, 2002).

To show the extent of these difficulties, Table 10 summarises all factors used to calculate N in crop residues and subsequent N<sub>2</sub>O emissions, (i) according to IPCC (2006) defaults for four crop types, and (ii) for four specific widely-grown crop species according to best estimates determined from UK-specific data (where these differ from IPCC factors). Note that values for OSR and sugar beet are not available as IPCC defaults. (The IPCC default for root crops relates to groundnuts, so is not relevant to sugar beet.) UK data include values for N% of residues taken from MAFF (1980) and average harvest indices taken from recently published research reports and papers. The very small root:shoot ratio for sugar beet is based on the limited evidence of Steen and Lindén (2008).

**Table 10. Factors used to estimate the N in crop residues (CR), associated N<sub>2</sub>O and GHG emissions according to IPCC (2006) and again (for specific crops) according to ‘best UK’ evidence (including unpublished data held by ADAS), and effects of CR removal in both cases. BG = below ground; AG = above ground; DM = dry matter; HI = harvest index, i.e. ratio of harvested DM to AG DM.**

	units	IPCC 2006 default factors				Best UK (from literature)			
		Grains	W Wheat	Pulses	Roots	Wheat	OSR	Sugar beet	Vining peas
‘Fresh’ Yield	<i>t ha<sup>-1</sup></i>	3.5	7.8	4	4	7.8	3.5	57	4.0
DM in fresh yield	<i>ratio</i>	0.88	0.855	0.91	0.94	0.85	0.91	0.23	0.33
Slope		1.09	1.61	1.13	1.07				
Intercept		0.88	0.4	0.85	1.54				
HI (implied)	<i>ratio</i>	0.451	0.375	0.423	0.403	0.50	0.35	0.70 <sup>2</sup>	0.25
BG to AG DM	<i>ratio</i>	0.22	0.23	0.19	0.2	0.2	0.2	0.01 <sup>2</sup>	0.2
AG N in DM	<i>kg kg<sup>-1</sup></i>	0.006	0.006	0.008	0.016	0.006	0.007	0.017	0.02
BG N in DM	<i>kg kg<sup>-1</sup></i>	0.009	0.009	0.008	0.014	0.009	0.009	0.009	0.009
AG DM of residues	<i>t ha<sup>-1</sup></i>	8.4	11.1	5.0	5.6	6.63	5.92	5.63	3.96
BG CR DM	<i>t ha<sup>-1</sup></i>	2.5	4.1	1.6	1.9	2.65	1.82	0.19	1.06
AG N in CR	<i>kg ha<sup>-1</sup></i>	50.2	66.8	39.7	89.0	39.8	41.4	95.7	79.2
BG N in CR	<i>kg ha<sup>-1</sup></i>	22.7	36.9	13.1	26.1	23.9	16.4	1.7	9.5
Returned CR N	<i>kg ha<sup>-1</sup></i>	72.8	103.7	52.8	115.1	63.6	57.8	97.4	88.7
Direct N <sub>2</sub> O-N emis’n	<i>kg ha<sup>-1</sup></i>	0.728	1.037	0.528	1.151	0.636	0.578	0.974	0.887
Indirect N <sub>2</sub> O-N emis’n <sup>1</sup>	<i>kg ha<sup>-1</sup></i>	0.164	0.233	0.119	0.259	0.143	0.130	0.219	0.200
Total N <sub>2</sub> O-N emission	<i>kg ha<sup>-1</sup></i>	0.892	1.270	0.647	1.410	0.780	0.708	1.193	1.087
Total N <sub>2</sub> O emission	<i>kg ha<sup>-1</sup></i>	1.402	1.996	1.016	2.216	1.225	1.112	1.874	1.708
<b>GHG emission <i>kg CO<sub>2</sub>e ha<sup>-1</sup></i></b>		<b>418</b>	<b>595</b>	<b>303</b>	<b>660</b>	<b>365</b>	<b>331</b>	<b>559</b>	<b>509</b>
		<i>With straw / haulm removed:</i>							
AG CR removed	<i>ratio</i>	0.5	0.5	0.5	0.8	0.5	0.50	0.8	0.8
AG N returned	<i>kg ha<sup>-1</sup></i>	25.1	33.4	19.9	17.8	19.9	20.7	19.1	31.0
Total CR N returned	<i>kg ha<sup>-1</sup></i>	47.7	70.3	32.9	43.9	43.8	37.1	52.8	42.9
<b>GHG emission <i>kg CO<sub>2</sub>e ha<sup>-1</sup></i></b>		<b>317</b>	<b>274</b>	<b>403</b>	<b>189</b>	<b>251</b>	<b>251</b>	<b>213</b>	<b>255</b>

<sup>1</sup>from leached nitrate; <sup>2</sup>this ratio for sugar beet is just for fine root (from Steen & Linden, 2008); it treats the tap root as AG, so is very small.

The impact of different calculation assumptions for crop residues on benchmark estimates of GHG emissions from crops, crop food products and crop biofuels can be seen in Table 11 to Table 13. Table 11 shows that using the best UK estimates rather than IPCC default values for crop residues results in slightly smaller GHG emissions for wheat, but larger GHG emissions for vining peas.

**Table 11. Effect of different assumptions for calculating crop residues on N-related and total GHG emissions from crops.**

	<b>Feed wheat</b>	<b>Bread wheat</b>	<b>OSR</b>	<b>Sugar beet</b>	<b>Vining peas</b>
<i>% emissions N-related:</i>					
Best UK estimates	74%	77%	79%	65%	46%
IPCC 2006 default values	76%	78%	79%	65%	29%
No emissions	71%	75%	76%	53%	0%
Residue removal	74%	76%	78%	56%	13%
<i>Change in total GHG emissions for crop:</i>					
IPCC 2006 default values	7%	6%	3%	2%	-24%
No emissions	-11%	-9%	-10%	-25%	-46%
Residue removal	-3%	-2%	-3%	-19%	-38%

**Table 12. Effect of different assumptions for calculating crop residues on N-related and total GHG emission intensities of crop-derived food products.**

	<b>Chicken</b>	<b>Bread</b>	<b>Cookin g oil</b>	<b>Sugar</b>	<b>Frozen peas</b>	<b>Whisky</b>
<i>% emissions N-related:</i>						
Best UK estimates	16%	35%	68%	30%	14%	14%
IPCC 2006 default values	17%	36%	68%	30%	7%	11%
No emissions	14%	32%	65%	21%	0%	8%
Residue removal	15%	34%	67%	24%	3%	9%
<i>Change in total GHG emissions for food product:</i>						
IPCC 2006 default values	1%	3%	2%	1%	-7%	1%
No emissions	-2%	-4%	-8%	-12%	-14%	-2%
Residue removal	-1%	-1%	-3%	-9%	-12%	-1%

Omitting N<sub>2</sub>O emissions from crop residues altogether (as is done in some GHG estimation methods) reduces benchmark estimates of GHG emissions by around 10% in wheat and OSR, 25% in sugar beet and 46% in vining peas. Removing residues (i.e. baling and carting off straw) reduces benchmark estimates of GHG emissions by 3% in wheat and OSR (though note that effects on carbon sequestration are not counted by this method).

The impact of the residue calculation method carries through into crop products, having a maximum impact of 2% on chicken, 4% on bread, 8% on cooking oil, 12% on sugar and frozen peas, as shown in Table 12, and around 7-10% on biofuels, as seen in Table 13.

**Table 13. Effect of different assumptions for calculating crop residues on N-related and total GHG emission intensities of crop derived biofuels.**

	<b>Wheat bioethanol</b>	<b>OSR biodiesel</b>	<b>Sugar beet bioethanol</b>
<i>% emissions N-related</i>			
Best UK estimates	61%	62%	30%
IPCC 2006 default values	63%	62%	29%
No emissions	58%	58%	21%
Residue removal	60%	60%	23%
<i>Change in total GHG emission intensities for biofuel products</i>			
IPCC 2006 default values	5%	2%	1%
No emissions	-9%	-7%	-12%
Residue removal	-2%	-3%	-9%

### ***Allocation of emissions from crop residues***

As well as affecting the amount of crop residue returned and, hence, associated N<sub>2</sub>O emissions, the decision on whether to remove or incorporate crop residues has an impact through allocation of GHG emissions (due to cropping) to the ‘parent’ crop. The PAS 2050 methodology (BSi, 2011) uses economic allocation with assumed prices of wheat grain (at £150 t<sup>-1</sup>) and wheat straw in the swath (at £100 ha<sup>-1</sup>); the grain accounts for 92% of the economic output so 92% of total GHG emissions would be allocated to grain, to the point of separation of grain and straw (i.e. all GHG emissions associated with growing the crop but not GHG emissions from drying grain), and 8% allocated to straw. For feed wheat, this reduces the benchmark GHG emission intensity from 467 kg CO<sub>2</sub>e t<sup>-1</sup> to 430 kg CO<sub>2</sub>e t<sup>-1</sup> (-7.7%). For OSR, the lower yields and lower value of rape straw relative to rapeseed give a proportionally higher allocation to the seed; assuming a rape-seed price of £350 t<sup>-1</sup> seed and £50 ha<sup>-1</sup> for rape straw in the swath, 96% of total GHG emissions would be allocated to the seed, so straw removal gives a reduction in the benchmark GHG emission of 3.9%. With the RED methodology (EC, 2009) the crop residues are treated as wastes, so no GHG emissions are allocated to them. Alternative methodologies that allocate residues by mass or energy content would allocate a much higher proportion of emissions to the residues than with the economic allocation used in PAS2050.

### 5.3. Farm-level variability in GHG emissions

#### 5.3.1. Methodology

The GHG emissions associated with five crops and nine crop products were assessed, as follows:

**Crops:**

- Breadmaking wheat;
- Feed wheat;
- OSR (OSR);
- Sugar beet; and
- Vining peas.

**Crop products:**

- 800 g white loaf of bread made from UK milling wheat (in association with Warburtons Ltd.)
- 1 kg of conventional meat from chickens fed with UK feed wheat (in association with the Co-operative Group)
- 1 bottle of whisky (0.7 litre) made from UK feed wheat (in association with SWRI)
- 1 litre of bioethanol made from UK feed wheat (in association with Vivergo Fuels Ltd)
- 1 litre of cooking oil made from UK OSR
- 1 litre of biodiesel made from UK OSR
- 1 kg of sugar made from UK sugar beet (in association with British Sugar plc)
- 1 litre of bioethanol from UK sugar beet (in association with British Sugar plc)
- 1 kg of frozen peas (in association with PGRO).

For each crop, generic Microsoft Excel workbooks were set up that accounted for all major sources of emissions:

- Production and consumption of diesel fuel used in agricultural machinery;
- Provision of seeds;
- Manufacture of N fertiliser;
- Manufacture of P fertiliser;
- Manufacture of K fertiliser;
- Provision of organic fertilisers;
- Manufacture of agrochemicals (fungicides, herbicides, pesticides, etc.);
- Manufacture of other farming inputs (lime, etc.);
- Soil N<sub>2</sub>O emissions from N fertiliser application;
- Soil N<sub>2</sub>O emissions from organic fertiliser application;
- Soil N<sub>2</sub>O emissions from crop residue incorporation;
- Soil CO<sub>2</sub> emissions from urea application;
- Soil CO<sub>2</sub> emissions from lime application;
- Emissions from cultivation of organic soils;
- Production and consumption of fuel used in crop drying, where relevant; and
- Production of electricity used in crop drying, where relevant.

Calculations followed PAS2050 methodology for food products (BSi, 2011) and RED methodology for biofuels (EC, 2009). Standard EFs were used across all crops. To assess field level variation,

input datasets were captured from farmer's fields by industry partners; field numbers for each crop were 656 with feed wheat, 318 with breadmaking wheat, 399 with OSR, 608 with sugar beet and 34 with vining peas. It should be noted that, after screening with an agreed protocol to remove unsuitable or dubious information in a systematic and objective manner, the numbers of these datasets were reduced for subsequent statistical analysis (mainly due to missing data items). It should also be noted that these datasets were not selected or structured to be representative of all or subsets of crops in the UK. It was merely intended that, by merit of large numbers of datasets derived from mainstream organisations within the UK arable industry, the extent of variability and the causes of uncertainty demonstrated from them could be taken as meaningful, and so would be helpful in devising mitigation strategies (see Section 10.4).

Data gathered from the farmers included the following:

- crop yield;
- manufactured fertiliser application rates and types (N, P and K fertilisers);
- organic fertiliser application rates;
- agrochemical application rates (fungicides, herbicides, pesticides, etc.);
- other farming input application rates (lime, etc.);
- crop drying information (initial and final moisture contents and type of drying energy); and
- diesel fuel consumption: (although information on diesel consumption was requested, this was rarely provided in direct form so it was estimated from the specification of farming operations).

N<sub>2</sub>O emissions from fertiliser and crop residues were calculated following IPCC (2006) guidelines, including both direct and indirect emissions.

Estimates of emissions associated with food and biofuel products were made in consultation with industry partners and so included values for generic processes and transportation activities. Details for each product are given with the results.

### **5.3.2. Results – Crops**

GHG emissions were estimated for each crop and product using the above methods. Detailed results are given in the full report from WP1 (available from the authors). The summary below concentrates on wheat and sugar beet as these had the largest datasets. Summary statistics are also given for the other crops and points of interest are noted. An overview of the impacts of N fertiliser type and rates, cultivation type and residue fate is made across all crops.

## Wheat

Figure 6 and Figure 7 show individual estimates of GHG emission intensities per tonne of wheat, derived from 596 feed wheat datasets and 288 breadmaking wheat datasets. Overall, there were large ranges in emission intensities, from 128 to 929 kg CO<sub>2</sub>e t<sup>-1</sup> for feed wheat, and from 210 to 750 kg CO<sub>2</sub>e t<sup>-1</sup> for breadmaking wheat, although the majority of results ranged between 250 and 500 kg CO<sub>2</sub>e t<sup>-1</sup>. The average values and standard deviations for total GHG emission intensities associated with feed and breadmaking wheat at the farm gate were 362 ± 82 kg CO<sub>2</sub>e t<sup>-1</sup> and 420 ± 91 kg CO<sub>2</sub>e t<sup>-1</sup>, respectively.

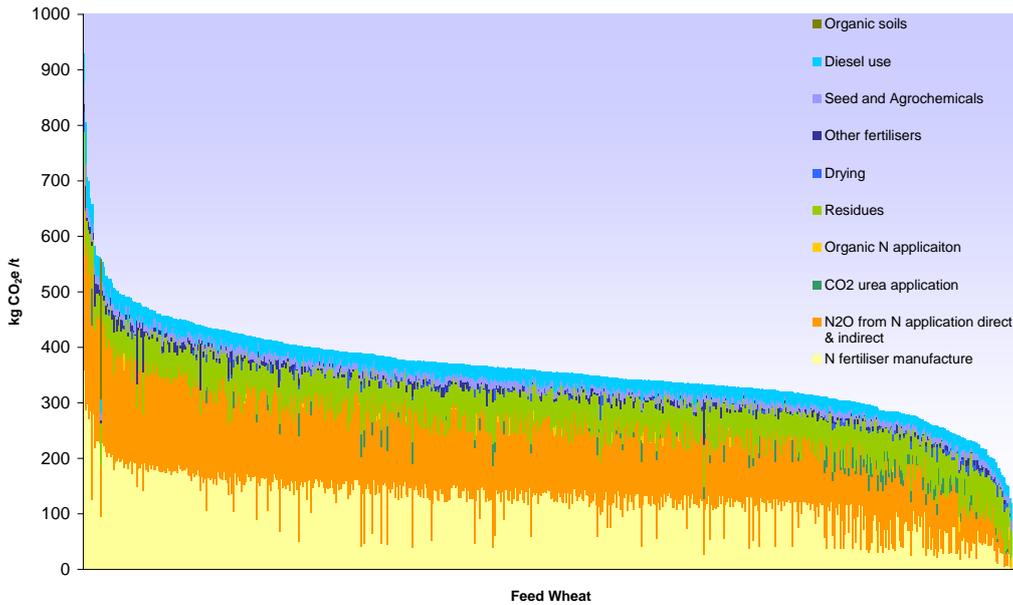


Figure 6. Total GHG emission intensities (kg CO<sub>2</sub>e t<sup>-1</sup>) associated with feed wheat estimated at the ‘farm gate’ with PAS 2050 methodology for 597 crops grown from 2005 to 2012.

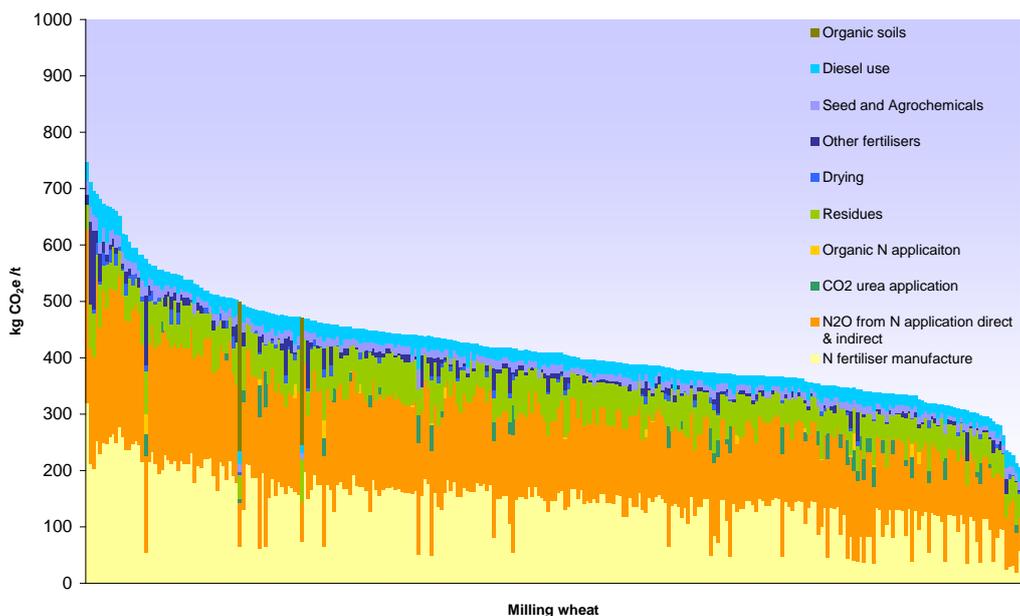
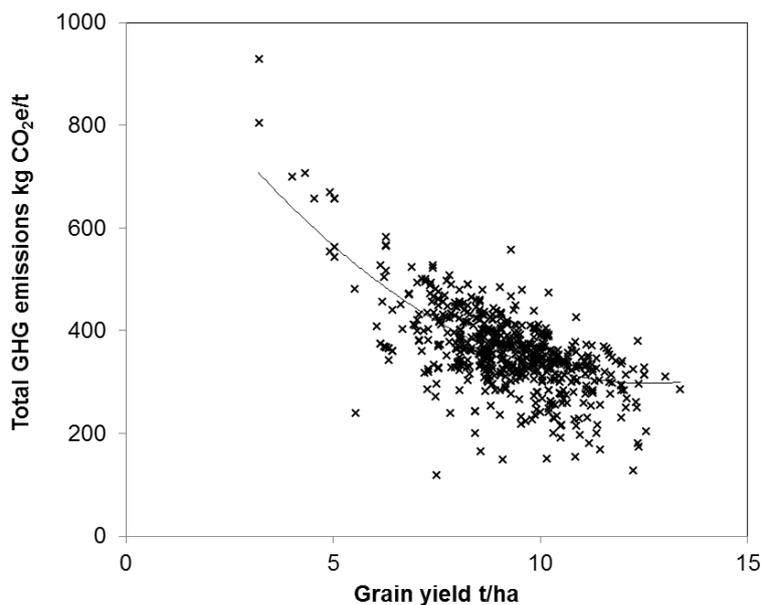
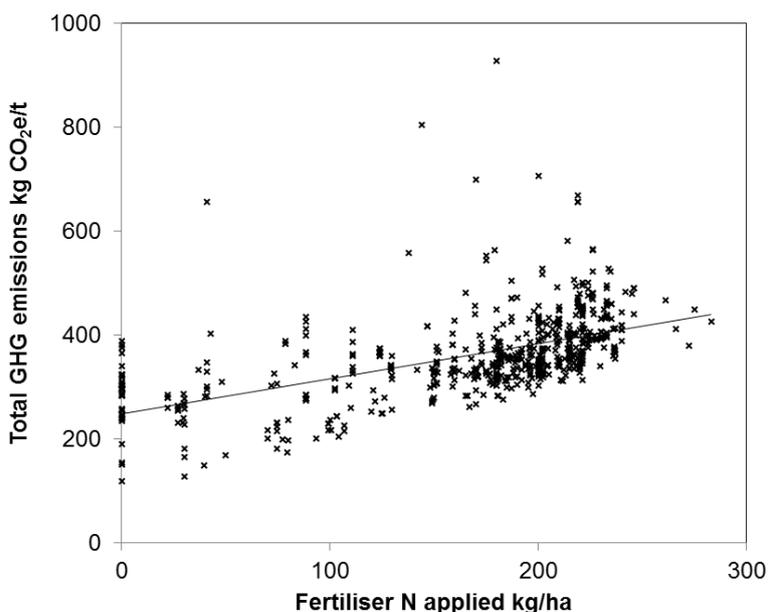


Figure 7. Total GHG emission intensities (kg CO<sub>2</sub>e t<sup>-1</sup>) associated with breadmaking wheat estimated at the ‘farm gate’ with PAS 2050 methodology for 288 crops grown from 2005 to 2012.

Figure 8 and Figure 9 show that major causes of variation in total GHG emission intensities for these crops were grain yields and N fertiliser application rates; the six feed wheat datasets with the highest total GHG emission intensities had unusually low wheat grain yields of less than 5 t ha<sup>-1</sup>; of the 12 feed wheat datasets with lowest total GHG emission intensities, all had fertiliser N application rates less than 85 kg ha<sup>-1</sup> and wheat yields greater than 9 t ha<sup>-1</sup>. It should be noted that no clear relationship between wheat yield and N fertiliser application rate could be readily discerned with these datasets; some high wheat yields occurred with small (or zero) N fertiliser application rates and some large fertiliser N application rates occurred with low wheat yields.



**Figure 8. Relationship between total GHG emission intensity and grain yield for feed wheat datasets (quadratic fit:  $R^2 = 0.45$ ).**



**Figure 9. Relationship between total GHG emission intensity and N fertiliser application rate for feed wheat datasets (linear fit:  $R^2=0.30$ ).**

Within these results for feed wheat crops, GHG emissions associated with N fertiliser manufacture ranged from 0 to 360 kg CO<sub>2</sub>e t<sup>-1</sup> and averaged at 122 kg CO<sub>2</sub>e t<sup>-1</sup>. Fertiliser-associated soil N<sub>2</sub>O emissions ranged from 0 to 310 kg CO<sub>2</sub>e t<sup>-1</sup>, averaging at 160 kg CO<sub>2</sub>e t<sup>-1</sup>. Overall, the range in N-related emissions stretched from 92 to 650 kg CO<sub>2</sub>e t<sup>-1</sup>, averaging 260 kg CO<sub>2</sub>e t<sup>-1</sup>. The proportions of total GHG emissions for feed wheat that are N related ranged from 36% to 94%, averaging 86%.

### Oilseed rape

The on farm variability in total GHG emissions and yields for OSR based on 353 datasets are shown in Figure 10, again with a large range being evident. Again a general trend can be seen for lower total GHG emissions being associated with higher OSR yields, and high total GHG emission intensities related to large fertiliser N application rates and large subsequent GHG emission intensities. Between 23% and 86% of total GHG emission intensities for OSR were N-related, with an average of 65%.

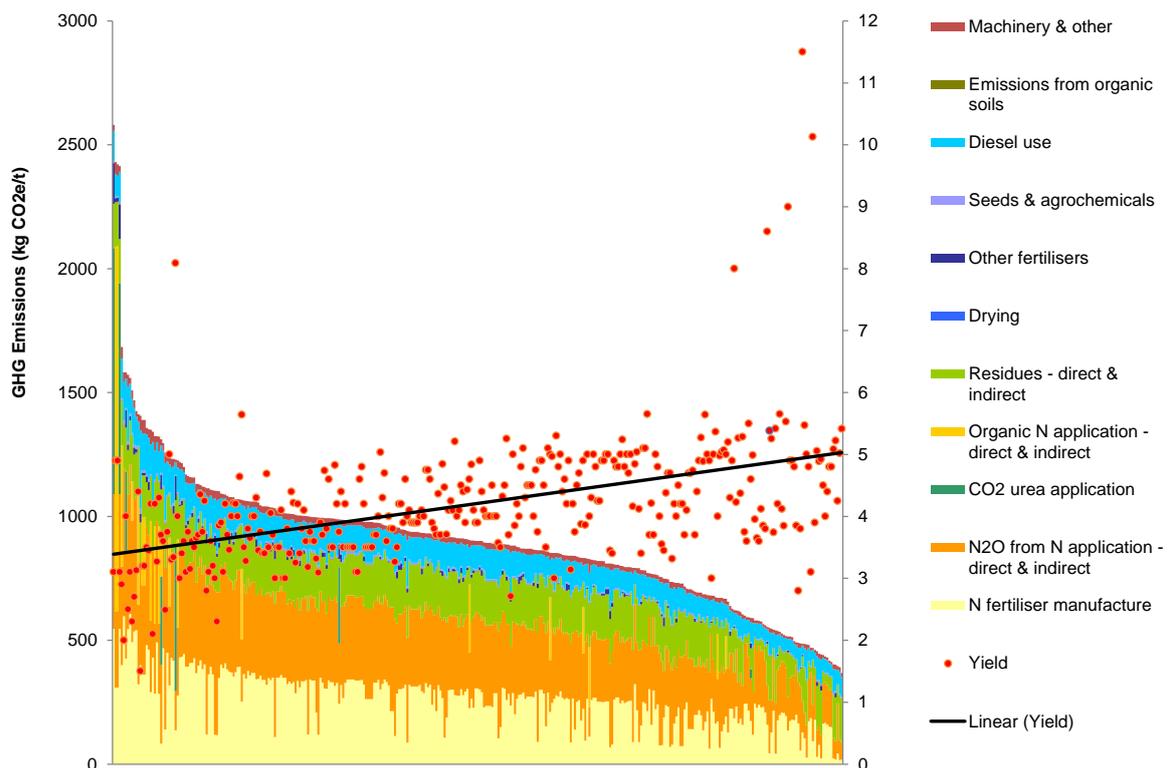


Figure 10. Variability of GHG emission intensities associated with OSR at farm gate for 353 datasets between 2005 and 2012 with the PAS 2050 methodology.

## Sugar beet

Figure 11 shows the variability of total GHG emission intensities, based on 511 datasets, to be even greater for sugar beet than for wheat and OSR, ranging from 29 to 475 kg CO<sub>2</sub>e t<sup>-1</sup>, and averaging at 76 kg t<sup>-1</sup>.

The emission intensities were very strongly correlated with sugar beet yield. In particular, low yields explained the small number of crops with very high intensities; no crop with total GHG emission intensities exceeding 141 kg CO<sub>2</sub>e t<sup>-1</sup> achieved sugar beet yields greater than 35 t ha<sup>-1</sup>. Variation in the percentage of sugar beet GHG emission intensities related to fertiliser N application ranged from 1 to 69% of total GHG emissions, averaging at 29%.

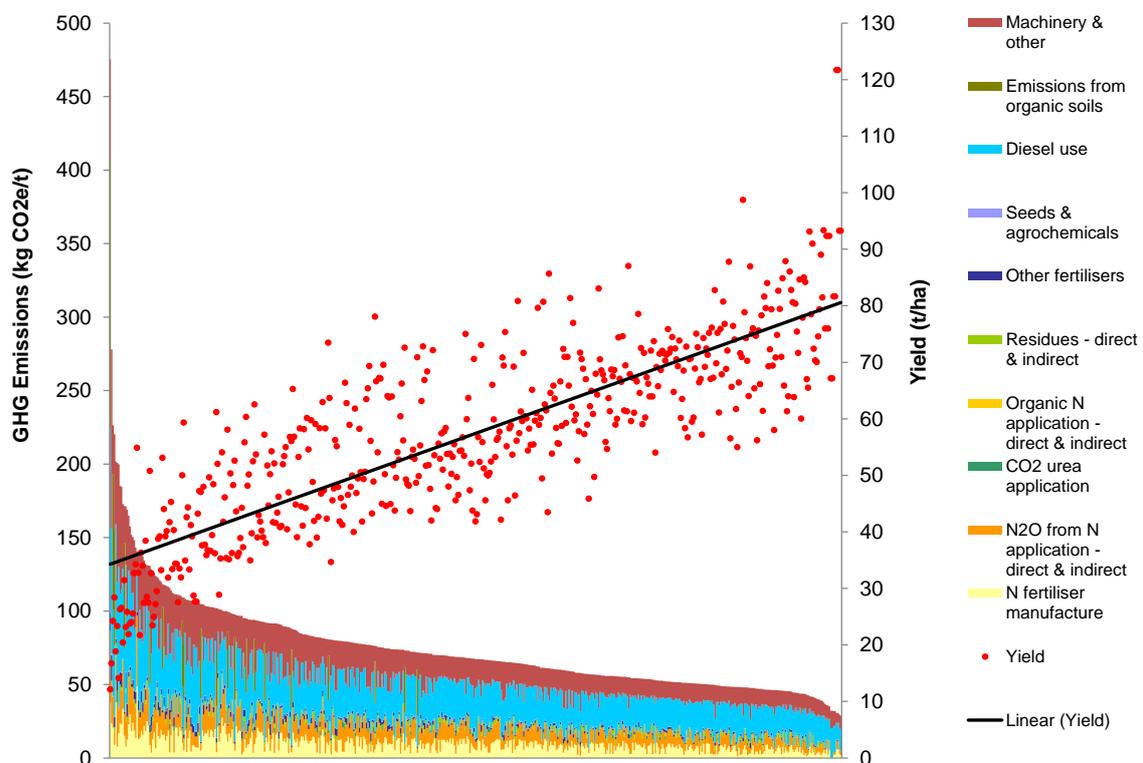
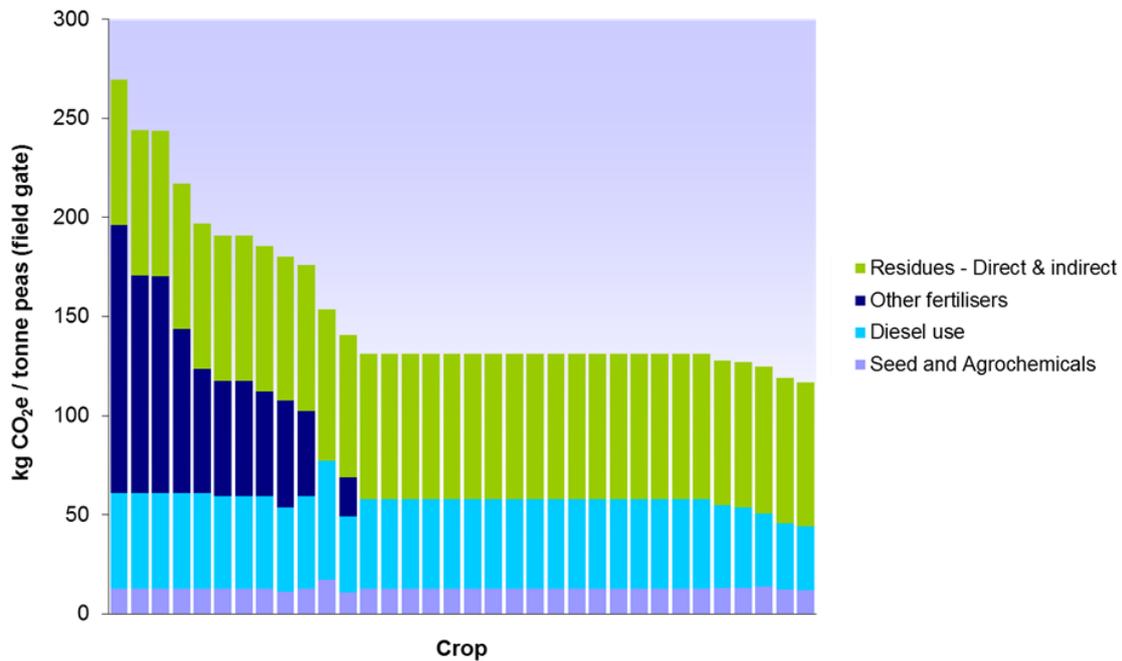


Figure 11. Variability of GHG emission intensities associated with sugar beet at farm gate for 511 datasets for 2011 and 2012 with the PAS 2050 methodology.

### Vining peas

There were far fewer datasets available for vining peas than for wheat, OSR or sugar beet, and the variation in total emission intensities was much less, as shown in Figure 12. Without fertiliser N applications there was less scope for big differences in total GHG emission intensities between individual crops. The apparent variation was almost entirely due to use of non-N fertilisers; these are likely to have rotational benefits so it could be argued that these should not be entirely attributed to the vining pea crop.



**Figure 12. Variability of GHG emission intensities ( $\text{kg CO}_2\text{e t}^{-1}$ ) associated with vining peas at farm gate for 34 datasets for 2008 and 2012 with the PAS 2050 methodology.**

Unless IPCC (1996) calculations are used, where emissions are deemed to come from N fixation by legumes, N-related GHG emissions from vining pea crops estimated by IPCC (2006) methodology come entirely from crop residues, averaging  $80 \text{ kg CO}_2\text{e t}^{-1}$  with very little variation (the only variation possible would relate to differences in yield or in residue removal).

#### 5.3.3. Results – Products

The impacts of farm-level variability on total GHG emission intensities of crop food products and biofuels have been calculated and presented in the full WP1 report. Given that these impacts are essentially the same as for the crops but scaled by the importance of the farm gate product to the overall GHG emission intensities of the retailed product or biofuel, details are not replicated here. However, some of the main outcomes are summarised in Table 14. Here it can be seen that farm-level variability can have large effects on the GHG emission intensities of these products. It should

be noted that relevant units are stated for all products apart from whisky for which results are quoted relative to the average.

**Table 14. Impacts of farm-level variability on total GHG emission intensities of crop products.**

Product	Units	Total GHG (CO <sub>2</sub> e) emission intensities				Percentage GHG emission intensities crop N-related			
		Mean	Min	Max	St Dev	Mean	Min	Max	St Dev
Chicken	kg <sup>-1</sup>	3.5	3	4.4		18%	70%	35%	
Bread	kg loaf <sup>-1</sup>	0.690	0.540	0.900	0.061	29%	16%	40%	4%
Cooking oil	kg litre <sup>-1</sup>	1.927	0.969	5.186	0.521	45%	22%	75%	6%
Sugar	kg kg <sup>-1</sup>	0.738	0.481	2.933	0.196	9%	0%	24%	3%
Frozen peas	kg kg <sup>-1</sup>	1.04	0.99	1.15					
Whisky	relative <sup>1</sup>	1	0.86	1.32	0.046	15%	5%	28%	3%
Bioethanol, wheat	kg litre <sup>-1</sup>	0.714	0.359	1.572	0.129	41%	22%	53%	3%
Biodiesel, OSR	kg litre <sup>-1</sup>	1.375	0.747	3.341	0.297	40%	20%	72%	6%
Bioethanol, beet	kg litre <sup>-1</sup>	0.704	0.472	2.744	0.177	10%	0%	26%	3%

<sup>1</sup>Estimates are quoted as a proportion of the mean value.

#### 5.3.4. Results – Effects of agronomic factors

For each crop, the results can be divided into different categories to assess how factors such as fertiliser type, soil type, residue incorporation and cultivation method impact on total GHG emissions, as summarised in Table 15 to Table 17. (Greater detail and implications for crop products are given in the full WP1 report.) Table 15 summarises total GHG emission intensities per tonne of produce for crops grown on different soil types. For consistency, the results presented were derived using the PAS2050 methodology. Whilst there are apparent differences in average values of total GHG emission intensities between soil types, these are inconclusive due to the degree of overlap indicated by relatively large standard deviations. Overall, total GHG emission intensities were highest for organic soils, although there were insufficient numbers of datasets to draw any statistically justifiable inferences. Soils classed as ‘organic’ have large associated GHG emissions from the mineralisation of soil carbon, calculated as 5 kg N<sub>2</sub>O-N ha<sup>-1</sup>, or 2,300 kg CO<sub>2</sub>e ha<sup>-1</sup>. Clearly this has a potentially significant impact on total GHG emission intensities per tonne of crop grown on organic soils despite the tendency for large reductions in fertiliser N requirements on these soils.

**Table 15. Field-level variability of total GHG emission intensities (kg CO<sub>2</sub>e t<sup>-1</sup>) of crops to soil type. SD = standard deviation; F= feed; B = bread.**

Soil type	Wheat (F)	Wheat (B)	OSR	Sugar beet
	<i>No. fields</i>			
Light	16	19	23	32
Medium	440	158	246	319
Heavy	140	108	84	140
Organic	1	2		20
	<i>Mean total GHG emission intensity (kg CO<sub>2</sub>e t<sup>-1</sup>)</i>			
Light	410	436	1023	67
Medium	360	408	877	76
Heavy	380	434	956	68
Organic	560	485		132
	<i>SD of total GHG emission intensity (kg CO<sub>2</sub>e t<sup>-1</sup>)</i>			
Light	120	85	231	20
Medium	82	92	324	31
Heavy	69	99	175	26
Organic	na	20		89

The automatic addition of these GHG emissions using IPCC 2006 methodology on the basis of soil classification is somewhat problematic around the boundaries; organic soils are classed as those with organic matter contents above 10%, so that soils with 9% organic matter are deemed to emit no CO<sub>2</sub> through mineralisation yet will tend to benefit from very low N fertiliser requirements due to mineralisation of soil-N in season. As such, neighbouring fields can produce crops with the highest GHG emissions (if classed as organic) or the lowest GHG emissions (if not classed as organic but if high levels of mineralisation result in low fertiliser N use). For non-organic soils, total GHG emission intensities associated with wheat and OSR crops were lowest on medium soils and highest on light soils, reflecting highest yields generally being recorded for medium soils and lowest yields on light soils. In contrast, the total GHG emission intensities associated with sugar beet were lowest for heavy soils and highest for medium soils. However, all these differences are marginal since they are well within the confidence limits (indicated by standard deviations).

Table 16 summarises total GHG emission intensities per tonne of crop with non-inversion tillage and ploughing. For consistency, the results presented were derived using the PAS2050 methodology. There were no statistically meaningful differences in total GHG emission intensities related to cultivation practices. Average GHG emission intensities were lower for non-inversion tillage for all crops which probably reflects lower diesel fuel consumption for cultivations. However, it should be noted that actual diesel use per ha was not commonly known by farmers, and it was necessary to derive these GHG emissions based on assumptions about the range of field operations conducted. Our calculations made no allowance for the exact implement used, soil type, working depth or conditions so must probably they obscured variability in actual GHG emissions.

**Table 16. Sensitivity of total GHG emission intensities (kg CO<sub>2e</sub> t<sup>-1</sup>) for crops to different cultivation practices.**

Crop	Non-inversion tillage			Ploughing		
	Count	Average	St Dev	Count	Average	St Dev
Feed wheat	202	355	73	362	361	86
Breadmaking wheat	191	410	85	90	430	103
OSR	190	893	318	163	924	261
Sugar beet	338	66	29	178	93	40

The effects of residue management practices – baling and removal, versus incorporation – on total GHG emission intensities per tonne of crops are shown in Table 17. For consistency, these results were derived using the PAS2050 methodology. For wheat and OSR, the incorporation of residues resulted in higher GHG emission intensities from crops than where straw was removed. In contrast, there was no significant difference between total GHG emission intensities for sugar beet with or without residue incorporation. It would be expected that total GHG emission intensities for any crop would be lower when residues were baled and removed since soil N<sub>2</sub>O emissions associated with incorporated residues would be avoided. The magnitude of this effect would depend on the N content of the residues and their relative yield. However, the differences in total GHG emission intensities indicated in Table 17 are not entirely due to soil N<sub>2</sub>O emissions because, by applying the PAS2050 methodology, it was necessary to allocate cultivation GHG emissions between the crop and any residues that were removed. Depending on the assumptions used in such allocations (i.e. based on relative economic value, energy content or mass) this could further magnify the effect on GHG intensities of different residue management practices.

**Table 17. Sensitivity of total GHG emission intensities (kg CO<sub>2e</sub> t<sup>-1</sup>) for crops to different residue management practices.**

Crop	Baled / Removed			Incorporated		
	Count	Average	St Dev	Count	Average	St Dev
Feed wheat	317	330	71	277	350	92
Breadmaking wheat	174	404	93	114	446	83
OSR	49	616	170	277	906	298
Sugar beet	299	75	39	21	76	31

## 5.4. Conclusions

Work Package 1 of the MIN-NO project set out to quantify the GHG emission intensities of a range of crops and crop-derived products in order to understand the importance of N<sub>2</sub>O and N fertiliser-related emissions in the context of other sources of variability and uncertainty.

It is clear that emissions associated with N fertiliser are very important to the total GHG emission intensities of fertilised arable crops such as wheat, OSR and sugar beet. Based on benchmark estimates, combined GHG emissions from N fertiliser manufacture, N<sub>2</sub>O soil emissions and N<sub>2</sub>O emissions from crop residues are responsible for 71% of total GHG emission intensities in feed wheat, 76% in breadmaking wheat, 76% in OSR and 49% in sugar beet. The emission intensities of vining peas, which receive no N fertiliser, are still 36% due to N<sub>2</sub>O emissions from crop residues.

These N-related emissions still make significant contributions to the total GHG intensities of the final products purchased by the consumer, especially for products with short supply chains and a dominant arable feedstock such as wheat for bread, rapeseed for cooking oil and beet for sugar. For these food products, benchmark estimates indicate that N-related emissions were 37%, 66% and 23% of total GHG emission intensities, respectively. For chicken, the importance of wheat's N related emissions within benchmark estimates of total GHG emission intensity was only 17%; this reflects that wheat is just one component of the diet; other N-related emissions would be associated with feeds derived from other crops. GHG emission intensities from frozen peas were dominated by energy use in the freezing process and storage, so that only 8% of the benchmark estimate of total GHG emission intensity was crop N-related. Similarly high energy costs in distilling meant that crop N-related emissions from whisky were only 14% of the benchmark estimate of total GHG emission intensity. For biofuels, in which total GHG emission intensities are of crucial importance from a regulatory perspective, N-related emissions formed the major portion of GHG intensities of bioethanol from wheat (65% for the benchmark estimate) and of biodiesel from OSR (62% for the benchmark estimate), with N related emissions from growing sugar beet being only 24% of the benchmark estimate for bioethanol. Regardless of the exact contributions to total GHG emission intensities, reductions in the emissions related to crop production would make meaningful reductions in the total GHG emission intensities of all products.

Recent improvements to N fertiliser manufacturing plants operated in the UK and the rest of Europe by CF Fertilisers UK, Yara and others have substantially reduced N<sub>2</sub>O losses from nitric acid production and, hence, have reduced GHG emissions of AN fertiliser available in the UK. Choosing to use 'abated' AN can decrease N manufacturing-related emissions by 40% from 6.3 to 3.4 kg CO<sub>2</sub>e kg<sup>-1</sup> N (using the GrowHow example in Table 2) which, in terms of the benchmark estimates, would reduce total crop GHG emission intensities by 17% for OSR, 14% or 15% for wheat, 7% for sugar beet, 14% for cooking oil, 7% for bread, 4% for sugar and whisky, and 3% for

chicken. Also, for biofuels, the benchmark estimates of total GHG emission intensities would be reduced by 13% for biodiesel from OSR, 12% for bioethanol from wheat and 4% for bioethanol from sugar beet.

The uncertainties in the estimation of N<sub>2</sub>O emissions from soil are large; in particular, the extremes of the uncertainty range exceeded the variability in total GHG emission intensities at farm level. It is quite likely possible that accurate emissions associated with UK arable crops and residues could be half the currently assumed levels (though it is also feasible that they could be double). The possibility of lower N<sub>2</sub>O EFs, as reflected in the PP-scenario, could reduce benchmark estimates of total GHG emission intensities by 20% for OSR, between 16 and 18% for wheat, 12% for sugar beet, 16% from cooking oil, 8% from bread, 4% from chicken, sugar and whisky, 15% for biodiesel from OSR, 14% for bioethanol from wheat and 4% for bioethanol from sugar beet.

Sensitivity analysis of total GHG emissions for crops and products using quite extensive farm-level datasets available to the MIN-NO project showed considerable variability. At the extremes, total GHG emissions of produce showed at least a 3-fold range and total GHG emission intensities of crop products showed a 1.5- to 2-fold range. Many factors appeared to influence this variability although it was not possible to distinguish these conclusively. Analysis of the main factors, including soil type, cultivation practices and residue management practices provided hints of underlying causes, but larger and more representative datasets would be needed to form statistically proven outcomes. Despite this, the analysis of crop datasets has added considerably to the understanding of factors affecting GHG emission intensities for these crops and their products. Previously, it was only possible to use national or regional averages of values in GHG emissions calculations. Now, given good examples of the type of data that form such averages, indications of variability in these values have emerged, enabling these to be assessed statistically with frequency distributions and standard deviations (two standard deviations covering ~60% of all values in each population).

Analysis of the crop datasets has demonstrated that, despite very large differences between extremes, most crops had total GHG emission intensities relatively close to the derived averages – the frequency distributions in many instances being quite peaked and fairly symmetrical (Figure 6, Figure 7, Figure 10, Figure 11). Standard deviations relative to average values of total GHG emission intensity for each of these crops were (depending on calculation methodology) between 22% and 23% for wheat, 28% to 32% for OSR and 47% to 50% for sugar beet. For food products, relative standard deviations in total GHG emission intensity were 4% for chicken, 9% for bread, 27% for cooking oil, 27% for sugar and 5% for whisky. For biofuels, relative standard deviations in total GHG emission intensity were 18% for bioethanol from wheat, 22% for biodiesel from OSR and

25% for bioethanol from sugar beet. Note that there were too few crops to estimate relative standard deviations for vining peas.

Quantification of variability in total GHG emission intensities for crops and their products is important to inform the interpretation of GHG intensity estimates in future, and in supporting their subsequent application in reporting and regulatory compliance. Furthermore, knowledge of variability between farms should inform participants in the supply chain about the possible role of feedstock selection in reducing the total GHG emission intensities of their products: seeking out suppliers who can provide crops with low GHG emission intensities and avoiding crops with high GHG emission intensities. It may be that such a practice would not impact materially on actual GHG emissions at a broader scale; this would depend on whether the pursuit of low GHG emission intensities was compatible with economic performance and whether practices employed by the 'best' farmers could be transferred easily to others, so that reduced GHG emission intensities could be replicated on an increasing proportion of farm-land. Leaving aside these and other practicalities, it is apparent from this work that, given adequate incentives, there could be considerable scope to reduce the GHG emission intensities of UK crop products through the selective choice of N fertilisers, and more modest scope through use of mitigation options and choice of supply chains. Such options will be considered further in Section 10.

## 6. Fertiliser Experiments – WP2A

### 6.1. Rationale and objectives

The objective of WP2A was to quantify by field experiment *direct* soil N<sub>2</sub>O emissions associated with UK arable cropping and to express N<sub>2</sub>O emissions in terms of both land area (kg ha<sup>-1</sup>) and emission intensity (kg t<sup>-1</sup> of harvested produce). One of the principal uncertainties being addressed in this section is the relationship between N fertiliser addition and N<sub>2</sub>O emissions. As we have seen in Section 5 IPCC reporting procedures assume (and therefore it is implicit in national inventory reports and carbon foot-printing calculations) that emissions are linearly related to N inputs at 0.01 kg N<sub>2</sub>O-N per kg N applied (IPCC, 2006). Experiments were designed in this project that would rigorously test this assumption. The dominant factors affecting emissions on a land area basis were expected to be N fertiliser amounts, soil type and rainfall. The key additional factor affecting intensities was expected to be the level of crop production, which in turn was expected to be affected by crop species, season of establishment (autumn or spring), regional climate and soil. Thus the experiments described here tested wide-ranging rates of fertiliser N on the main UK arable crops that receive significant quantities of N fertiliser (winter wheat, winter barley, spring barley, sugar beet and winter OSR), over three seasons (which happened to show extreme differences), in the contrasting regions of England and Scotland, with contrasting soil types. The experiments included intensive measurements of N<sub>2</sub>O emissions, along with measures of crop growth and yield, enabling N<sub>2</sub>O emissions to be analysed and interpreted on a land area or tonnage basis.

### 6.2. Sites and seasons

Twenty-four experiments were carried out over 3 harvest years (eight in each of 2010, 2011 and 2012) on commercial farms in three arable areas of England and on a research farm in Scotland. The areas were chosen to have generally contrasting soil textures (Table 18). Sites chosen within each area were on broadly similar but not identical soils. Area 1 was on chalky boulder clay soils around Boxworth in eastern England, Area 2 was on sandy soils around Gleadthorpe in central England, Area 3 was generally on alluvial silty clay loams around Terrington in eastern England and the glacial till soils in Area 4 at the Bush Estate in central Scotland ranged from sandy clay loam to clay. All sites were chosen to have a soil mineral N status of <80 kg N ha<sup>-1</sup> to 90 cm depth in spring (Table 18); no manure had been applied to any site in the previous two years. Further site details are shown in Table 18.

Table 18. Details of soils, cropping and N application rates at sites used for fertiliser experiments in WP2A.

Area number and name	Soil type (Soil series)	Background SMN (0-90 cm) kg N ha <sup>-1</sup>	Crop (variety), year N applied	Drilling date	Harvest date	Previous crop (harvest year)	Follow on crop (harvest year)	Recommended N application rate (kg N ha <sup>-1</sup> )
<b>1. Boxworth, East England</b>	Clay loam (Hanslope)	40	<b>Winter wheat</b> (Robigus) 2010	25 Sep 09	01 Sep 10	Winter OSR (2009)	Winter wheat (2011)	250
	Clay (Hanslope)	30	<b>Winter OSR</b> (Castille) 2010	27 Aug 09	28 Jul 10	Winter wheat (2009)	Winter wheat (2011)	220
	Clay loam (Hanslope)	35	<b>Winter wheat</b> (Invicta) 2011	18 Oct 10	17 Aug 11	Winter wheat (2010)	Winter beans (2012)	250
	Clay (Hanslope)	30	<b>Winter OSR</b> (Catana) 2011	25 Aug 10	28 Jul 11	Winter wheat (2010)	Winter wheat (2012)	220
	Clay (Hanslope)	40	<b>Winter wheat</b> (Solstice) 2012	22 Sep 11	21 Aug 12	Winter OSR (2011)	Winter wheat (2013)	250
	<b>2. Gleadthorpe, Central England</b>	Sandy loam (Newport)	20	<b>Spring barley</b> (Quench) 2010	11 Mar 10	09 Aug 10	Leek (2009)	Stubble
Sandy loam (Newport)		20	<b>Sugar Beet</b> (Bullfinch) 2010	07 Apr 10	17 Jan 11	Winter wheat (2009)	NA	120
Sandy loam (Wick)		15	<b>Winter barley</b> (Saffron) 2011	21 Sep 10	22 Jul 11	Winter wheat (2010)	Winter wheat (2012)	200
Sandy loam (Wick)		25	<b>Winter wheat</b> (Alchemy) 2012	10 Nov 11	14 Sep 12	Winter barley (2011)	NA	180
Sand (Newport)		20	<b>Winter OSR</b> (Ovation) 2012	24 Aug 11	10 Aug 12	Winter wheat (2011)	Winter wheat (2013)	190

Area number and name	Soil type (Soil series)	Background SMN (0-90 cm) kg N ha <sup>-1</sup>	Crop (variety), year N applied	Drilling date	Harvest date	Previous crop (harvest year)	Follow on crop (harvest year)	Recommended N application rate (kg N ha <sup>-1</sup> )
<b>3. Terrington, East England</b>	Silty clay loam (Wisbech)	45	<b>Winter wheat</b> (Duxford) 2010	09 Oct 09	03 Sep 10	Winter wheat (2009)	Winter wheat (2011)	220
	Sandy loam (Blacktoft)	65	<b>Winter wheat</b> (Solstice) 2011	09 Oct 10	17 Aug 11	Winter OSR (2010)	Winter wheat (2012)	230
	Silty clay loam (Wisbech)	20	<b>Sugar beet</b> (Opta) 2011	27 Mar 11	26 Oct 11	Winter wheat (2010)	Winter wheat (2012)	120
	Clay loam (Agney)	40	<b>Winter barley</b> (Cassata) 2012	10 Oct 11	09 Aug 12	Winter wheat (2011)	Winter wheat (2013)	200
	Clay loam (Agney)	50	<b>Sugar beet</b> (Columbus) 2012	27 Mar 12	26 Oct 12	Winter wheat (2011)	Winter wheat (2013)	120
<b>4. Bush Estate, Central Scotland</b>	Sandy clay loam (Macmerry)	70	<b>Winter barley</b> (Saffron) 2010	24 Aug 09	05 Aug 10	Spring barley (2009)	Stubble	175
	Sandy clay loam (Macmerry)	20	<b>Winter wheat</b> (Solstice) 2010	07 Oct 09	03 Sep 10	Spring barley (2009)	Stubble	185
	Sandy clay loam (Macmerry)	15	<b>Spring barley</b> (Westminster) 2010	25 Mar 10	16 Sep 10	Spring barley (2009)	Stubble	120
	Clay loam (Macmerry)	35	<b>Winter OSR</b> (Sequoia) 2011	26 Aug 10	16 Aug 11	Spring barley (2010)	Stubble	220
	Clay loam (Macmerry)	50	<b>Winter wheat</b> (Cordial) 2011	28 Sep 10	05 Sep 11	Spring barley (2010)	Stubble	200
	Clay loam (Macmerry)	15	<b>Spring barley</b> (Westminster) 2011	21 Mar 11	14 Sep 11	Spring barley (2010)	Stubble	120
	Clay (Macmerry)	45	<b>Winter OSR</b> (Sequoia) 2012	31 Aug 11	05 Sep 12	Spring barley (2011)	Stubble	220
	Clay (Macmerry)	30	<b>Winter wheat</b> (Grafton) 2012	04 Oct 11	14 Sep 12	Spring barley (2011)	Stubble	220
	Clay (Macmerry)	30	<b>Spring barley</b> (Westminster) 2012	31 Mar 12	30 Aug 12	Spring barley (2011)	Stubble	120

### 6.3. Experimental design

*Direct* N<sub>2</sub>O fluxes, crop yields and crop N uptake were measured following spring ammonium nitrate (AN; 34.5% N) fertiliser applications to growing arable crops, namely winter wheat, winter OSR, winter barley, spring barley and sugar beet (Table 18). Each crop type was studied each year at one of the four locations. The experiments were located in commercially grown crops in England and were drilled as required for the size of the experiment in Scotland. The five fertiliser treatments were replicated (x3) and arranged in a randomised block design of 15 plots (each 6 x 24 m). The fertiliser was applied by hand at 5 different N application rates spanning the likely economic optimum (N<sub>opt</sub>); 0 kg N ha<sup>-1</sup> (i.e. untreated control), 40%, 80%, 120% and 160% of the recommended N rate based on guidelines in 'The Fertiliser Recommendations Manual (RB209)' (Anon., 2000; Anon., 2010) for the English sites and in 'Nitrogen recommendations for cereals, OSR and potatoes' (TN625) (Sinclair *et al.*, 2009) for Scotland. For spring barley, in order to ensure the crop N optima were reached, N application rates were 0 kg N ha<sup>-1</sup>, 60%, 120%, 180% and 240% of the recommended N rate. In the text, the treatments will be referred to simply as the 0N treatment, the 60% treatment, the 120% treatment etc. Following recommended practice (Anon., 2000; Anon., 2010; Sinclair *et al.*, 2009) and depending on the crop type, up to three separate fertiliser applications were made in order to reach the targeted application rates. So that the experiments were as commercially relevant to the UK agricultural industry as possible, the amount of N fertiliser applied at each application timing also followed recommended practice for that rate and that crop (Anon., 2000; Anon., 2010) such that across the four fertilised treatments, at each split, the N applied was seldom linearly proportional to the final application rates. To prevent any nutrient deficiencies, overall basal phosphorus, potassium, magnesium and sulphur were applied according to site requirements following soil analysis, and applications of agro-chemicals were made as required to control weeds, pests, diseases and lodging. Following crop harvest, cereal straw was chopped and returned to the soil, or baled and removed, reflecting the commercial practice in each area. Cropping practices after harvest were typical of the crop rotation in each area (Table 18).

### 6.4. Materials and methods

#### 6.4.1. Direct nitrous oxide-N emission measurements

In order to maintain an undisturbed area for crop sampling, *direct* N<sub>2</sub>O emissions were measured from one half (3x24 m) of each plot, reserving the other half for crop measures. Five static flux chambers (40 cm wide x 40 cm long x 25 cm high) covering a total surface area of 0.8 m<sup>2</sup> were regularly positioned in the N<sub>2</sub>O measurement area. The chambers were of white (i.e. reflective) PVC and un-vented with a water-filled channel running around the upper rim of the chamber allowing an air-tight seal to form following chamber enclosure with a lid (Cardenas *et al.*, 2010; Dobbie and Smith, 2003; Jones *et al.*, 2005; Smith *et al.*, 2012). Chambers were pushed into the

soil up to a depth of 5 cm and remained in place throughout the experiment, except during N fertiliser applications, drilling and harvesting when chambers were removed, locations were marked, and chambers were re-instated to the same position as prior to removal.

Chambers remained open except for a short time on each sampling day. On that day, ten samples of ambient air were taken to represent time zero (T0) N<sub>2</sub>O samples. From each chamber, after a 40-minute enclosure period (T40) a headspace sample was taken using a 50-ml syringe. Using a double needle system the sample was flushed through a pre-evacuated 20-22 ml glass vial fitted with a chloro-butyl rubber septum and held at atmospheric pressure. The N<sub>2</sub>O flux was calculated using an assumed linear increase in N<sub>2</sub>O concentration from the ambient N<sub>2</sub>O concentration (T0) to the N<sub>2</sub>O concentration inside the chamber after 40-minutes enclosure (T40) (Cardenas *et al.*, 2010; Chadwick *et al.*, 2014; Smith *et al.*, 2012). Throughout each experiment, the linearity of emissions through time was checked routinely from three chambers located on the highest N rate plots. A minimum of five samples were taken from each chamber at 15 min intervals commencing at closure i.e. T0 and spanning the T40 sampling time. Previous field experiments in the UK have demonstrated a linear increase in N<sub>2</sub>O concentration within a chamber and this was supported by the results from a desk study which reviewed UK experimental data where the increase in N<sub>2</sub>O concentration within a field chamber had been measured during enclosure (Chadwick *et al.*, 2014). More than 90% of the 1,970 measurements of chamber headspace accumulation demonstrated a linear increase in N<sub>2</sub>O concentration (Chadwick *et al.*, 2014).

In order to minimise the effect of diurnal variation, gas sampling was carried out between 10:00 am and 14:00 pm and where possible between 10:00 am and 12:00 pm as suggested by IAEA (1992) and referred to in the IPCC good practice guidance (IPCC, 2000). Gas samples were analysed as soon as possible after collection (to minimise potential leakage) using gas chromatographs fitted with an electron-capture detector and an automated sample injection system. Following receipt in the laboratory, three replicates of one standard N<sub>2</sub>O gas were kept with the samples and were used to verify sample integrity during storage. The gas chromatographs were calibrated on a daily basis using certified N<sub>2</sub>O standard gas mixtures. The gas chromatographs used in this project were also involved in inter-laboratory ring tests to ensure the comparability of data reported by different labs. In order to permit sampling from a growing crop, at the time of sampling an additional chamber was stacked (using the water-filled channel) onto each permanent chamber and the chamber enclosure period extended.

The N<sub>2</sub>O sampling strategy was weighted so approximately 50% of the measurements were targeted during the period where previous UK studies have shown that the greatest emissions were likely to occur i.e. 0-6 weeks after N fertiliser application (Chadwick *et al.*, 2000; Dobbie and Smith, 2003; Smith *et al.*, 2012) and following IPCC good practice guidance to make more

intensive measurements after fertiliser application (IPCC, 2000). Following each N fertiliser application, N<sub>2</sub>O flux measurements were carried out on four consecutive days after the onset of fertiliser granule degradation. Measurements were taken on three days in the subsequent week, twice weekly for the next fortnight and thereafter decreasing in frequency to monthly sampling towards the end of the 12 month sampling period. Prior to the first fertiliser application N<sub>2</sub>O measurements were taken to provide baseline information. Additional measurements were taken after the main post-harvest soil cultivation and, where possible, after the first rain after cultivation. This sampling schedule resulted in an annual total of 40-50 sampling days starting from the day of the first fertiliser application. Measurements were taken over 12 months to follow IPCC good practice guidance (IPCC, 2000) and so that the results were directly comparable to the IPCC default EF (IPCC, 2006).

#### **6.4.2. Statistical analysis of N<sub>2</sub>O emissions**

N<sub>2</sub>O fluxes from the five replicate chambers per plot were averaged (arithmetic mean) and the annual cumulative N<sub>2</sub>O emission from each treatment was estimated by interpolating fluxes linearly between sampling points and integrating the daily emissions over 12 months. N<sub>2</sub>O EFs for direct soil emissions (EF<sub>1</sub>) were calculated by subtracting the mean integrated annual flux from the control plots; they were expressed as the percentage of total-N applied as fertiliser.

Yield-scaled N<sub>2</sub>O emission intensity values (kg N<sub>2</sub>O-N t<sup>-1</sup>) were calculated by dividing the annual cumulative N<sub>2</sub>O emission (kg N<sub>2</sub>O-N ha<sup>-1</sup>) by the harvested crop yield (t ha<sup>-1</sup>).

Responses of top soil mineral-N (SMN) and N<sub>2</sub>O emissions with N application rate were analysed by curve fitting using GENSTAT version 12 (Lawes Agricultural Trust, 2009). For each experiment, linear and quadratic functions were fitted against the five rates of fertiliser N for:

1. daily top soil SMN on every measurement occasion
2. daily N<sub>2</sub>O emissions (kg N<sub>2</sub>O-N ha<sup>-1</sup> d<sup>-1</sup>) on every measurement occasion
3. cumulative N<sub>2</sub>O emissions (kg N<sub>2</sub>O-N ha<sup>-1</sup>) on every measurement occasion
4. annual cumulative N<sub>2</sub>O emissions (kg N<sub>2</sub>O-N ha<sup>-1</sup>)
5. annual yield-scaled N<sub>2</sub>O emissions (kg N<sub>2</sub>O-N t<sup>-1</sup>).

The function which best explained the data was deemed to be the fit which accounted for most of the variance. If the variance explained differed by <0.05%, choice of fit was classed as marginal i.e. both linear and quadratic equally well explained the data. It was taken that there was no significant fit if  $P > 0.05$ .

Cross site analysis of the annual cumulative N<sub>2</sub>O-N emissions (kg ha<sup>-1</sup>) for the 0N and 120% treatment and for the EF (120% treatment; % total-N applied) was performed in Genstat, taking

account of the unbalanced design, to examine effects of harvest year, crop and region (Scotland, England). Prior to the analysis, absolute emissions ( $\text{kg ha}^{-1}$ ) were log transformed and the resultant mean values back transformed. Data from the OSR crop in 2012 at site 4 were deemed to be aberrant and so were excluded from the cross-site analysis.

#### **6.4.3. Soil and meteorological measurements**

To assess effects of soil temperature and soil moisture on  $\text{N}_2\text{O}$  emissions, soil temperature (at a depth of 5 cm) was logged continuously on-site, whilst air temperature and daily rainfall were recorded at a meteorological station within 7.5 km. On every  $\text{N}_2\text{O}$  measurement occasion, representative soil samples were taken (0-10 cm) from each plot for the determination of gravimetric moisture content and soil mineral N (ammonium-N and nitrate-N) content. Following collection, soil samples were kept cool (target  $<4^\circ\text{C}$ ) and analysed fresh within 24 hours of receipt at the laboratory. The concentrations of nitrate-N and ammonium-N were measured by extraction of 20 g of moist soil in 100 ml of 1M KCl shaken for 1 hour and analysed using a continuous flow system (Timberline TL-2800 Ammonia Analyzer, and Skalar SAN++) based on the principle of gas diffusion across a membrane sensed by electrical conductivity (Carlson, 1978) or colorimetric determination of ammonium-N. The initial procedure measured the ammonium concentration and a zinc or cadmium cartridge was then used to reduce nitrate to ammonium, allowing measurement of the total inorganic N concentration (Carlson, 1986). The nitrate concentration was then calculated as the difference between total inorganic N and ammonium N.

Topsoil dry bulk densities (0-5.5 cm) were measured on the 0N and the 80% treatments (6 samples in total) following the first N fertiliser application, following harvest and after primary and secondary autumn soil cultivations. The soil bulk density was used to convert the soil gravimetric moisture content to water-filled pore space (WFPS). At the start of the experiment and prior to the first fertiliser application, soil pH, extractable P, K and Mg concentrations, particle size distribution and total N and carbon levels were measured from across each block (0-15 cm sampling depth), (Table 19). Background soil mineral N was also measured on a block basis at the start of the experiment (0-90 cm sampling depths) (Table 18).

Table 19. Mean (n=3) soil characteristics (0-15 cm depth, before N application except dry bulk density) at all experimental sites. P = phosphorus (England: Olsen's; Scotland: Morgan's method), K = potassium, Mg = magnesium, BD = dry bulk density (when 1<sup>st</sup> N fertiliser applied).

Area number & name	Crop, year N applied	Clay (%)	pH	P (mg kg <sup>-1</sup> ; index)	K (mg kg <sup>-1</sup> ; index)	Mg (mg kg <sup>-1</sup> ; index)	Total N (%)	Total C (%)	BD (g cm <sup>-3</sup> )
<b>1. Boxworth</b>	Winter wheat, 2010	34	7.6	21 / 2	210 / 2+	108 / 3	0.19	2.0	1.29
	Winter OSR, 2010	41	8.2	11 / 1	236 / 2+	65 / 2	0.21	3.2	1.21
	Winter wheat, 2011	32	7.5	27 / 3	174 / 2-	122 / 3	0.14	1.6	1.43
	Winter OSR, 2011	47	7.8	28 / 3	253 / 3	117 / 3	0.25	2.7	1.13
	Winter wheat, 2012	49	8.2	16 / 2	277 / 3	97 / 2	0.24	2.7	1.08
<b>2. Gleadthorpe</b>	Spring barley, 2010	11	7.3	36 / 3	196 / 2+	102 / 3	0.08	0.9	1.40
	Sugar Beet, 2010	11	6.8	28 / 3	109 / 1	105 / 3	0.10	1.2	1.12
	Winter barley, 2011	10	6.3	37 / 3	96 / 1	132 / 3	0.13	1.5	1.33
	Winter wheat, 2012	11	6.0	47 / 4	118 / 1	111 / 3	0.13	1.6	1.33
	Winter OSR, 2012	3	6.6	76 / 5	164 / 2-	60 / 2	0.10	1.1	1.24
<b>3. Terrington</b>	Winter wheat, 2010	35	8.3	12 / 1	221 / 2+	113 / 3	0.16	2.7	1.26
	Winter wheat, 2011	11	8.3	17 / 2	147 / 2-	43 / 1	0.09	1.8	1.34
	Sugar beet, 2011	29	7.8	34 / 3	408 / 4	342 / 5	0.18	2.0	1.18
	Winter barley, 2012	22	8.1	23 / 2	217 / 2+	138 / 3	0.14	1.7	1.41
	Sugar beet, 2012	20	8.1	18 / 2	154 / 2-	191 / 4	0.12	1.6	1.30
<b>4. Bush Estate, Central Scotland</b>	Winter barley, 2010	36	6.2	11	122	269	0.14	4.0	1.23
	Winter wheat, 2010	36	6.2	11	122	269	0.14	4.0	1.26
	Spring barley, 2010	36	6.2	11	122	269	0.14	4.0	1.27
	Winter OSR 2011	34	6.7	12	95	235	0.16	4.9	1.26
	Winter wheat, 2011	34	6.7	12	161	272	0.15	4.9	1.26
	Spring barley, 2011	34	6.7	12	118	200	0.14	4.9	1.25
	Winter OSR, 2012	37	5.9	11	153	237	0.17	2.2	1.26
	Winter wheat, 2012	37	5.7	11	153	237	0.17	2.2	1.26
	Spring barley, 2012	37	5.9	11	153	237	0.17	1.9	1.26

#### **6.4.4. Crop yield and N recovery**

Whole crop samples were taken at intervals during the growing season and pre-harvest in order to measure crop N uptake. Samples were taken from the other half of the plot from the N<sub>2</sub>O measurements and from three treatments only; 0N, 80% and 160% of the recommended N rate.

For cereals, on each sampling occasion the crop was cut as close to the ground as possible within two quadrat areas per plot (each quadrat had an area of 0.5 m<sup>2</sup>) and bulked together. The crop fresh weight was recorded and a sub-sample taken for dry matter determination and for laboratory analysis of total N in plant tissue. Prior to drying at 80°C for 16-24 hours the fresh weight of the sub-sample was noted, as was the subsequent sub-sample dry weight. For OSR, four quadrat areas per plot were used to gather an appropriate amount of plant tissue. Total N analysis of plant material was performed using the Dumas combustion method with a Leco FP-228 N determinator (Leco Corporation, St. Joseph, Michigan) and Thermo Scientific Flash 2000 (Cambridge).

N uptake by sugar beet was assessed by removing whole plants from 4 x 1 m lengths of the crop row on each sampling occasion. The sugar beet shoots were treated in the same way as the other whole crop samples; sugar beet roots were washed, a total fresh weight recorded and sub-sampled by selection of 5 typical beet. From each beet a core was taken to give a fresh sample of approximately 1 kg and the dry matter was determined following drying and the sample analysed for total N.

#### **6.4.5. Crop N uptake at harvest**

Samples for total crop N uptake were taken from each plot at harvest. For cereals, ten sampling points were selected randomly in each half-plot (not used for N<sub>2</sub>O sampling), which were >0.5 m inside the edge of the plot and avoided wheelings and bare soil patches. At each sampling point 10 adjacent shoots were cut at ground level. Shoots from all 10 sampling points were then bulked and, in the laboratory, the ears were cut at the collar and fresh weights of ear and straw recorded. The ears and a subsample of the straw were dried in an oven at 80°C for 24 hours and the dry weight recorded. Following drying, the ears were threshed to separate the grain and chaff and weights noted. The grain was re-dried at 100°C overnight and this dry weight recorded. After adding a portion of the chaff, the straw sub-sample was submitted for total N analysis of non-grain.

For OSR, pre-harvest samples were taken at about the time when the crops would normally have been swathed/desiccated, typically 10-14 days before harvest. Before sampling, the number of plants that would constitute a 20% sub-sample from a 1 m<sup>2</sup> area was determined; at 4 representative points across the trial area the numbers of plants in 1 m<sup>2</sup> were counted and 20% of this number of plants was sampled at each point. This procedure was repeated in each half-plot, cutting stems at ground level with care so as not to lose seeds, and bulking material from all four

points. The fresh weight of the whole sample was recorded. The plants were divided into stems and pods, dried for 24 hours at 80°C or until there was no further weight loss and dry weights recorded. The seed was threshed out and seed and pod wall dry weights recorded. Samples of the pod wall and stems were mixed and were submitted with samples of seed for analysis of total N in plant tissue. Following harvest an assessment of plants (stubble) per m<sup>2</sup> was carried out using 4 x 0.5 m<sup>2</sup> quadrats per plot.

For sugar beet, twenty roots plus tops were selected at random and their fresh weights were recorded in the field. After recording fresh weights, dry matter was determined on sub-samples of the tops (and related fallen leaves) and a sample was submitted for analysis of total N. The roots were washed, their total fresh weight recorded and they were then sub-sampled. Five typical beets were selected, quartered and a sliver or a core was taken from each beet to give a fresh sample of approximately 1 kg. Dry matter was determined following drying and a sample was submitted for analysis of total N in plant tissue and for sugar content.

#### **6.4.6. Harvested yield**

For cereals and OSR, the yield at harvest was determined using a plot combine harvester following an assessment of lodging. Grain yield was corrected to 85% dry matter (DM) for cereals and 91% DM for OSR. A representative grain sub-sample was taken either using the automated sampler fitted to the combine, or by taking a representative sample from the grain stream coming out of the combine auger. The grain sample dry matter was recorded and the sample submitted for analysis by near-infrared spectroscopy (NIRS) of total N%; protein was expressed as a percentage in grain dry matter using a conversion factor (N x 5.7; MAFF, 1986).

Prior to harvest the plant population was determined on sugar beet plots. The row spacing was measured and the numbers of plants in 20 m lengths of row were counted and recorded, repeating four times. The plant population was calculated as:

$$\text{Plant population} = 10000/\text{row spacing (m)} \times \text{number of plants in 1 m row} \quad \text{Equation 1}$$

Plots were then harvested by hand lifting 20 rows x 1.5 m (or 3 rows x 9 m, dependant on the direction of rows in relation to plots). The beets were topped, roots were counted and bagged, and sent for analysis by British Sugar for dirty weight, clean weight, sugar content, amino-N content, and P content and K content. Mean fresh beet yield, and mean sugar yield in t ha<sup>-1</sup> were calculated by multiplying plant population by mean root weight. Beet yields were adjusted for sugar content by British Sugar's agreed formula, as follows:

$$\text{Adjusted yield} = \text{Clean Root Yield} \times (2.017 - (4.3032 \times (0.91311^{\text{sugar\%}}))) \quad \text{Equation 2}$$

#### 6.4.7. Statistical analysis of harvest data

##### ***N surplus and apparent fertiliser N recovery***

Within an individual experiment, N surplus and apparent fertiliser N recovery were calculated on a treatment basis as follows:

$$\text{N surplus (kg ha}^{-1}\text{)} = \text{Crop N uptake (kg ha}^{-1}\text{)} - \text{Fertiliser N applied (kg ha}^{-1}\text{)} \quad \text{Equation 3}$$

$$\text{Apparent fertiliser N recovery (\%)} = \frac{[\text{Crop N uptake (treatment)} - \text{Crop N uptake (control)}]}{\text{Fertiliser N applied (kg ha}^{-1}\text{)}} \times 100 \quad \text{Equation 4}$$

##### ***Grain yield response curves and deriving economic optimum N rates ( $N_{opt}$ )***

The method adopted to fit yield response curves and to derive  $N_{opt}$  values was much the same as that used to develop the 'The Fertiliser Manual (RB209)' (Anon, 2000; Anon, 2010). Several mathematical functions describe the tendency for diminishing responses of crop yield to increasing fertiliser N (e.g. bilinear, quadratic, linear over quadratic, exponential, linear plus exponential). Most of these provide good descriptions of most experimental datasets, though for individual experiments one function may fit better than another. When analysing multiple experiments it is advantageous for one function to be chosen for all data, so that comparisons can be made between experiments on a common basis. Recommendations in 'The Fertiliser Manual (RB209)' are based on the linear plus exponential (LpE) function for cereal and oilseed crops (George, 1984), and a bilinear approach for sugar beet (Jaggard *et al.*, 2009). However, to ensure comparability between crops, the LpE function below has been fitted to data from all the experiments reported here.

$$y = a + b.r^N + c.N \quad \text{Equation 5}$$

where  $y$  is yield (in t ha<sup>-1</sup> at 85% dry matter for cereals, 91% dry matter for OSR and in t ha<sup>-1</sup> of sugar for sugar beet),  $N$  is total fertiliser N applied in kg ha<sup>-1</sup>, and  $a$ ,  $b$ ,  $c$  and  $r$  are parameters determined by statistical fitting. For those sites where the value of  $r$  was outside the range 0.8299-0.9999, the curve was refitted with  $r$  fixed at 0.99.

$N_{opt}$  values were calculated [ $N_{opt} = (\ln(k-c) - \ln(b \ln(r))) / \ln(r)$ ] where  $k$  is the breakeven ratio (BER) of crop value to N cost, but expressed in tonnes kg<sup>-1</sup> N rather than kg kg<sup>-1</sup> as with BER quoted in 'The Fertiliser Manual (RB209)' (Anon, 2000; Anon, 2010). For consistency, the BER values used to estimate optima here were taken from 'The Fertiliser Manual (RB209)' i.e. 5:1 kg:kg for cereals and 2.5:1 kg:kg for OSR (Anon, 2000; Anon, 2010). The BER for sugar beet was 27:1 kg:kg and was

set as the mid-point of the range created by recent fluctuations in prices of fertiliser and beet. Yields at  $N_{opt}$  ( $Y_{opt}$ ) were also calculated.

### **Grain N (or protein) response curves**

Grain protein concentrations can be calculated by multiplying grain N concentrations by 5.7. However, results here are presented as grain N. A response curve was fitted independently to each set of N concentration data for each site. Either a Normal Type curve with Depletion (NTD) or a straight line (SL) function was used, depending on which fit had the smaller Residual Mean Squares (RMS). The function for the NTD curve is:-

$$y = d + c.exp(-exp(-a.(N - b))) \quad \text{Equation 6}$$

where  $y$  is grain N (%),  $a$ ,  $b$ ,  $c$  and  $d$  are fitted parameters determined by fitting, and  $N$  is applied N ( $\text{kg ha}^{-1}$ ).

The function for the SL function is:-

$$N\% = a + b.N \quad \text{Equation 7}$$

Grain N% values were derived for the N rates that proved economically optimal.

## **6.5. Results**

### **6.5.1. Weather**

The annual rainfall and the average annual temperature for the three years for each area are shown alongside their long-term averages (LTA; 30-year mean 1980-2009) in Table 20. In 2010, 2011 and 2012, the annual rainfall from the English areas was respectively c.90%, c.65% and c.135% of LTA. In contrast, Scottish area's rainfall was c.80%, c.135% and c.180% of LTA for 2010, 2011 and 2012 respectively.

In both 2010 and 2011, in all three English areas, the annual rainfall was considerably less than the LTA with especially low rainfall in spring and early summer (Table 20), during and shortly after the time when the main fertiliser applications were made. April rainfall was c.35% and <10% of the LTA in 2010 and in 2011, respectively. The exceedingly low rainfall in 2011 was accompanied by an increase of 1.5 to 3.5°C in mean April air temperature above LTA (Table 21). In 2011, March was also a particularly dry month. At Gleadthorpe (central England), rainfall was c.25% of the monthly LTA, whereas at Boxworth and Terrington (both eastern England) rainfall was c.15% of the monthly LTA. During this time of low rainfall the soil WFPS in all experiments generally remained <40% (sugar beet) or steadily decreased as the soil dried. However, although England in

2012 had a relatively dry February (c.25% of the LTA) and March (c.70% of the LTA), April was very wet with monthly totals c.220-320% of LTA. Rainfall was also high in June and July 2012 where monthly totals in the three English areas ranged from c.160-305% of LTA. With the exception of the OSR grown on a sand at Gleadthorpe, WFPS reflected the wet April, June and July in 2012, with values generally 60% or greater in April and about 50% in June and July.

Whilst LTA rainfall in eastern England is modest and distributed uniformly through the year, rainfall near Edinburgh is greater in total, and its distribution tends towards autumn and winter, being drier from February to June and wetter from September to January (Table 20). In the three years of these experiments, the Scottish sites were relatively wet (compared to English sites) in 2010 (even though rainfall was only c.80% of LTA), then it was very wet in 2011 (c.135% of LTA) and extremely wet in 2012 (c.180% of LTA). Much of the heavy rain fell in summer months with July being particularly wet in all three seasons; c.145%, c.250% and c.395% of the July LTA was recorded in 2010, 2011 and 2012 respectively. As in England, in 2012 February (c.60% of LTA) and March (c.35% of LTA) were relatively dry. In contrast, April to September were extremely wet; c.180-325% of LTA. Temperatures in Scotland were generally cool during the three years of these experiments. However, the spring of 2011 had warm periods (the mean April air temperature was c.2.5°C above LTA), autumn 2011 was warm (c.0.5-2.0°C above LTA) and the early spring of 2012 was generally warm (mean air temperature in March was 2.0°C above LTA). The rainfall distribution was important in determining patterns of WFPS. In Scotland the dry spring of 2010 (c.60% of LTA) resulted in very low WFPS, often less than 40% during April and May when fertiliser applications were made. Following summer rainfall during July and August WFPS increased to around 50%. The summers of 2011 and 2012 were wetter in Scotland, and through much of the summer period in these years WFPS values were close to 60%.

Table 20. Monthly cumulative rainfall (mm) in 2010, 2011 and 2012, and the long term average (LTA) monthly cumulative rainfall (1980–2009).

Area	Year	Jan	Feb	March	April	May	June	July	Aug	Sept	Oct	Nov	Dec	Total
<b>1. Boxworth</b>	LTA	45	32	38	46	47	53	45	47	56 55	60	54	42	564
	2010	40	65	27	15	28	27	18	136		46	26	17	501
	2011	52	29	5	2	21	49	47	47	27	15	41	45	379
	2012	40	10	24	101	41	103	115	35	35	71	73	83	731
<b>2. Gleadthorpe</b>	LTA	54	41	44	54	50	70	48	56	55	65	56	58	650
	2010	59	59	53	16	23	51	43	52	59	58	62	37	572
	2011	30	58	10	3	35	35	32	36	29	26	26	76	397
	2012	46	10	28	173	32	112	84	84	50	71	100	100	890
<b>3. Terrington</b>	LTA	57	37	46	44	49	49	57	56	56	64	60	49	618
	2010	53	69	51	17	24	35	25	124	79	63	31	29	600
	2011	52	48	6	7	25	50	52	52	32	28	50	47	449
	2012	48	9	35	116	38	148	134	44	61	62	109	93	897
<b>4. Bush Estate, Central Scotland</b>	LTA	101	67	74	60	68	76	81	80	80	108	94	91	979
	2010	22	534	51	32	47	29	117	55	86	64	192	59	807
	2011	74	181	113	43	90	133	202	23	124	73	96	159	1,312
	2012	93	39	25	160	141	246	319	155	146	175	112	171	1,775

Table 21. Mean monthly air temperature in 2010, 2011 and 2012 versus the long term average (LTA) mean monthly air temperature (1980–2009)

Area	Year	Jan	Feb	March	April	May	June	July	Aug	Sept	Oct	Nov	Dec
<b>1. Boxworth</b>	LTA	4.1	4.2	6.5	8.6	11.7	14.8	17.2	17.3	14.6	10.9	6.9	4.5
	2010	4.3	5.0	9.1	11.6	13.5	16.3	20.2	20.9	16.8	13.0	8.7	4.6
	2011	4.0	4.8	7.3	10.1	12.2	15.0	18.0	19.4	15.6	12.2	8.5	4.7
	2012	3.3	3.1	5.7	7.4	10.3	13.2	16.3	17.9	13.5	10.2	6.8	3.5
<b>2. Gleadthorpe</b>	LTA	4.2	4.4	6.4	8.7	11.6	13.3	15.6	14.9	12.8	10.3	7.2	5.3
	2010	1.1	2.0	5.8	8.4	10.2	14.6	16.6	15.1	13.6	9.9	4.7	-1.7
	2011	3.2	6.2	6.2	11.3	11.9	13.7	15.0	15.4	15.0	11.2	8.7	5.6
	2012	4.9	3.9	7.6	7.0	11.4	13.5	15.3	16.0	12.3	8.4	6.1	4.0
<b>3. Terrington</b>	LTA	4.5	4.4	6.5	8.7	11.8	14.5	17.0	17.1	14.5	11.1	7.0	4.9
	2010	1.6	2.8	6.3	9.0	11.1	15.6	18.2	16.0	14.0	9.4	4.9	-0.5
	2011	5.1	5.1	6.7	12.1	9.6	15.0	15.4	16.4	16.1	13.0	9.7	5.2
	2012	3.5	7.8	7.6	11.8	14.0	16.0	17.6	13.5	9.8	6.7	4.9	3.5
<b>4. Bush Estate, Central Scotland</b>	LTA	3.3	3.5	4.9	6.9	9.6	12.2	14.3	14.0	11.8	8.6	5.7	3.5
	2010	1.4	1.5	4.7	7.4	9.6	13.6	14.4	13.4	12.4	9.0	4.1	-0.5
	2011	3.1	3.9	4.6	9.3	9.3	11.4	12.7	12.8	12.2	9.8	7.5	3.5
	2012	5.0	5.1	6.9	4.8	8.2	10.6	12.2	13.4	10.4	5.9	4.4	2.3

### 6.5.2. Annual N<sub>2</sub>O emissions

Annual cumulative direct N<sub>2</sub>O-N emissions from the twenty-four experiments varied markedly between England and Scotland and between years (Table 24). At the English sites in 2010 and 2011 emissions were generally low immediately following N fertiliser application due to the dry spring conditions and low soil moisture contents. Emissions in these years followed later rainfall events, with peak emissions occurring up to c.4 months and up to c.9 months after the last fertiliser split had been applied in 2010 and 2011 respectively. The peak emissions were generally <20 g N<sub>2</sub>O-N ha<sup>-1</sup> d<sup>-1</sup>, but ranged up to c.70 g N<sub>2</sub>O-N ha<sup>-1</sup> d<sup>-1</sup> from winter wheat at Terrington in 2010 and up to c.160 g N<sub>2</sub>O-N ha<sup>-1</sup> d<sup>-1</sup> from OSR at Boxworth in 2011. In contrast in 2012 at the English sites, emissions generally occurred soon after April fertiliser applications, with peak emissions of up to c.300 g N<sub>2</sub>O-N ha<sup>-1</sup> d<sup>-1</sup> from winter barley at Terrington. However, peak emissions from OSR at Gleadthorpe (with earlier fertiliser applications) and from the sugar beet at Terrington (with later fertiliser applications) did not occur until June and early October respectively, probably due to their different fertiliser application timings compared to winter cereals.

In Scotland annual emissions were generally higher than those observed at the English sites, but there was considerable annual variability. In 2010-11 emissions following fertiliser application were generally <50 g N<sub>2</sub>O-N ha<sup>-1</sup> d<sup>-1</sup> but increased during the summer months in response to rainfall events in this period with peak emissions of up to c.215 g N<sub>2</sub>O-N ha<sup>-1</sup> d<sup>-1</sup> from winter barley in 2010 and up to c.175 g N<sub>2</sub>O-N ha<sup>-1</sup> d<sup>-1</sup> from spring barley in 2011. An earlier spring peak of up to c.425 g N<sub>2</sub>O-N ha<sup>-1</sup> d<sup>-1</sup> was measured from winter OSR in 2011 following the onset of rainfall and an increase in temperature. 2012 was an unusually wet year in Central Scotland and this was reflected in high N<sub>2</sub>O emissions across all treatments, which continued for much of the summer period. Peak emissions of over 300 g N<sub>2</sub>O-N ha<sup>-1</sup> d<sup>-1</sup> were observed for winter wheat and spring barley in 2012.

#### **Background emissions**

Where no fertiliser N was applied the annual N<sub>2</sub>O emission (i.e. background emission) ranged from 0.01 to 6.78 kg N<sub>2</sub>O-N ha<sup>-1</sup> (Table 24) and varied considerably between England and Scotland, and between years (Table 22). The mean annual N<sub>2</sub>O-N emission from the 0N treatment was generally much greater ( $P<0.001$ ) in Scotland at 1.93 kg N<sub>2</sub>O-N ha<sup>-1</sup> than in England at 0.42 kg N<sub>2</sub>O-N ha<sup>-1</sup>. The successive years 2010, 2011 and 2012 showed significant ( $P<0.01$ ) increasing annual average background emissions of 0.58, 0.76 and 1.10 kg N<sub>2</sub>O-N ha<sup>-1</sup> respectively. There was, however, a significant interaction ( $P<0.01$ ) between country and year with the lowest emission measured in England in 2011, whereas the lowest emission was measured in Scotland in 2010 (Table 22).

**Table 22. Mean cumulative annual N<sub>2</sub>O emissions (kg N<sub>2</sub>O-N ha<sup>-1</sup>; log values analysed and means then back transformed) measured from the control (0N) treatment. Back transformed lower and upper confidence limits in parentheses.**

Year of N application	Country	
	England (n=5)	Scotland (n=3)
<b>2010</b>	0.47 (0.23-0.76)	0.80 (0.41-1.33)
<b>2011</b>	0.26 (0.06-0.50)	2.41 (1.58-3.53)
<b>2012</b>	0.57 (0.31-0.89)	3.30 (2.23-4.76)

Year,  $P < 0.01$ ; Country,  $P < 0.001$ ; Year\*Country,  $P < 0.01$

**Table 23. Mean annual N<sub>2</sub>O EF (% total N applied) for the 120% treatment. SED in parentheses.**

Year of N application	Country	
	England (n=5)	Scotland (n=3)
<b>2010</b>	0.31 (0.12)	0.85 (0.15)
<b>2011</b>	0.21 (0.12)	1.50 (0.15)
<b>2012</b>	0.91 (0.12)	0.68 (0.15)

Year,  $P < 0.05$ ; LSD = 0.26; Country,  $P < 0.001$ , LSD = 0.22; Year\*Country,  $P < 0.001$ ; LSD = 0.43

### ***Fertiliser induced emissions***

As with the emissions from the 0N treatment, the calculated EFs (the percentage of N applied which was lost as N<sub>2</sub>O) at 120% of the recommended N rate were greatly influenced by country and year (Table 23), and ranged from 0.06 to 2.82%. The mean EF from Scotland of 1.0% of total N applied was more than twice that from England of 0.48% ( $P < 0.001$ ; LSD = 0.22); these compare to the IPCC Tier 1 EF<sub>1</sub> of 1.0% (IPCC, 2006). The year also significantly ( $P < 0.05$ ; LSD = 0.26) affected the EF with emissions increasing in order from 2010 (0.51%) to 2011 (0.69%) and then again to 2012 (0.83%). There was, however, a significant interaction ( $P < 0.001$ ; LSD = 0.43) between country and year with lower mean EFs in England than Scotland in 2010 and 2011 and no difference in 2012. The lowest mean EF (0.21%) in England was in 2011 and the greatest (0.91%) was in 2012. In contrast the lowest mean EF (0.68%) in Scotland was in 2012 and the greatest (1.50%) was in 2011. Presumably, although the highest rainfall in Scotland was measured in 2012, the very wet conditions following fertiliser application promoted denitrification to di-nitrogen rather than to N<sub>2</sub>O. The site-season interaction for these EFs (Table 23) probably differed from that for background emissions (Table 22) because fertiliser-related and background emissions were distributed through the year in a different way.

At the English sites in 2010 and 2011 the EFs at 120% of the recommended rate were generally about 0.3% or less of total-N applied; the largest EFs of 0.46% and 0.64% were from winter wheat at Boxworth and sugar beet at Gleadthorpe respectively (Table 24). Wet and warm soil conditions in August were associated with the main N<sub>2</sub>O emission event shortly after the winter wheat harvest. Table 20 and Table 21 show an approximate threefold increase in the monthly rainfall and an approximate 3.5°C increase in the air temperature compared to the LTA for August. In the sugar beet experiment, the peak emission 45 days after the last fertiliser application of c.15 g N<sub>2</sub>O-N ha<sup>-1</sup> d<sup>-1</sup> was also related to rainfall and an associated increase in soil WFPS. Further emissions were also measured in December and January, before and shortly after the sugar beet was harvested. As indicated in Table 21, in late November and early December air temperatures at the site were uncharacteristically cold with the mean daily air temperature as low as -13.5°C. The cold weather had prevented the scheduled harvest of the sugar beet so the crop remained in the field. Consequently, the sugar beet tops were badly damaged by freezing temperatures and started to decompose. It is likely that rather than being directly related to the N fertiliser applications, the measured winter emissions were associated with the decomposition of the beet tops and/or freeze-thaw perturbations (Kaiser *et al.*, 1998; Mørkved *et al.*, 2006). Table 22 shows the estimated cumulative emissions and EFs (0.38% of total-N applied from the 120% treatment) which were estimated to arise from the N fertiliser application only i.e. excluding the winter emissions.

Similarly to 2010, in otherwise dry soil conditions, N<sub>2</sub>O emissions in 2011 at the English sites were generally associated with a few dates following rainfall when soil moisture content increased. The largest EF from the 120% treatment in 2011 was 0.52% for OSR at Boxworth (Table 24) and resulted from N fertiliser application to wet soils (WFPS >70%) in early spring. Despite UK soil conditions generally being moist or wet in February, cold temperatures typically inhibit N<sub>2</sub>O losses (McTaggart *et al.*, 1997; Thorman *et al.*, 2007). In this experiment however, the first split of N fertiliser applied in late February coincided with a rise in air temperature to >10°C, hence the relatively large N<sub>2</sub>O emissions of up to c.160 g N<sub>2</sub>O-N ha<sup>-1</sup> d<sup>-1</sup>.

At the English sites in 2012 the EFs from the 120% N treatment were all ≥0.65% of total-N applied with peak emissions in late spring or early summer ranging from c.35 to 295 g N<sub>2</sub>O-N ha<sup>-1</sup> d<sup>-1</sup> as a result of significant rainfall (Table 20). The exception was with sugar beet at Terrington where the largest EF of 1.25% of total-N applied occurred; however, here the N<sub>2</sub>O emission pattern over time was different to the other crops. Although the peak emission of c.70 g N<sub>2</sub>O-N ha<sup>-1</sup> d<sup>-1</sup> was associated with rainfall it did not occur until October when the topsoil SMN had been depleted. It is unclear why emissions from sugar beet occurred so late in the year and particularly the peak emission; this may have resulted from direct emissions from fallen sugar beet leaves or possibly it reflected emissions from the sub-soil.

Table 24. Effect of spring ammonium nitrate fertiliser N at selected application rates (nil and N rates spanning the recommended N rate) on mean (n=3) cumulative N<sub>2</sub>O emissions estimated over the 12 months following N application.

Area	Crop type	Treatment – N fertiliser application rate as % of the recommended rate (spring barley rates in parentheses)					
		kg N <sub>2</sub> O-N ha <sup>-1</sup>				% of total N applied	
		0%	80% (60%)	120% (120%)	LSD (5% level)	80% (60%)	120% (120%)
<b>2010</b>							
<b>1. Boxworth</b>	Winter wheat	0.45	1.20	1.83	1.20	0.37	0.46
	Winter OSR	0.41	0.53	0.76	0.19	0.07	0.13
<b>2. Gleadthorpe</b>	Spring barley	0.33	0.38	0.47	0.16	0.07	0.10
	Sugar Beet*	0.31 (0.22)*	1.38 (0.79)*	1.21 (0.74)*	0.62 (0.39)	1.07 (0.57)*	0.64 (0.38)*
<b>3. Terrington</b>	Winter wheat	0.80	1.33	1.28	0.73	0.30	0.19
<b>4. Bush Estate</b>	Winter barley	1.92	2.70	3.39	1.73	0.55	0.68
	Winter wheat	0.74	1.59	2.69	1.50	0.58	0.88
	Spring barley	0.15	1.18	1.59	1.04	1.43	1.00
<b>2011</b>							
<b>1. Boxworth</b>	Winter wheat	0.14	0.33	0.34	0.15	0.10	0.07
	Winter OSR	0.27	1.11	1.65	0.27	0.48	0.52
<b>2. Gleadthorpe</b>	Winter barley	0.35	0.88	1.10	0.27	0.33	0.31
<b>3. Terrington</b>	Winter wheat	0.84	0.86	1.04	0.27	0.01	0.07
	Sugar beet	0.18	0.34	0.29	0.18	0.17	0.08
<b>4. Bush Estate</b>	Winter OSR	4.82	8.44	9.44	3.49	2.26	1.93
	Winter wheat	1.62	2.24	1.77	1.36	0.39	0.06
	Spring barley	0.38	2.92	4.28	2.08	3.53	2.71
<b>2012</b>							
<b>1. Boxworth</b>	Winter wheat	0.68	2.05	2.63	0.95	0.68	0.65
<b>2. Gleadthorpe</b>	Winter wheat	0.40	1.58	1.88	0.90	0.84	0.68
	Winter OSR	0.66	1.78	2.71	1.04	0.74	0.89
<b>3. Terrington</b>	Winter barley	0.36	1.72	3.01	0.87	0.85	1.10
	Sugar beet	1.50	2.18	3.32	0.96	0.71	1.25
<b>4. Bush Estate</b>	Winter OSR	0.01	0.97	2.93	1.67	0.54	1.09
	Winter wheat	6.78	7.67	7.47	2.45	0.49	0.25
	Spring barley	3.46	3.49	4.48	1.29	0.04	0.69

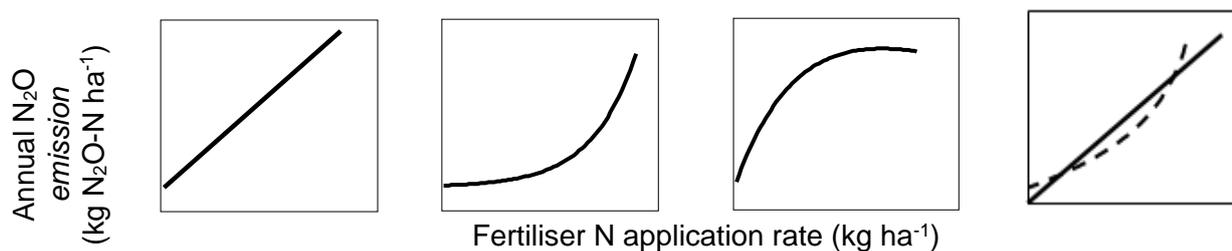
\* Values in parentheses represent the cumulative N<sub>2</sub>O-N emission estimated to arise from N fertiliser application only and was measured over 207 days (02/11/10). The value outside the parentheses includes emissions associated with decomposition of the sugar beet tops and possibly also due to freeze-thaw effects.

In Scotland in 2010 the largest EF of 1.0% from the 120% treatment was from spring barley; this possibly reflects the poorer ability of the young root systems of spring cereals to recover applied fertiliser N. In 2011 the largest Scottish EF from the 120% treatment (2.71%) was also observed from spring barley. Again it is possible that the established root systems of the winter crops could have exploited (or 'immobilised') the fertiliser N more thoroughly. However, a large part of the annual flux observed during 2011 occurred in a period where only monthly measurements were being made, so it is possible that the magnitudes of fluxes (and their EFs) were slightly overestimated. In 2012 the largest Scottish EF from the 120% treatments (1.09%) was observed for winter OSR. Due to earlier fertiliser application, N<sub>2</sub>O emissions for OSR were also much earlier in the year than for cereals; they occurred at a time of less rainfall and therefore more optimal conditions for N<sub>2</sub>O (rather than N<sub>2</sub>) production. The wet weather in Scotland during 2012 was associated with a rather different pattern of emissions to other years. Total area-based emissions were high, ranging from 0.97-7.67 kg N<sub>2</sub>O-N ha<sup>-1</sup> y<sup>-1</sup>, but emissions from the zero and controls were also high (except for winter OSR) resulting in EFs similar to or less than default values.

### **Effect of N fertiliser application rate**

The cumulative annual N<sub>2</sub>O emissions in each experiment were described in relation to total fertiliser N applied by one of four curves: linear, increasing quadratic, decreasing quadratic, and indeterminable (Table 25). Over all three years, there were approximately similar numbers of linear and non-linear (increasing quadratic) responses. More experiments showed a non-linear (increasing quadratic) response in 2012, the wettest year, when the emissions were largest. Two experiments in 2011 were best described by a non-linear (decreasing quadratic) relationship; winter OSR at Boxworth and at Bush Estate.

**Table 25. The occurrence of significant (P<0.05) linear and non-linear (quadratic) responses of annual N<sub>2</sub>O emission to fertiliser N application rate from all 24 experiments.**



Year	Occurrence of N <sub>2</sub> O response			
	Linear	Increasing quadratic	Decreasing quadratic	Indeterminable
2010	4	4	-	-
2011	4	2	2	-
2012	2	5	-	1
Total	10	11	2	1

There was an apparent difference in response shapes between England and Scotland, with eight of the ten linear responses being from English sites, and six of the eleven non-linear (increasing

quadratic) responses being from Scotland. Of the five English non-linear (increasing quadratic) responses, three occurred in 2012.

### **Responses to fertiliser N over 12 months**

By analysing responses to applied N of both topsoil mineral N (0-10 cm) and N<sub>2</sub>O emissions (daily and cumulative), it was possible to show the main dates by which the eventual response shape of N<sub>2</sub>O emissions had developed and to give an insight into what was driving the N<sub>2</sub>O response.

#### *Soil mineral N*

Figure 13 provides an example (winter OSR fertilised in 2011 at Boxworth) of how a typical response of SMN (0-10 cm) to fertiliser N rate evolved through successive sampling dates.

Inspecting these over the 24 experiments it was clear that the response of SMN to N rate reflected the quantity and pattern of fertiliser application in each split, so the N timing regime used here appears to have, in itself, caused deviations from a linear response. Table 26 shows the amounts of fertiliser N applied at each split for each treatment in the example experiment chosen. In order for the experiments to be as commercially relevant as possible fertiliser N was applied following guidelines in 'The Fertiliser Manual' (RB209) (Anon., 2000; Anon., 2010) for the English sites and in 'Nitrogen recommendations for cereals, OSR and potatoes' (TN625) (Sinclair *et al.*, 2009) for Scotland. Table 26 shows that, in following these guidelines, although the total amount of N increased linearly, this was not necessarily the case at each split application.

**Table 26. Fertiliser N applied (kg ha<sup>-1</sup>) for each split and treatment on OSR at Boxworth in 2011.**

Treatment	Fertiliser N applied in each split application (kg N ha <sup>-1</sup> )		
	1st	2nd	Total
<b>0% recommended rate</b>	0	0	0
<b>40% recommended rate</b>	90	0	90
<b>80% recommended rate</b>	90	90	180
<b>120% recommended rate</b>	135	135	270
<b>160% recommended rate</b>	180	180	360

Data following the first fertiliser application (24<sup>th</sup> February) e.g. on the 25<sup>th</sup> and 26<sup>th</sup> February (highlighted; black, solid line) in Figure 13 clearly show the SMN concentrations mirroring the fertiliser N applications of 90 kg ha<sup>-1</sup> for both the 40% and 80% treatments. Similarly, following the second fertiliser application (29<sup>th</sup> March) e.g. on the 30<sup>th</sup> and 31<sup>st</sup> March (highlighted; black, dashed line) the lack of fertiliser applied for the 40% treatment can be seen with c.25 kg ha<sup>-1</sup> SMN; roughly the same quantity as was measured two days earlier. Conversely, the SMN quantities all significantly increased in the other treatments where N had been applied. In this example, a linear fit best described the data, although it clearly wasn't able to account for the low SMN in the 40% treatment. Indeed, in many of the experiments, as a result of the N application pattern, the line of

best fit did not perfectly describe the data and caused inconsistency over time between linear and non-linear fits (Figure 13). The inconsistency was exacerbated when variability in SMN data was high, when there were few N rates (e.g. in this OSR example after the 1<sup>st</sup> fertiliser application) or (unlike in this example), when there was little SMN remaining after previous applications in the 40% (or 60%) treatment, and no N was applied in the last application.

The shape of the SMN response to N rate was also influenced by transformations through time, presumably due to mineralisation or immobilisation as well as crop N uptake. This is illustrated in the period between 30<sup>th</sup> March (highlighted; black, dashed line) and 21<sup>st</sup> June (highlighted; green, solid line) in Figure 13 where short term effects of fertiliser on mineralisation and immobilisation, as well as crop uptake (data not shown), changed the response from a linear to a non-linear fit.

#### *Nitrous oxide – N<sub>2</sub>O*

Figure 14 shows the response to fertiliser N rate of N<sub>2</sub>O emission (g N<sub>2</sub>O-N ha<sup>-1</sup> d<sup>-1</sup>) measured on each sampling occasion. Analysis of the daily N<sub>2</sub>O emissions from all the experiments indicated that, given large enough emissions, the response of N<sub>2</sub>O emission to N rate generally reflected the response of SMN to N rate (Figure 13). This is evident on the 26<sup>th</sup> February when the peak N<sub>2</sub>O emission (c.160 g N<sub>2</sub>O-N ha<sup>-1</sup> d<sup>-1</sup>; Figure 14) occurred and resulted in a linear response of N<sub>2</sub>O emission to N rate, mirroring the linear response of SMN to N rate (Figure 13, highlighted; black, solid line). However, even with relatively large N<sub>2</sub>O emissions (e.g. 25<sup>th</sup> February, highlighted; black, solid line) this was not always the case. The inherent variability associated with N<sub>2</sub>O emissions tended to prevent N<sub>2</sub>O responses always being explained by the SMN responses. Hence, more than with the SMN data, linear or quadratic shapes were less common in the N<sub>2</sub>O data and there was inconsistency over time in which of these gave the better fit (Figure 14).

It is not surprising that SMN appears as a strong driver of N<sub>2</sub>O emissions, but these data also show that availability of SMN alone is insufficient to induce N<sub>2</sub>O emissions and to confer its response to N rate onto the N<sub>2</sub>O response. Daily emissions measured on the 1<sup>st</sup> and 4<sup>th</sup> April (Figure 13) were small ( $\leq$  c.6 g N<sub>2</sub>O-N ha<sup>-1</sup> d<sup>-1</sup>) with no significant ( $P > 0.05$ ) response of N<sub>2</sub>O to N rate, despite the presence of large amounts of mineral N displaying a clear non-linear response (Figure 13, highlighted; blue, solid line). It is apparent that the shape of the SMN response to N rate will only be conferred onto the N<sub>2</sub>O emission response if the soil conditions are sufficiently wet. On the 26<sup>th</sup> February (Figure 13 and Figure 14, highlighted; black, solid line) soil conditions were very wet (WFPS >80%) which, along with the recently applied fertiliser N, generated large N<sub>2</sub>O emissions with the same linear response pattern as seen with the SMN. Whereas by the 1<sup>st</sup> April the WFPS had been steadily declining as a result of a very dry March (Table 20) so, although there was still plenty of available N, the soil was too dry for the N<sub>2</sub>O response to N rate to follow that of the SMN.

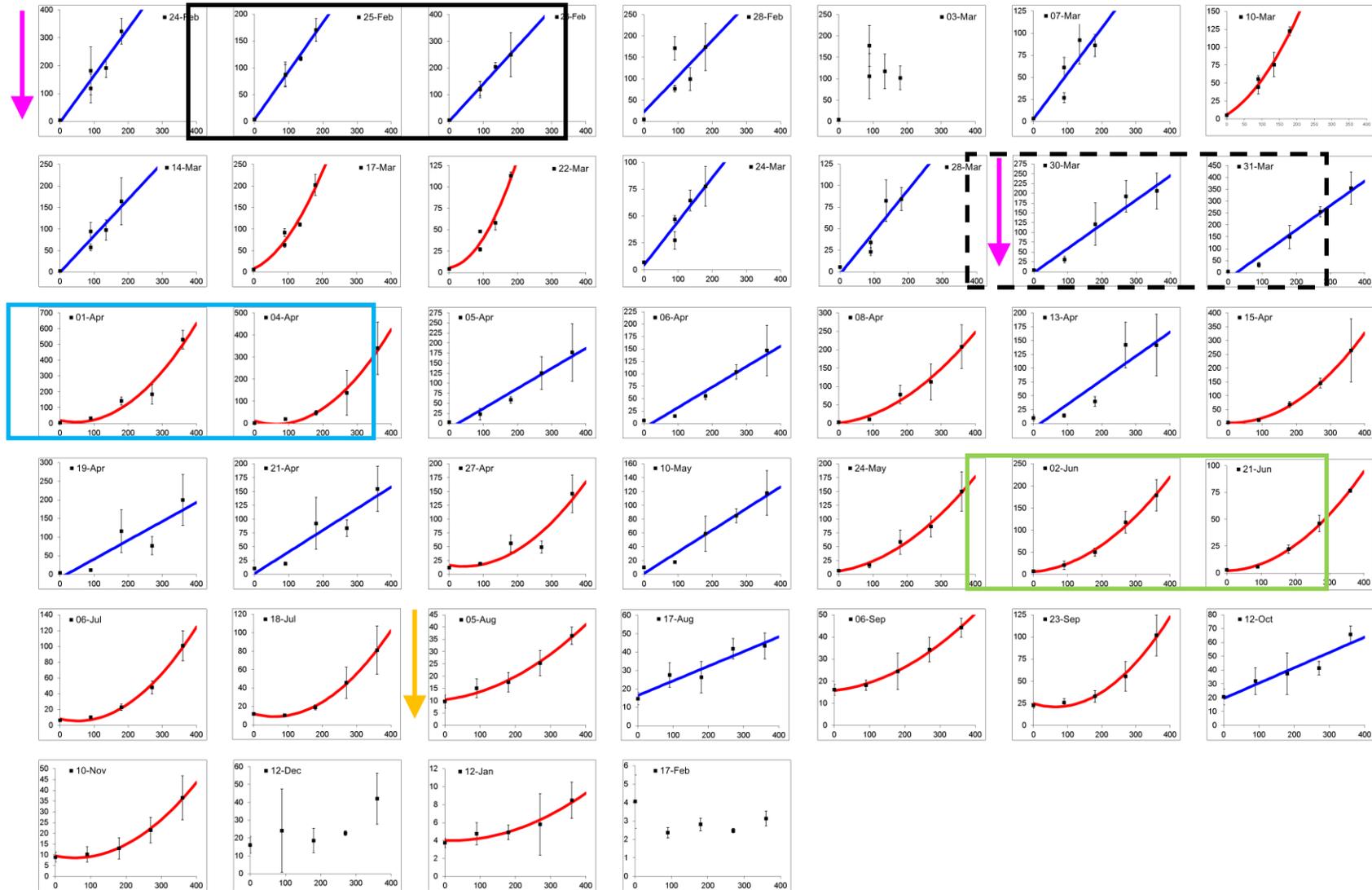
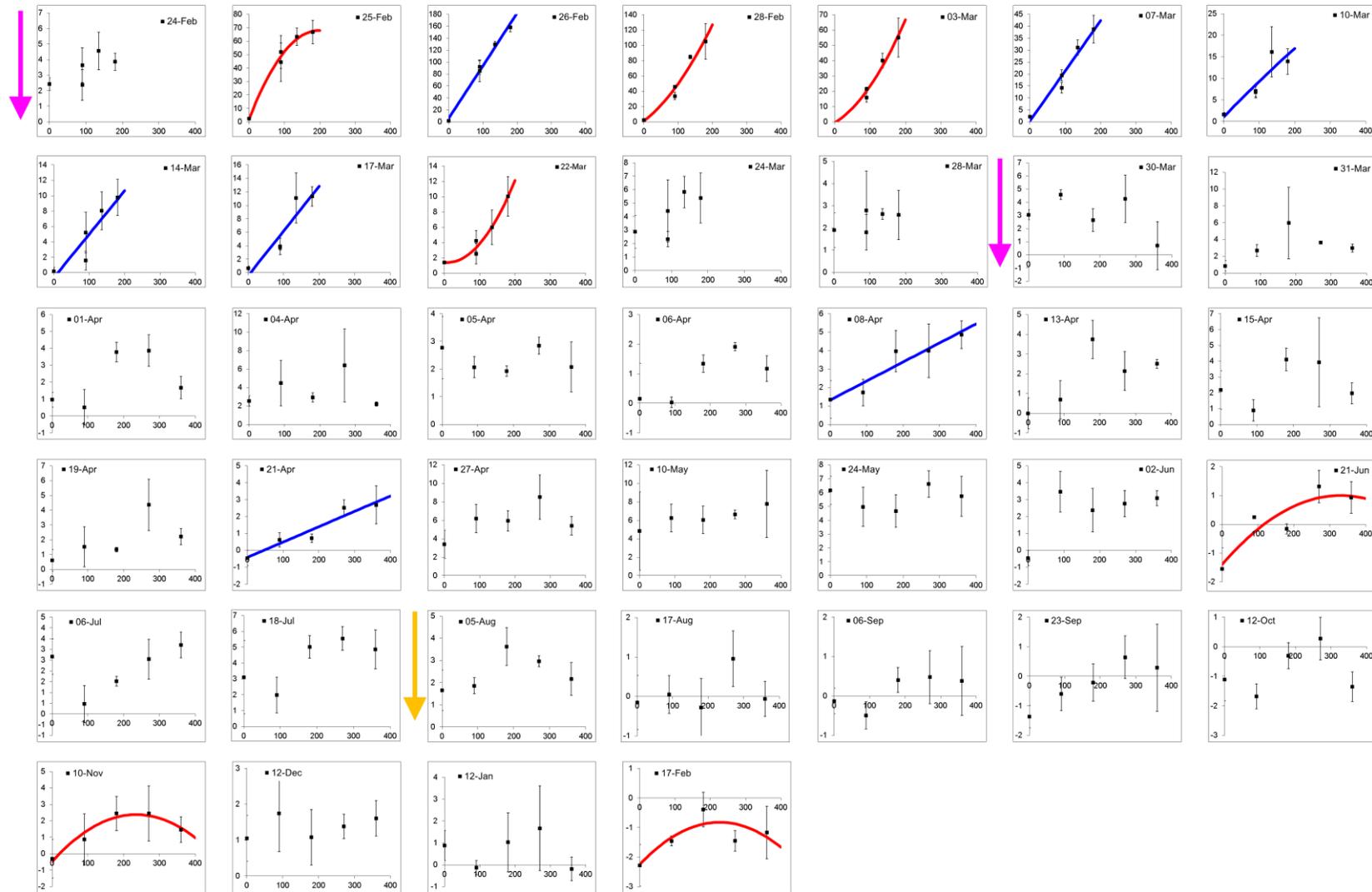


Figure 13. Development over 12 months after the first N fertiliser application of the response of topsoil mineral N ( $\text{kg NO}_3\text{-N} \ \& \ \text{NH}_4\text{-N ha}^{-1}$ ) to applied N for winter OSR fertilised in 2011 at Boxworth. Each graph shows a successive date, from left to right and then row by row from the top. X-axes all show total N applied (up to that date) from nil to  $360 \text{ kg ha}^{-1}$ . Y-axes all show SMN ( $\text{kg ha}^{-1}$ ), but with axis maxima varying, so all shapes can be seen. Blue fit = linear; red fit = non-linear (quadratic). Graphs without lines indicate no significant relationship with N applied. Pink arrows denote fertiliser applications. Orange arrow denotes harvest. Highlighted boxes are referred to in the text.



**Figure 14.** The responses of daily N<sub>2</sub>O flux over the 12 months after winter OSR received fertiliser N in spring 2011 at Boxworth. Each graph shows a successive date, from left to right and then row by row from the top. X-axes all show total N applied (until that date) from nil to 360 kg ha<sup>-1</sup>. Y-axes all show daily N<sub>2</sub>O emission (g N<sub>2</sub>O-N ha<sup>-1</sup> d<sup>-1</sup>), but with axis maxima varying, so all shapes can be seen. Blue fit = linear; red fit = non-linear (quadratic). Graphs without lines indicate no significant relationship with N applied. Pink arrows denote fertiliser application. Orange arrow denotes harvest. Highlighted dates are referred to in the text.

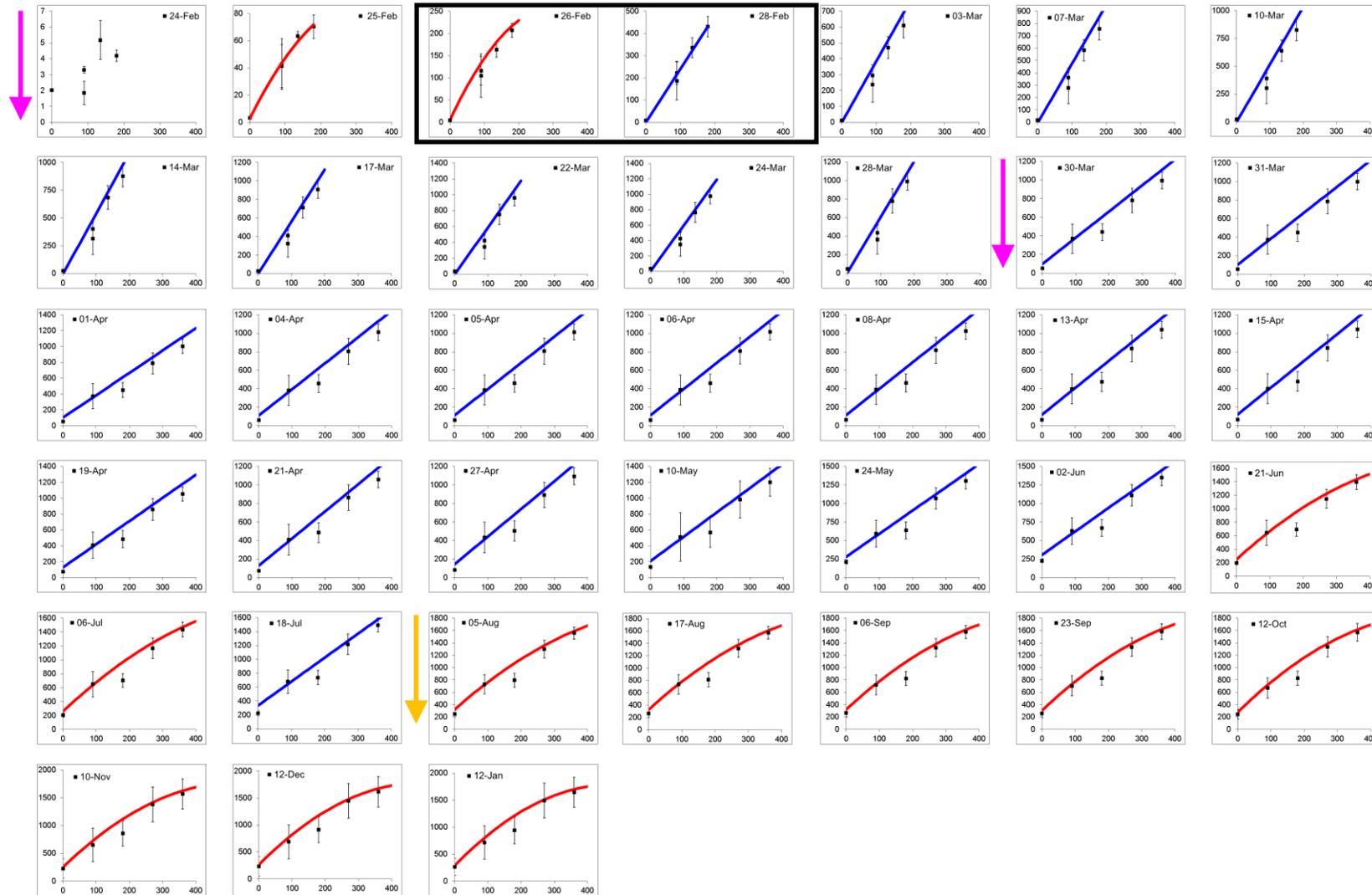


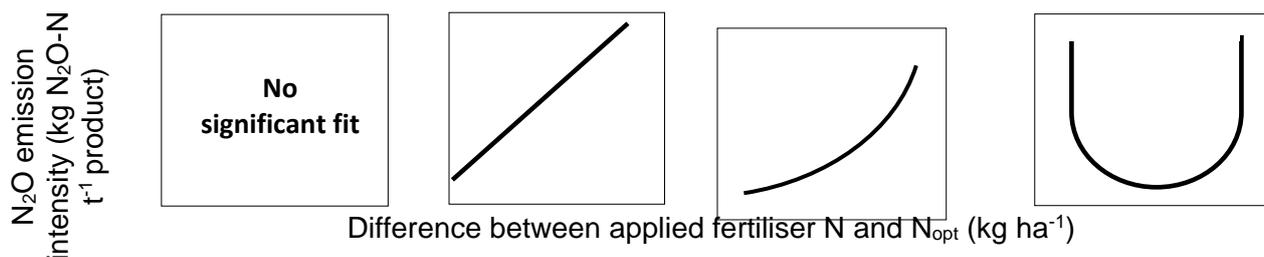
Figure 15. Development over 12 months of the response of cumulative N<sub>2</sub>O flux to N fertiliser applied to winter OSR in spring 2011 at Boxworth. Each graph shows a successive date, from left to right and then row by row from the top. X-axes all show N applied up to that date from nil to 360 kg ha<sup>-1</sup>. Y-axes all show cumulative N<sub>2</sub>O emission (g N<sub>2</sub>O-N ha<sup>-1</sup>), but with axis maxima varying, so all shapes can be seen. Blue fit = linear; red fit = non-linear (quadratic). Graphs without lines indicate no significant relationship with N applied. Pink arrows denote fertiliser application. Orange arrow denotes harvest. Highlighted dates are referred to in the text.

It was evident from examining data from all experiments that, because of the contrast in N<sub>2</sub>O emissions between dates, the largest 'peak' daily N<sub>2</sub>O emissions were dominant in determining the final cumulative N<sub>2</sub>O response curve. Using the same example as for Figure 13 and Figure 14, Figure 15 illustrates evolution of the cumulative N<sub>2</sub>O emission over the successive sampling dates, and shows how the final response pattern develops through the year. On the 26<sup>th</sup> February (highlighted; black, solid line) the response of the cumulative N<sub>2</sub>O emission was non-linear. However, the large daily emissions measured on the 26<sup>th</sup>, which exhibited a linear response to N rate (Figure 14) were large enough to 'switch' the cumulative N<sub>2</sub>O response to a linear fit by the next sampling date (28<sup>th</sup> February) and the best fit relating cumulative N<sub>2</sub>O emission to N rate remained linear for nearly the next 4 months. However, despite no daily emissions after the middle of March being >10 g N<sub>2</sub>O-N ha<sup>-1</sup> d<sup>-1</sup>, and frequently being <5 g N<sub>2</sub>O-N ha<sup>-1</sup> d<sup>-1</sup> (Figure 14), the combination of variability in the daily N<sub>2</sub>O emissions and the interpolation method used to calculate cumulative emissions meant that the final response of cumulative N<sub>2</sub>O emission to fertiliser N rate was non-linear (decreasing) (Figure 14). It would appear that in some of our experiments the final shape of the cumulative N<sub>2</sub>O response to N rate was merely the product of 'luck' rather than the result of a distinct emission event. This occurred not only when, as in this example, the dominant peak was measured near the beginning of the 12 month measurement period; it was particularly apparent in the low emission conditions prevalent in England in 2010 and 2011 when peak N<sub>2</sub>O emissions were generally <20 g N<sub>2</sub>O-N ha<sup>-1</sup> d<sup>-1</sup>.

### ***Nitrous oxide emission intensities – N<sub>2</sub>O per tonne***

The combined responses of cumulative N<sub>2</sub>O per ha and crop yield to fertiliser N rate gave rise to observed shapes of emission intensity (N<sub>2</sub>O t<sup>-1</sup>) responses, as introduced in Figure 1b. Observed curves are summarised in Table 27. There were ten experiments where the line of best fit for N<sub>2</sub>O t<sup>-1</sup> was quadratic, supporting the 'U-shape' hypothesis proposed at the outset (Figure 1b), although not all these English experiments showed a non-linear (increasing quadratic) relationships for N<sub>2</sub>O ha<sup>-1</sup> (Figure 1a). The extent to which linear and non-linear responses could be distinguished by statistical tests depended on the extent of curvature and the precision of the data; the slightly non-linear, 'hockey stick' response, as would be predicted from combining a linear response in emission per ha with a curvilinear response in crop yield (Figure 1b) were difficult to detect with certainty. Despite ten observed linear relationships between N rate and N<sub>2</sub>O per ha (Table 25), there were still four experiments with no significant relationship between emission intensity and applied N, and seven responses for which a linear relationship was the best fit (Table 27). This left 13 of the 24 experiments with some degree of significant curvature, of which ten could be considered 'U-shaped' (i.e. showed a minimum which wasn't at 0N applied). Again, curved responses were much more common in Scotland (c.65% of total responses were 'U-shaped') and less common at English sites (c.15% of total responses were 'U-shaped').

**Table 27. The occurrence of significant ( $P < 0.05$ ) linear and non-linear (quadratic) responses of  $N_2O$  emission intensity to the difference between fertiliser N supply and  $N_{opt}$  from all 24 experiments.**



Year	Occurrence of response			
2010	2	2	-	4
2011	2	2	1	3
2012	-	3	2	3
Total	4	7	3	10

### 6.5.3. Crop yields and N optima

Crop yields achieved with  $N_{opt}$  in each of the experiments (Table 28) were on average 11% greater than the average UK yield for that crop in that year. The range of  $Y_{opt}$  was from 67% to 165% of average annual UK yields. Only four experiments yielded less than average. The poorest performing crops were OSR at Bush Estate in 2012 where conditions were very wet and dull, and winter wheat at Boxworth in 2011 where establishment was poor and the spring was very dry, which inhibited compensatory growth. The best yields were from sugar beet at Gleadthorpe in 2010 (91 t ha<sup>-1</sup>), and winter wheat at Terrington in 2011 (12.0 t ha<sup>-1</sup>).

In all cases there was a positive response to fertiliser N (Table 28); the smallest responses were for sugar beet, for which on average 16% of the optimal yield arose due to fertiliser N application, compared to 50% and 54% for winter and spring barley, 40% for OSR and 48% for winter wheat respectively.

The average N recommendation for crops in all experiments was 186 kg ha<sup>-1</sup>, and average  $N_{opt}$  was 191 kg ha<sup>-1</sup>. However, this does not signify that the recommendations were precise; optima varied from 20 kg ha<sup>-1</sup> (Boxworth 2012, winter wheat) to more than 350 kg ha<sup>-1</sup> (in 3 experiments; Table 28), and the difference between  $N_{opt}$  and recommended N was greater than 100 kg ha<sup>-1</sup> in half of the experiments (Table 28). The average difference between  $N_{opt}$  and recommended N was +19 kg ha<sup>-1</sup>. There was a tendency for  $N_{opt}$  rates of the winter wheat crops to increase with  $Y_{opt}$  ( $R^2 = 0.34$ ); the experiment with the greatest  $Y_{opt}$  (Terrington, 2011) also showed the greatest  $N_{opt}$ .

**Table 28. Crop yields at nil and  $N_{opt}$ , with the recommended N rate ( $N_{rec}$ ) and  $N_{opt}$  for all 24 experiments in which effects of N rate on  $N_2O$  emissions were tested.**

Area	Year	Crop	Yield with nil N	$Y_{opt}$	$N_{rec}$	$N_{opt}^*$	$N_{opt}$ minus $N_{rec}$
			<i>t ha<sup>-1</sup></i>	<i>t ha<sup>-1</sup></i>	<i>kg ha<sup>-1</sup></i>	<i>kg ha<sup>-1</sup></i>	<i>kg ha<sup>-1</sup></i>
<b>1. Boxworth</b>	2010	Winter wheat	5.5	8.9	250	148	-102
	2010	Winter OSR	1.6	>3.9	220	>352	>132
<b>2. Gleadthorpe</b>	2010	Spring barley	3.0	5.6	110	83	-27
	2010	Sugar Beet*	69	91	120	85	-35
<b>3. Terrington</b>	2010	Winter wheat	3.9	9.2	220	209	-11
<b>4. Bush Estate</b>	2010	Winter barley	3.1	6.8	175	338	163
	2010	Winter wheat	2.5	7.3	185	296	111
	2010	Spring barley	0.8	6.1	120	255	135
<b>1. Boxworth</b>	2011	Winter wheat	3.7	5.4	250	133	-117
	2011	Winter OSR	1.9	3.9	220	191	-29
<b>2. Gleadthorpe</b>	2011	Winter barley	2.3	6.3	200	156	-44
<b>3. Terrington</b>	2011	Winter wheat	6.3	12	230	355	125
	2011	Sugar beet	65	>78	120	>192	>72
<b>4. Bush Estate</b>	2011	Winter OSR	2.2	>4.5	220	>360	>140
	2011	Winter wheat	5.0	9.4	200	164	-36
	2011	Spring barley	4.5	6.9	120	114	-6
<b>1. Boxworth</b>	2012	Winter wheat	5.3	6.8	250	21	-229
<b>2. Gleadthorpe</b>	2012	Winter wheat	2.2	6.8	180	187	7
	2012	Winter OSR	3.6	4.3	190	80	-110
<b>3. Terrington</b>	2012	Winter barley	3.7	5.4	200	171	-29
	2012	Sugar beet	70	>77	120	>190	>70
<b>4. Bush Estate</b>	2012	Winter OSR	1.8	2.3	220	140	-80
	2012	Winter wheat	3.1	7.1	220	199	-21
	2012	Spring barley	3.1	6.1	120	169	49

\*Break Even Ratios (BER; kg crop produce per kg fertiliser N) for determination of N optima were 5 for cereals and 2.5 for OSR as in The Fertiliser Manual 'RB209'; to derive comparable optima for sugar beet, an average BER of 27 was estimated by ADAS and British Sugar, based on recent variation in sugar beet and fertiliser prices.

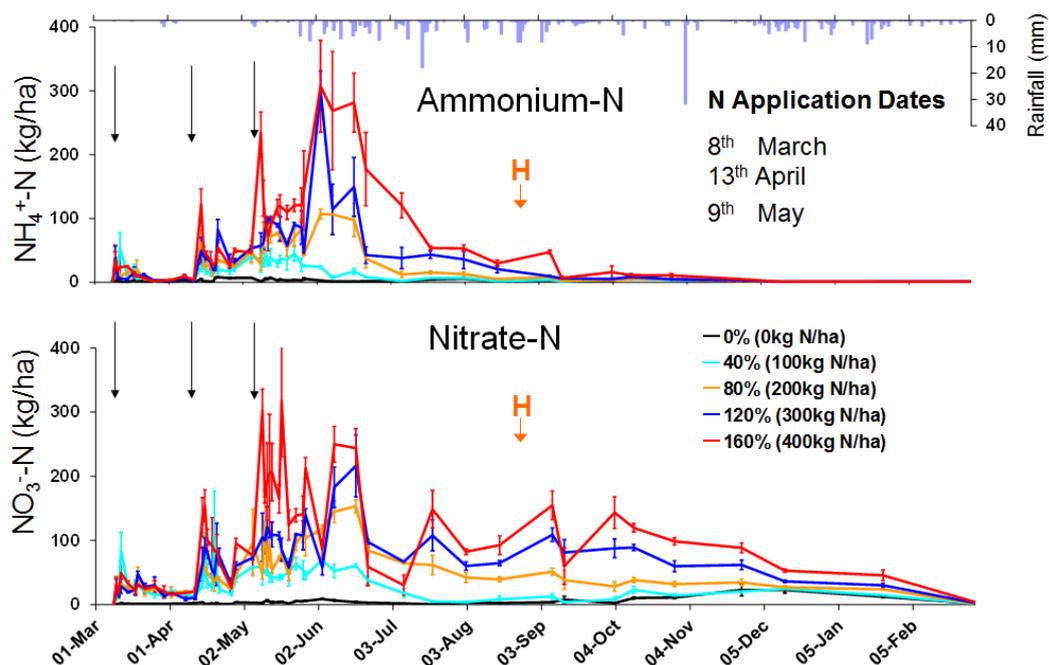
#### 6.5.4. Example response curves

To demonstrate the different responses of  $N_2O$  emissions to fertiliser N rate, and the consequent implications for  $N_2O$  emission intensity, datasets are presented below from example experiments which contrasted in the shapes of responses of  $N_2O$  ha<sup>-1</sup> and  $N_2O$  t<sup>-1</sup> to N rate.

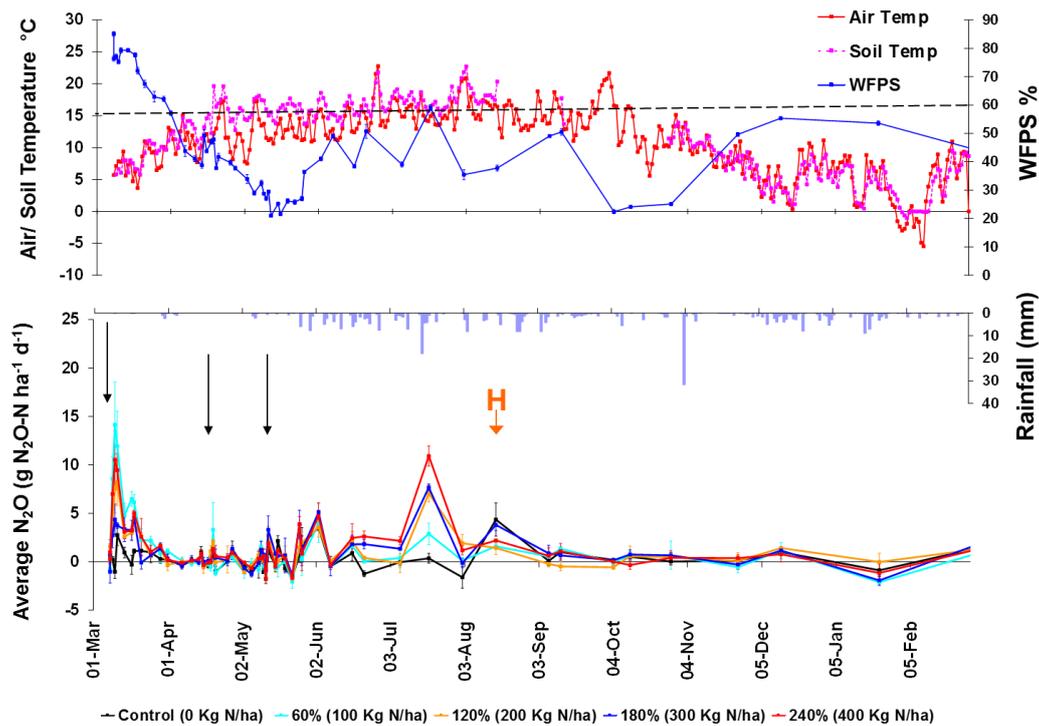
##### **Example 1, Linear hectare<sup>-1</sup>, Linear tonne<sup>-1</sup>: winter wheat at Boxworth in 2011**

In this case, the response of the annual  $N_2O$  emission per hectare to fertiliser N rate was linear (Figure 18) as assumed by the IPCC, and not as hypothesised in this study. At harvest the N surplus (crop N uptake – fertiliser N applied) from the top N rate (160% treatment) was c.275 kg N and there were large amounts of mineral N still in the top 10 cm of soil at this time (Figure 16). It is

unusual for  $c.100 \text{ kg ha}^{-1}$  of  $\text{NO}_3\text{-N}$  still to be present in the top 10 cm of the soil at harvest but this arose here after extremely low rainfall (Figure 17) and exceptionally dry soil conditions after the first two fertiliser applications, and hence very poor crop N uptake and grain yield. In this case the wheat yield at  $N_{\text{opt}}$  was only  $5.4 \text{ t ha}^{-1}$  with a fertiliser N recovery of just 15-35%. Less than  $30 \text{ kg N}$  (above the 0N treatment) had been taken up by the crop from the top N rate at the end of May and even at harvest in mid-August only  $c.60 \text{ kg N}$  (above the 0N treatment) was measured in the crop. When it rained in late May, the crop subsequently had a limited capacity to take up N due to its lack of growth prior to the rainfall. Despite considerable amounts of N in the soil even until harvest, soil conditions did not permit significant emissions of  $\text{N}_2\text{O}$ . The soil was wet enough (75-80% WFPS) to induce a small  $\text{N}_2\text{O}$  emission of up to  $c.15 \text{ g N}_2\text{O-N ha}^{-1} \text{ d}^{-1}$  following the first fertiliser N application (Figure 17). Thereafter, the negligible rainfall resulted in a rapid decline in the WFPS to  $c.20\%$ , during which time daily  $\text{N}_2\text{O}$  emissions were very low (generally  $<5 \text{ g N}_2\text{O-N ha}^{-1} \text{ d}^{-1}$ ). A second  $\text{N}_2\text{O}$  emission of up to  $c.10 \text{ g N}_2\text{O-N ha}^{-1} \text{ d}^{-1}$  was measured in mid-July following significant rainfall ( $c.20 \text{ mm}$ ) and a rise in WFPS. In the months after harvest, WFPS remained low (20-50%) and by the time it had risen to almost 60% the temperature had dropped to about  $c.5 \text{ }^\circ\text{C}$ , which would have inhibited emissions (Dobbie *et al.*, 1999; Dobbie and Smith, 2001; Smith *et al.*, 2012).

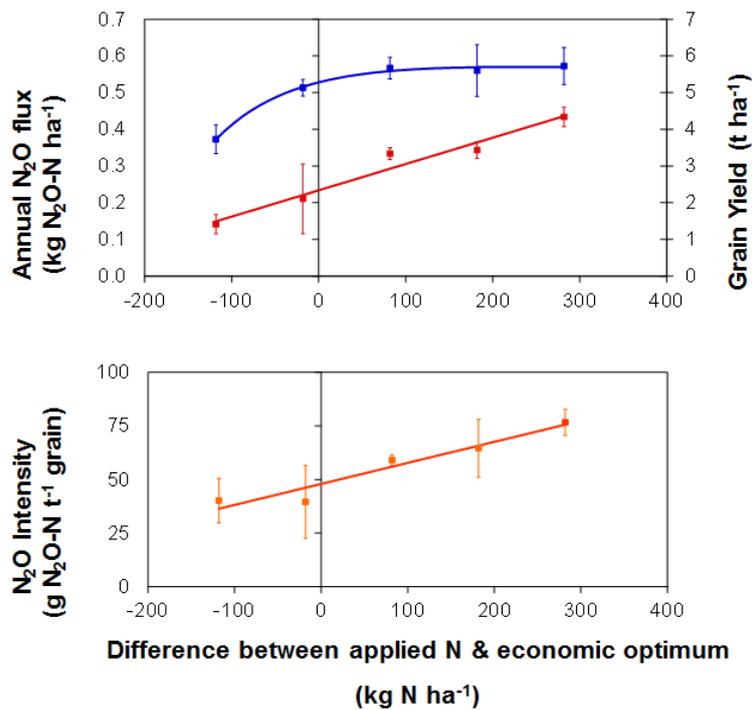


**Figure 16. Example 1 (winter wheat at Boxworth in 2011) topsoil (0-10 cm) ammonium-N ( $\text{NH}_4^+\text{-N}$ ) (top) and nitrate-N ( $\text{NO}_3^-\text{-N}$ ) (below) over 12 months after fertiliser application (arrows); error bars represent 2 x standard error (SE) of the mean. 'H' denotes date of harvest. Hanging blue bars represent total daily rainfall (mm).**



**Figure 17. Example 1 (winter wheat at Boxworth in 2011) mean daily air and soil temperature (°C), percentage water filled pore space (WFPS) (Top) and mean daily N<sub>2</sub>O emissions (g N<sub>2</sub>O-N ha<sup>-1</sup> d<sup>-1</sup>) (below). Error bars represent 2 x SE of the mean. Arrows denote N application date and ‘H’ date of harvest. Hanging blue bars represent total daily rainfall (mm). Dashed line represents 60% WFPS.**

A linear response of N<sub>2</sub>O emission to N rate would, according to our hypothesis, result in a slightly non-linear, ‘hockey stick’ emission intensity curve (Figure 18b). This was not the case however, probably because the response in grain yield was poor and N<sub>2</sub>O emissions were small (particularly from the 0N treatment); when the annual N<sub>2</sub>O emission was expressed per tonne (emission intensity) the best fit was a linear relationship (Figure 18).



**Figure 18. Example 1 (winter wheat at Boxworth in 2011) mean grain yield ( $\text{t ha}^{-1}$ ), cumulative annual  $\text{N}_2\text{O}$  flux ( $\text{kg N}_2\text{O-N ha}^{-1}$ ) (top) and  $\text{N}_2\text{O}$  intensity ( $\text{g N}_2\text{O-N t}^{-1}$  grain) (below) against the difference between applied N and  $\text{N}_{\text{opt}}$  ( $133 \text{ kg N ha}^{-1}$ ). Error bars represent  $2 \times \text{SE}$  of the mean. Bold lines show best fitted relationships with applied N ( $P < 0.05$ ).**

**Example 2, Linear hectare<sup>-1</sup>, ‘U-shaped’ tonne<sup>-1</sup>: Spring barley at Gleadthorpe in 2010**

The response of the annual  $\text{N}_2\text{O}$  emission to fertiliser N rate exhibited in Example 2 was linear (Figure 21). However, in contrast to Example 1, at harvest the N surplus from the top N rate (240% treatment) was only  $c.20 \text{ kg ha}^{-1}$  and the fertiliser N recovery was  $c.70\%$ . This was reflected in the relatively low amount of mineral N (up to  $c.25 \text{ kg ha}^{-1}$ ) measured in the soil (0-10 cm) around harvest-time in early August (Figure 19). After the second and main fertiliser application in late April, SMN content in the top 10 cm of soil rapidly declined during May (Figure 19), which corresponded to the greatest period of crop N uptake (data not shown). The peak  $\text{N}_2\text{O}$  emission, although small at up to  $10 \text{ g N}_2\text{O-N ha}^{-1} \text{ d}^{-1}$ , occurred within 2 weeks of the first N fertiliser application, which followed rainfall and an associated rise in WFPS (Figure 20). At this time, however, there was no effect of N rate on  $\text{N}_2\text{O}$  emissions as all treatments that had received fertiliser had the same amount of N applied ( $40 \text{ kg ha}^{-1}$ ). Subsequently, despite further rainfall, the sandy loam soil texture must have allowed rapid drainage because the WFPS generally was  $<30\%$ , which restricted  $\text{N}_2\text{O}$  emissions typically to  $<5 \text{ g N}_2\text{O-N ha}^{-1} \text{ d}^{-1}$ . A secondary very small  $\text{N}_2\text{O}$  emission was measured in mid-July, which increased linearly with N rate up to  $6 \text{ g N}_2\text{O-N ha}^{-1} \text{ d}^{-1}$ . At this time the WFPS had increased as a result of rainfall, although only to about 30%, and the mineral N in topsoil (0-10 cm) was just  $c.30 \text{ kg ha}^{-1}$ , both conditions serving to constrain  $\text{N}_2\text{O}$  emissions.

There was a good yield response of the spring barley to low fertiliser N rates in this experiment (Figure 21). Even though the fertiliser recovery was good at all N rates (c.75%) it is likely that the dry conditions during grain fill prevented the crop from yielding well, which resulted in a  $Y_{opt}$  of only  $5.6 \text{ t ha}^{-1}$  (Figure 21). In this experiment, although the response of the annual  $\text{N}_2\text{O}$  emission to fertiliser N rate was linear, when the  $\text{N}_2\text{O}$  emissions were scaled per tonne the resultant emission intensity curve conformed with our proposed 'U-shape' (Figure 21).

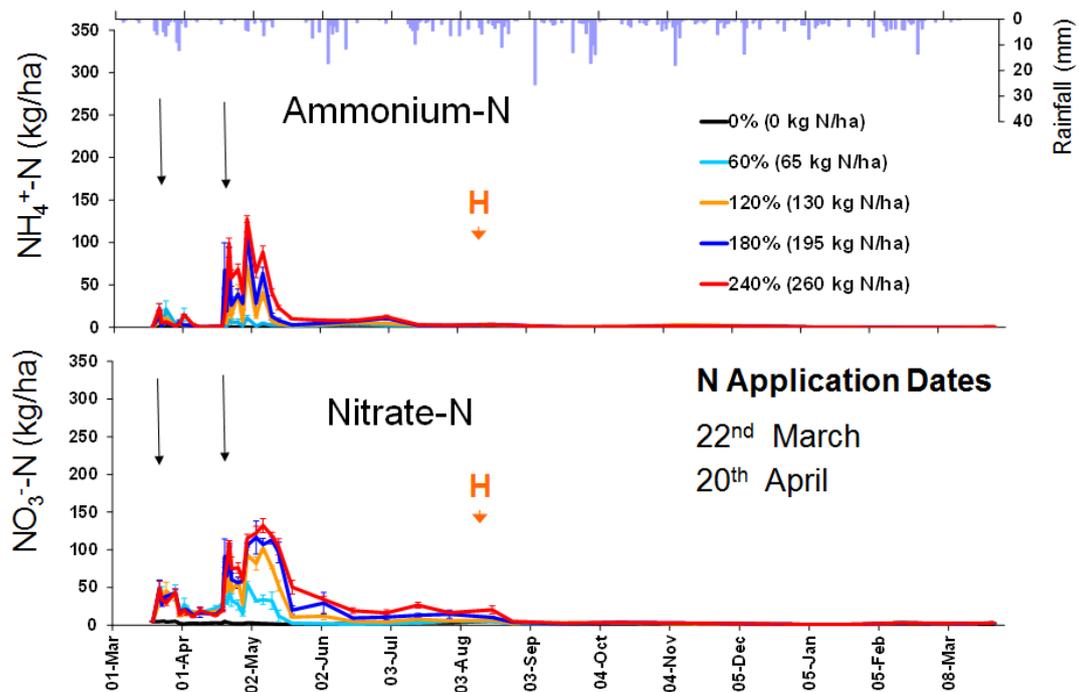


Figure 19. Example 2 (spring barley at Gleadthorpe in 2010) topsoil (0-10 cm) ammonium-N ( $\text{NH}_4^+\text{-N}$ ) (top) and nitrate-N ( $\text{NO}_3^-\text{-N}$ ) (below); error bars represent  $2 \times \text{SE}$  of the mean. Arrows denote N application and 'H' date of spring barley harvest. Hanging blue bars represent daily rainfall (mm).

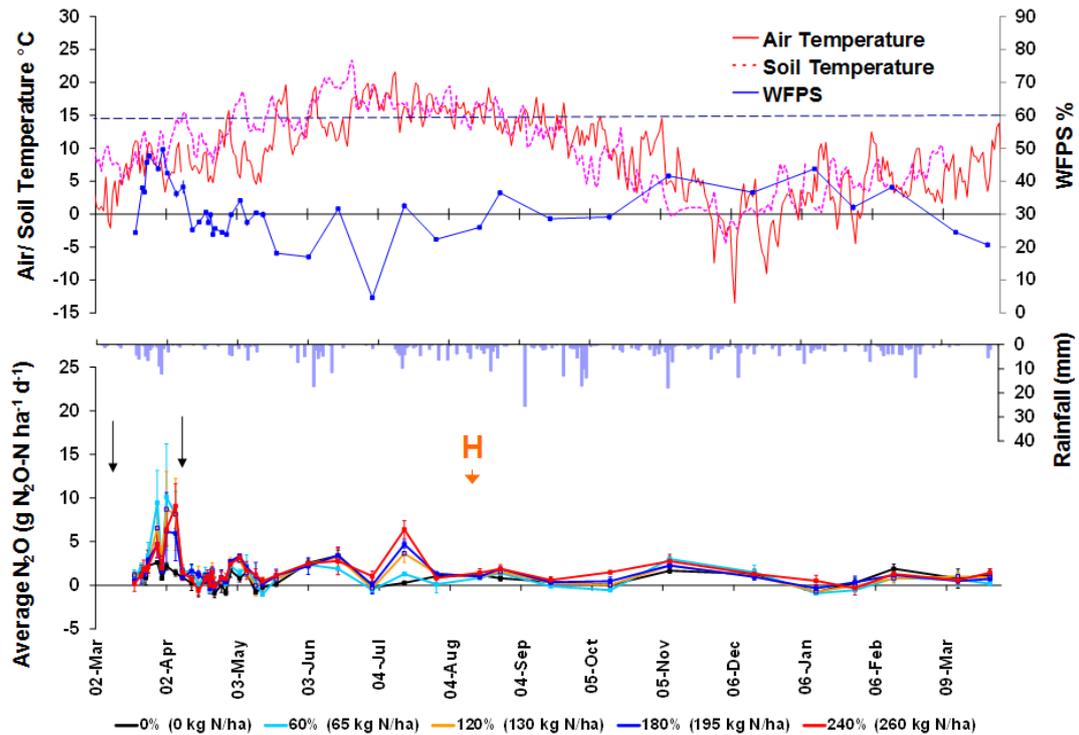


Figure 20. Example 2, 2010, spring barley, Gleadthorpe, mean daily air and soil temperature ( $^{\circ}\text{C}$ ), percentage water filled pore space (WFPS) (Top) and mean daily  $\text{N}_2\text{O}$  emissions ( $\text{g N}_2\text{O-N ha}^{-1} \text{d}^{-1}$ ) (below). Error bars represent  $2 \times \text{SE}$  of the mean. Arrows denote N application date and 'H' date of spring barley harvest. Bars represent total daily rainfall (mm). Dotted line represents 60% WFPS.

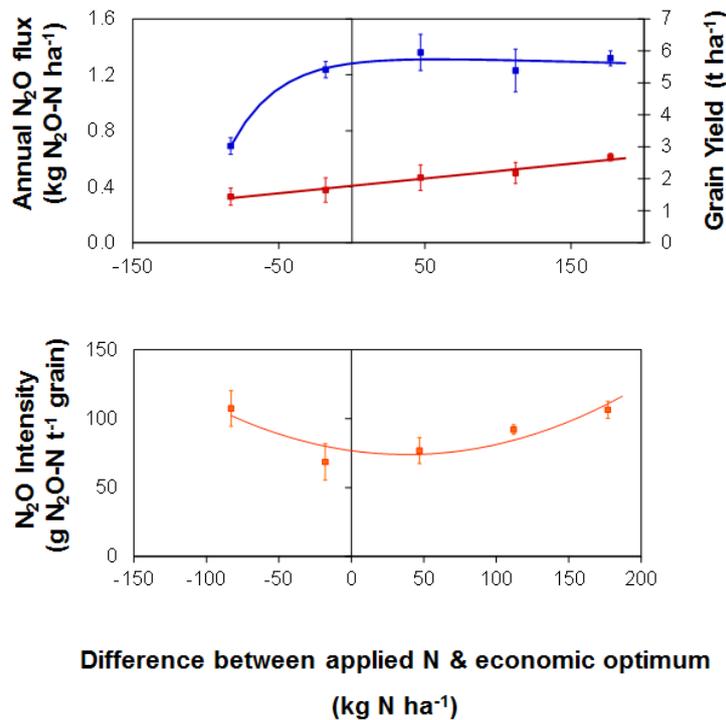


Figure 21. Example 2, 2010, spring barley, Gleadthorpe, mean grain yield ( $\text{t ha}^{-1}$ ), cumulative annual  $\text{N}_2\text{O}$  flux ( $\text{kg N}_2\text{O-N ha}^{-1}$ ) (top) and  $\text{N}_2\text{O}$  intensity ( $\text{g N}_2\text{O-N t}^{-1} \text{grain}$ ) (below) against the difference between applied N and  $\text{N}_{\text{opt}}$  ( $83 \text{ kg N ha}^{-1}$ ). Error bars represent  $2 \times \text{SE}$  of the mean. Fitted values represent the lines of best fit ( $P < 0.05$ ).

**Example 3, Curved per hectare, Curved 'U-shape' per tonne: 2011, winter barley, Gleadthorpe**

The response of annual N<sub>2</sub>O emission per hectare to fertiliser N rate in Example 3 demonstrated a non-linear relationship (Figure 24). As with the first example, there was an N surplus at harvest from the 160% treatment of c.155 kg N ha<sup>-1</sup>, coupled with a poor fertiliser N recovery (c.40%), although from the 40% and 80% treatments the fertiliser N recovery was very good at c.75%. Around harvest, in late July, mineral N (c.55 kg N ha<sup>-1</sup>) remained in the top 10 cm of soil from the highest N rate treatment (Figure 22). Elevated soil (0-10 cm) NO<sub>3</sub>-N values from the top N rate continued into November with a small peak (c.20 kg NO<sub>3</sub>-N ha<sup>-1</sup>) stimulated by soil cultivation. Although the measured peak N<sub>2</sub>O emissions was not large (up to c.20 g N<sub>2</sub>O-N ha<sup>-1</sup> d<sup>-1</sup>) the main emission period was clearly measured around harvest and during the following 6 months (Figure 23).

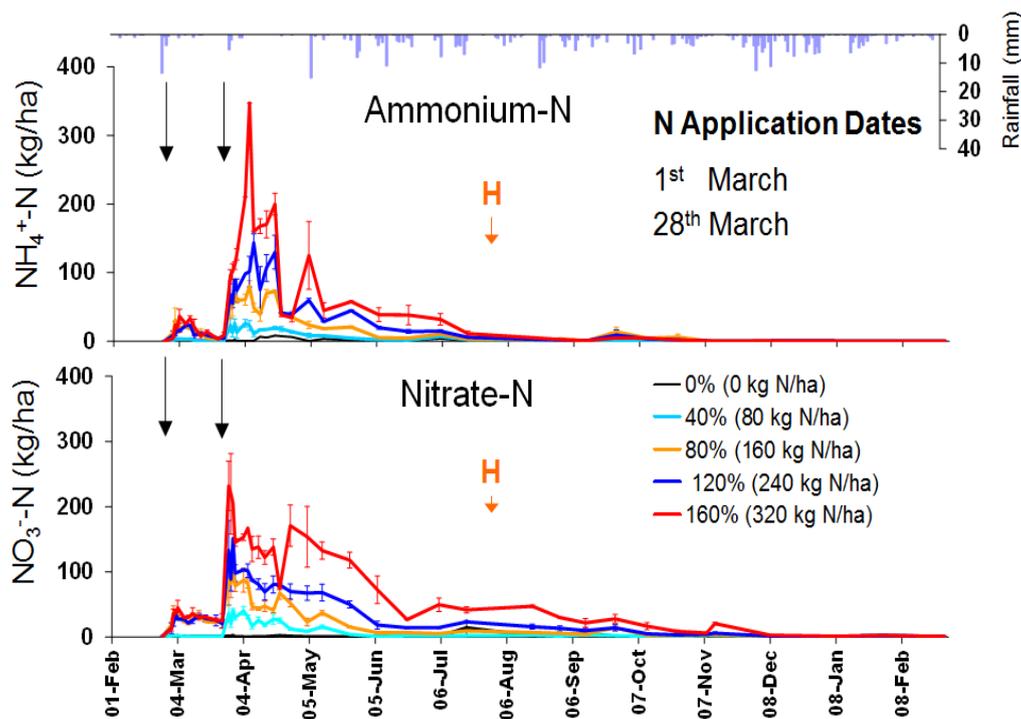
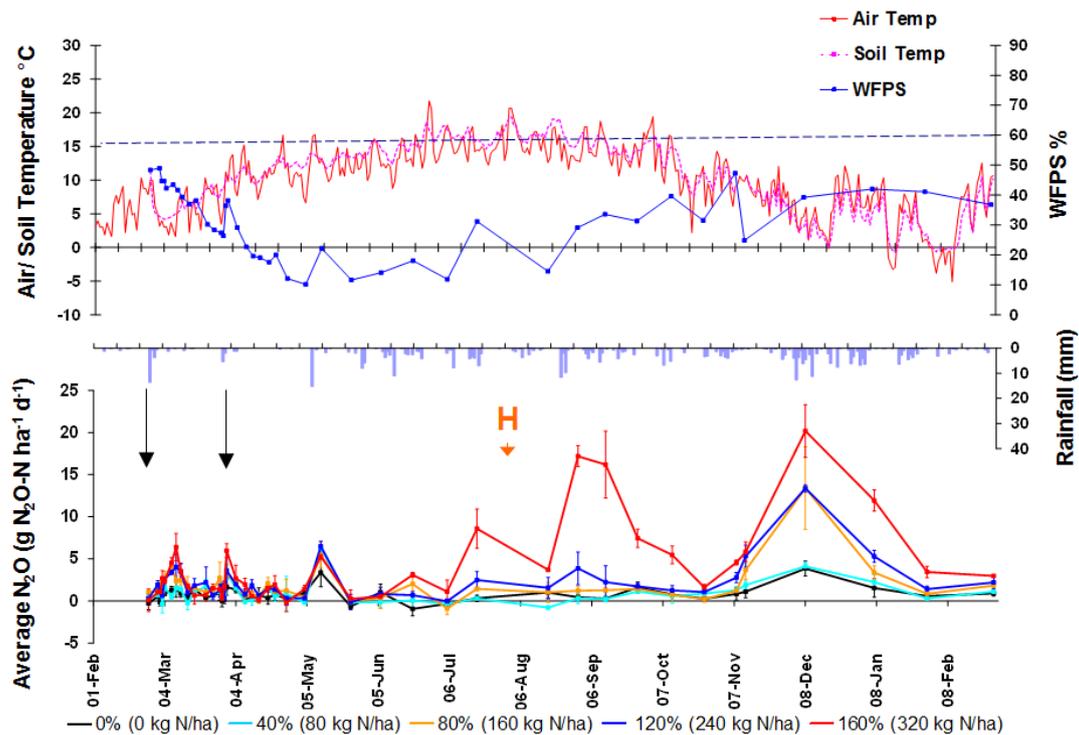


Figure 22. Example 3, 2011, winter barley, Gleadthorpe, soil mineral N (0-10 cm) presented as mean ammonium-N (NH<sub>4</sub><sup>+</sup>-N) (top) and nitrate-N (NO<sub>3</sub><sup>-</sup>-N) (below), error bars represent 2 x SE of the mean. Arrows denote fertiliser application date and 'H' date of winter barley harvest. Bars represent total daily rainfall (mm).



**Figure 23. Example 3, 2011, winter barley, Gleadthorpe, mean daily air and soil temperature ( $^{\circ}\text{C}$ ), percentage water filled pore space (WFPS) (Top) and mean daily  $\text{N}_2\text{O}$  emissions ( $\text{g N}_2\text{O-N ha}^{-1} \text{d}^{-1}$ ) (below). Error bars represent  $2 \times \text{SE}$  of the mean. Arrows denote N application date and 'H' date of winter barley harvest. Bars represent total daily rainfall (mm). Dotted line represents 60% WFPS.**

The first of the two fertiliser applications was applied to a moist (c.50% WFPS) but relatively cold soil ( $<5^{\circ}\text{C}$ ), which would have inhibited  $\text{N}_2\text{O}$  loss. Subsequently (including the period after the second fertiliser application), although the temperature rose, rainfall was extremely low (Table 20) such that the soil moisture steadily declined to a minimum WFPS of c.10% and emissions of  $\text{N}_2\text{O}$  were constrained by dry soil (Figure 23). Due to its light, sandy loam texture, even when the soil was very wet in late autumn/winter, its WFPS never exceeded 50%. The later emissions of  $\text{N}_2\text{O}$  were associated with rainfall events and despite soil mineral N contents (0-10 cm) of  $<5 \text{ kg N ha}^{-1}$ , the December and early January  $\text{N}_2\text{O}$  emission peaks were associated with considerable December rainfall, which in contrast to the rainfall measured in March to November was greater than the LTA (Table 20). Since there was minimal mineral N measured in the topsoil, it would seem likely that these  $\text{N}_2\text{O}$  emissions originated from deeper in the soil, the porous nature of the soil enabling released  $\text{N}_2\text{O}$  to diffuse to the soil surface.

$Y_{\text{opt}}$  for winter barley was  $6.2 \text{ t ha}^{-1}$  ( $N_{\text{opt}}$ :  $157 \text{ kg ha}^{-1}$ ) and yield declined at higher N rates (Figure 24). When the annual  $\text{N}_2\text{O}$  emissions were expressed on a yield basis, the line of best fit was a non-linear 'U-shape'.

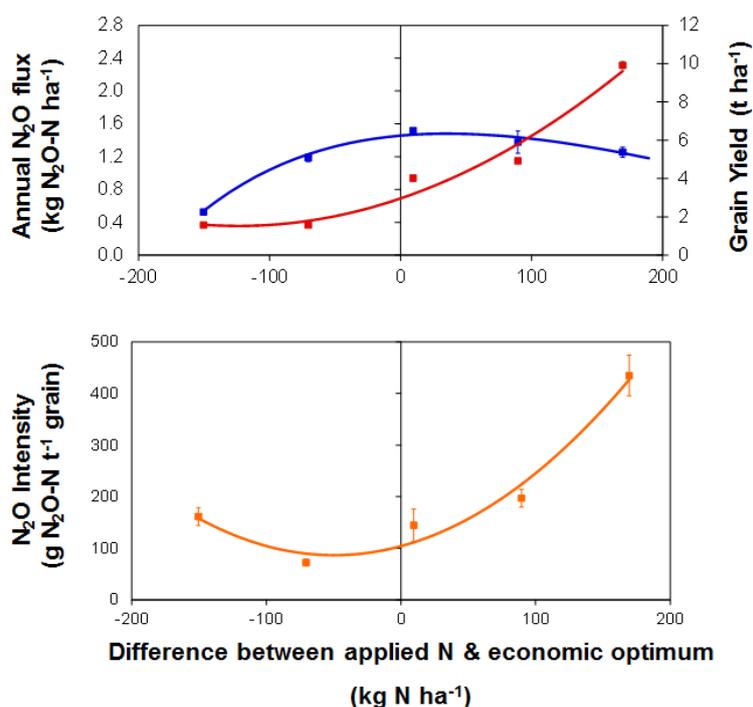


Figure 24. Example 3, 2011, winter barley, Gleadthorpe, mean grain yield (t ha<sup>-1</sup>), cumulative annual N<sub>2</sub>O flux (kg N<sub>2</sub>O-N ha<sup>-1</sup>) (top) and N<sub>2</sub>O intensity (g N<sub>2</sub>O-N t<sup>-1</sup> grain) (below) against the difference between applied N and N<sub>opt</sub> (157 kg N ha<sup>-1</sup>). Error bars represent 2 x SE of the mean. Fitted values represent the lines of best fit (P <0.05).

**Example 4, Curved per hectare, Curved 'increasing quadratic' per tonne: 2012, winter barley, Terrington**

As in Example 3, the response of annual N<sub>2</sub>O emission per hectare to fertiliser N rate in Example 4 demonstrated a non-linear (quadratic increasing) relationship (Figure 27); however the timing of the main peak emission was not as hypothesised to give such a relationship. There were large effects of fertiliser N, particularly following the second N application, on SMN in the topsoil (0-10cm). However, NH<sub>4</sub>-N had all but disappeared by the end of May and NO<sub>3</sub>-N in the topsoil remained relatively low (< c.30 kg N ha<sup>-1</sup>) for the rest of the summer, and indistinguishable from the levels measured in the 0N treatment for the rest of the 12 months after N application (Figure 25).

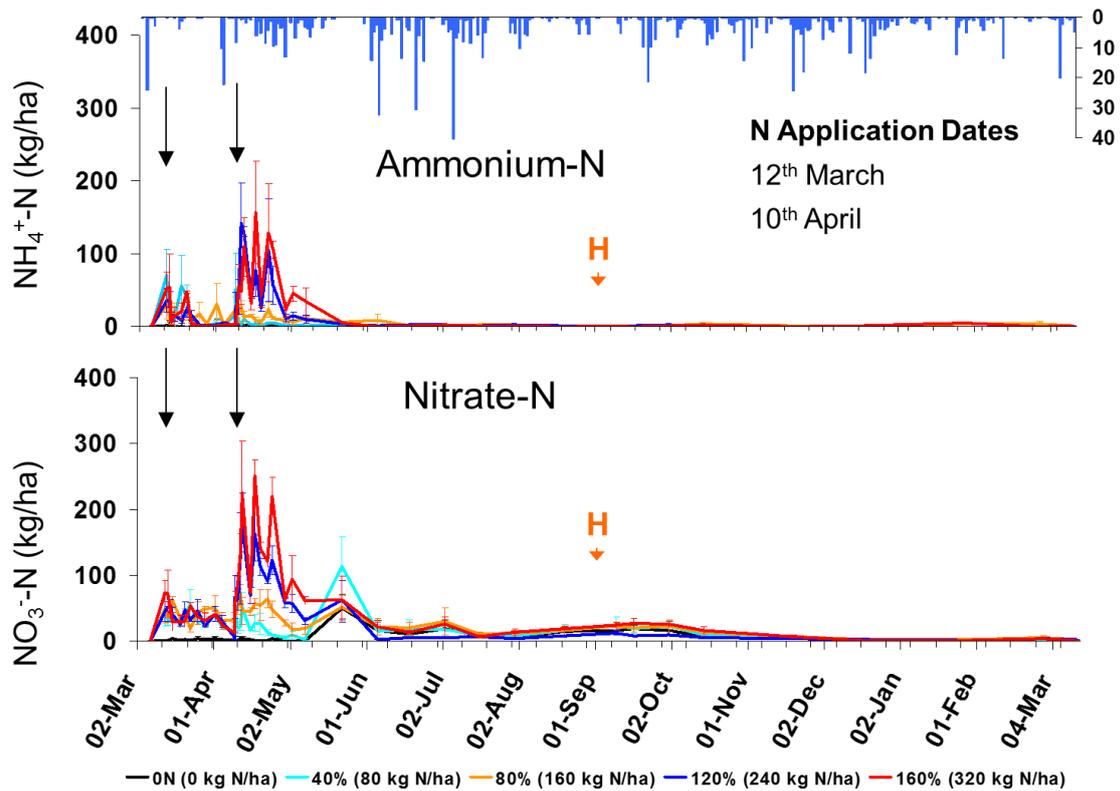


Figure 25. Example 4, winter barley, 2012, Terrington, soil mineral N (0-10 cm) presented as mean ammonium-N ( $\text{NH}_4^+\text{-N}$ ) (top) and nitrate-N ( $\text{NO}_3^-\text{-N}$ ) (below), error bars represent 2 x SE of the mean. Arrows denote fertiliser application date and 'H' date of winter barley harvest. Blue hanging bars represent total daily rainfall (mm).

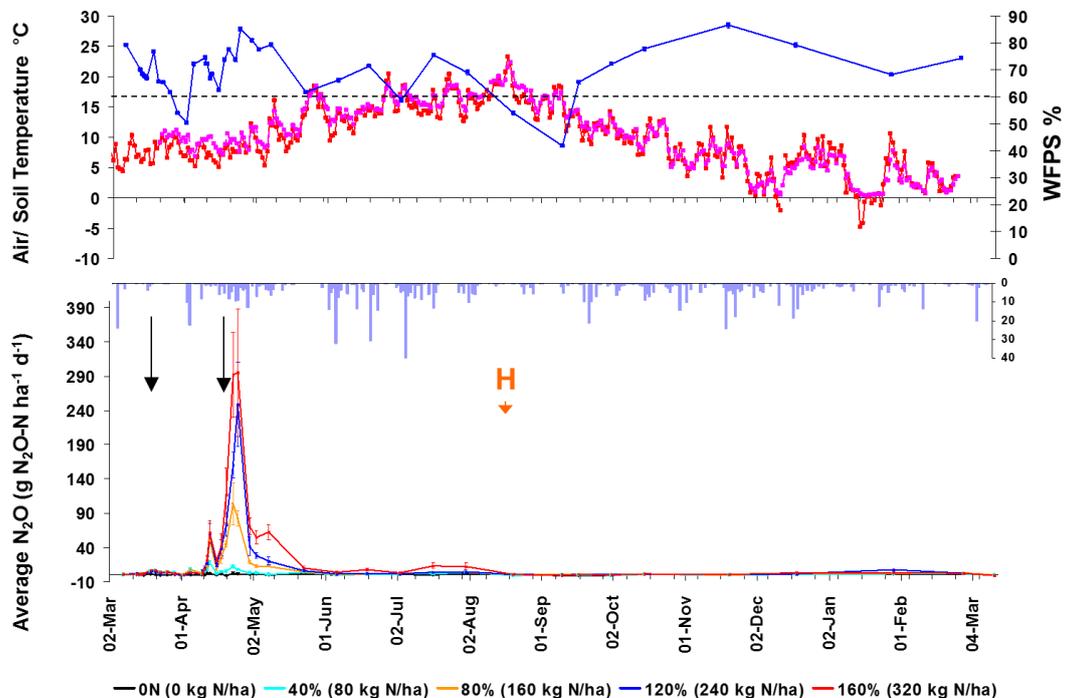
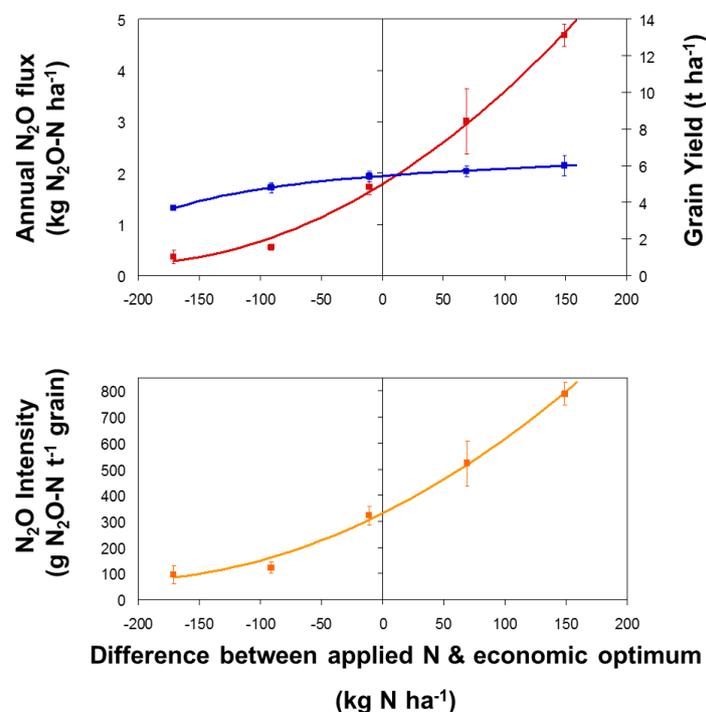


Figure 26. Example 4, winter barley, 2012, Terrington, mean daily air and soil temperature ( $^{\circ}\text{C}$ ), percentage water filled pore space (WFPS) (Top) and mean daily  $\text{N}_2\text{O}$  emissions ( $\text{g N}_2\text{O-N ha}^{-1} \text{d}^{-1}$ ) (below). Error bars represent 2 x SE of the mean. Arrows denote N application date and 'H' date of winter barley harvest. Bars represent total daily rainfall (mm). Dotted line represents 60% WFPS.

There were very large treatment-related N<sub>2</sub>O emissions in the 30 days after the second fertiliser application in mid-April (Figure 26). The huge emission peak (up to c.300 g N<sub>2</sub>O-N ha<sup>-1</sup> d<sup>-1</sup>) coincided with heavy rainfall (Figure 26 and Table 20), causing the WFPS to remain >70%, and soil temperatures were approaching 10°C (Figure 26). Although the topsoil remained wet (generally >60% WFPS) for much of the subsequent summer and the soil warmed to almost 20°C, N<sub>2</sub>O emissions were small (<15 g N<sub>2</sub>O-N ha<sup>-1</sup> d<sup>-1</sup>) (Figure 26). This was presumably due to the relatively low concentrations of SMN, which dramatically decreased in late April immediately following the huge peak N<sub>2</sub>O emission (Figure 25). It is likely that at this time in the exceptionally wet conditions, the N<sub>2</sub>O was predominantly produced during denitrification, which may also have generated large N losses as N<sub>2</sub>. Measurements of N<sub>2</sub> losses are technically challenging, but a number of studies have shown that losses can be equal to or greater than those of N<sub>2</sub>O depending on soil conditions (Arah *et al.* 1993; Cardenas *et al.* 2003).

After poor establishment and in the wet and dull summer conditions of this experiment the winter barley crop was low yielding, and did not respond well to fertiliser, resulting in an N surplus at harvest from the 160% treatment of c.175 kg N ha<sup>-1</sup>. The fertiliser recovery was poor from all treatments at c.25% and the N<sub>opt</sub> of 171 kg ha<sup>-1</sup> was used very inefficiently, giving a grain yield of only 5.4 t ha<sup>-1</sup> (Figure 27). When the annual N<sub>2</sub>O emissions were expressed on a yield basis (emission intensity), the line of best fit was a non-linear (increasing quadratic) response, which is similar to the response in emissions per hectare.



**Figure 27. Example 4, winter barley, 2012, Terrington, mean grain yield (t ha<sup>-1</sup>), cumulative annual N<sub>2</sub>O flux (kg N<sub>2</sub>O-N ha<sup>-1</sup>) (top) and N<sub>2</sub>O intensity (g N<sub>2</sub>O-N t<sup>-1</sup> grain) (below) against the difference between applied N and N<sub>opt</sub> (171 kg N ha<sup>-1</sup>). Error bars represent 2 x SE of the mean. Fitted values represent the lines of best fit (P <0.05).**

## 6.6. Discussion of fertiliser effects on N<sub>2</sub>O emissions

Our prime concern here is the comparison of EFs from these experiments with the default EFs devised by the IPCC and with previous UK work. The four example experiments will be used to illustrate the main factors which influenced the responses of annual N<sub>2</sub>O emission (kg ha<sup>-1</sup>) to fertiliser N rate and influence the shape of the emission intensity curves (g N<sub>2</sub>O–N t<sup>-1</sup>).

### 6.6.1. Fertiliser induced emission factors

The IPCC Tier 1 methodology for calculating *direct* N<sub>2</sub>O emission from soil following the application of manufactured N fertiliser adopts an EF of 1% of total-N applied (EF<sub>1</sub>; IPCC, 2006). This default is greater than the overall average annual EF of 0.67% from all the experiments conducted here, but similar to the mean EF for wetter conditions in Scotland and all sites in 2012 (Table 23). Previously reported EFs following the application of AN fertiliser to winter cereals in the UK have rarely been greater than, or even close to, the IPCC 2006 default (Table 29), so the relatively low EFs measured in this study were not unexpected, but at the English sites in 2010 and 2011 EFs were particularly affected by the dry spring conditions when fertiliser was being applied (Table 20).

For the English sites, the range of fertiliser induced annual EFs from the 120% treatment ranged from 0.10 to 0.64% (or 0.46% if the higher EF from sugar beet, Terrington, 2010 is excluded) in 2010, 0.07 to 0.52% in 2011 and 0.65 to 1.25% in 2012. The EFs from these first two years were of a similar magnitude to that reported from a separate experiment also carried out in 2010 and in Area 2 of this study (Gleadthorpe, central England), where an annual EF of 0.13% total-N applied was measured from a winter wheat crop following the application at the recommended N rate of ammonium nitrate fertiliser (Smith *et al.*, 2013). For the Scottish sites, the range of fertiliser induced annual EFs from the 120% treatment tended to be greater here at 0.68 to 1.0% in 2010, 0.06 to 2.82% in 2011 and 0.25 to 1.09% in 2012.

Table 29. Previous estimates of N<sub>2</sub>O emissions (% total-N applied) from UK arable crop production following application of ammonium nitrate fertiliser (a) corrected, and (b) uncorrected for background emission.

Site	Year N applied	Crop type	N rate (kg ha <sup>-1</sup> )	N <sub>2</sub> O emission (% N applied)	Measurement period (months)	Reference
<b>(a) Corrected for background emissions</b>						
Central England	2010	Winter wheat	160	0.13	12	Smith <i>et al.</i> , 2013
Central England	2008	Winter wheat	210	0.44	12	Defra report - AC0101, 2010
Central England	2007	Winter barley	210	1.09	12	Defra report - AC0101, 2010
East England	2008	Winter wheat	210	0.63	12	Defra report - AC0101, 2010
East England	2007	Winter wheat	210	0.81	12	Defra report - AC0101, 2010
East England	2004	Winter wheat	220	0.45	12	Smith <i>et al.</i> , 2012
East England	2005	Winter wheat	160	0.23	c.6	Smith <i>et al.</i> , 2012
South-west England	2008	Winter wheat	210	3.17	12	Defra report - AC0101, 2010
Central Scotland	2005	Winter wheat	220	1.72	c.6	Smith <i>et al.</i> , 2012
Central Scotland	2003	Winter wheat	180	0.21	c.3.5 (April-August)	Smith <i>et al.</i> , 2012
<b>(b) Uncorrected for background emissions</b>						
Central England	1996	Winter wheat	190	0.6	Drilling to harvest	Webb <i>et al.</i> , 2004
Central England	1996	Winter wheat	190	0.6	Drilling to harvest	Webb <i>et al.</i> , 2004
Central England	1999	Winter wheat	180	0.6	Drilling to harvest	Webb <i>et al.</i> , 2004
Central England	2000	Winter wheat	180	0.2	Drilling to harvest	Webb <i>et al.</i> , 2004
Central England	2000	Winter barley	140	0.6	Drilling to harvest	Webb <i>et al.</i> , 2004
East England	1996	Winter wheat	180	0.6	Drilling to harvest	Webb <i>et al.</i> , 2004
East England	1997	Winter wheat	160	0.7	Drilling to harvest	Webb <i>et al.</i> , 2004
East England	1999	Winter wheat	150	0.7	Drilling to harvest	Webb <i>et al.</i> , 2004
East England	1999	Winter wheat	180	1.0	Drilling to harvest	Webb <i>et al.</i> , 2004
East England	1999	Winter wheat	190	0.7	12	Dobbie and Smith, 2003
East England	2000	Winter wheat	190	0.6	Drilling to harvest	Webb <i>et al.</i> , 2004
East England	2000	Winter wheat	230	0.4	Drilling to harvest	Webb <i>et al.</i> , 2004
East England	2001	Winter barley	132	0.5	12#	Dobbie and Smith, 2003
Central Scotland	1996	Winter wheat	200	0.4	>12	Dobbie <i>et al.</i> , 1999
Central Scotland	1997	Winter wheat	200	0.5	>12	Dobbie <i>et al.</i> , 1999
Central Scotland	1994	Winter wheat	180	0.7	c.8	Smith <i>et al.</i> , 1998b
Central Scotland	1994	Spring barley	120	0.2	c.8	Smith <i>et al.</i> , 1998b
Central Scotland	1998	OSR	120	1.0	c.4 (May-August)	Dobbie <i>et al.</i> , 1999

# N<sub>2</sub>O measured only during the growing season; winter fluxes estimated to give an estimated annual flux.

Previously reported annual EFs measured following the split application of ammonium nitrate fertiliser to cereal crops in the UK are shown in Table 29. The mean EF from the 10 experiments (corrected for background emissions) was 0.89% total-N applied (range 0.13-3.17%) (Table 29), which was greater than the overall mean EF of 0.67% from this study, but closer to our mean Scottish EF of 1% total N applied. Excluding EFs measured from the typically 'wetter' UK locations of south-west England and central Scotland, the mean EF using data from the remaining 'drier' areas of central and east England was 0.54% total-N applied (Table 29) and similar to the mean English EF of 0.48% total N applied measured in this study. Reported UK EFs, uncorrected for background emissions, are also shown in Table 29. Even though these were overestimated as they also included the 0N (background) N<sub>2</sub>O emission as well as any emission induced by fertiliser N application, the mean EF of 0.59% of total-N applied (range 0.2-1.0%) was still less than our overall mean EF and less than the IPCC default.

### **6.6.2. Background emissions**

The wide range in measured background emissions of 0.01 to 6.78 kg N ha<sup>-1</sup> in this study (Table 22 and Table 24) tends to reflect previous research. In a previous UK study (Defra, 2010b), background emissions from winter cereals also ranged from 0.45 to 3.25 kg N ha<sup>-1</sup>, and in an earlier study background fluxes from winter wheat at ADAS Boxworth (Cambridgeshire), ADAS Terrington (Norfolk) and SRUC Bush Estate (Edinburgh, Scotland) were measured at 0.49, 1.52 and 0.38 kg N ha<sup>-1</sup> respectively (Smith *et al.*, 2012). Smith *et al.* (2012) concluded that, due to the substantial variation in background emissions, their measurement was a crucial part of quantifying correctly N<sub>2</sub>O emissions arising from applied N. This view was also supported by van Groenigen *et al.* (2004). Moreover, following measurement of a background emission of 1.52 kg N ha<sup>-1</sup> from a silage maize crop, which corresponded to c.50% of the emission from the N fertilised plots, van Groenigen *et al.* (2004) proposed accounting for background emissions in N<sub>2</sub>O budgets. It is clear that in general the background emissions measured in our study from England (Boxworth, Gleadthorpe and Terrington areas) were at the lower end of the range previously reported, whereas those from Scotland (Bush Estate) were at the higher end of the range. Background emissions are likely to be influenced by multiple sources and affected by interactions between several factors including cropping history, soil pH, soil N and carbon content, soil moisture, atmospheric N deposition, etc. In this study, it would seem likely that an interaction between these factors was often responsible for the larger emissions, but also that the size of the background emissions was strongly related to soil moisture hence rainfall as well as soil type. In England annual rainfall totals increased in the order 2011, 2010 and 2012 in line with increasing background emissions. Similarly in Scotland annual rainfall totals and background emissions both increased in the order 2010, 2011 and 2012. Annual rainfall totals in Scotland (807-1,775 mm) were also considerably greater than those in England (379-897 mm; Table 20), probably affecting the size of the large Scottish background N<sub>2</sub>O emissions. Chen *et al.* (2008) reported a relatively

high background N<sub>2</sub>O emission (4.01 kg N ha<sup>-1</sup>) from winter wheat production in China, which they attributed to high rainfall (1,100 mm). In addition, Zhang and Han (2008) found that irrigation increased background emissions from c.0.7 kg N ha<sup>-1</sup> to c.1 kg N ha<sup>-1</sup>. It is clear, however, that site specific conditions (e.g. rainfall pattern and soil conditions) control the magnitude of the background emission. It is probable that the high background emissions in Scotland may also have been influenced by the high topsoil carbon content (c.3-7%) which is likely to have been associated with a high mineralisation potential, hence increased background N<sub>2</sub>O emissions.

### **6.6.3. Effect of N fertiliser application rate**

In all twenty-four experiments, the annual emissions of N<sub>2</sub>O were strongly affected by the amount of N fertiliser applied. Both linear and non-linear (increasing and decreasing quadratic) responses of N<sub>2</sub>O emission to N fertiliser application rate were observed. There was no consistent pattern between sites or crop types, but possibly there were more increasing quadratic responses in Scotland and in England during the wettest year, 2012.

Work from outside the UK has also demonstrated a mix of non-linear and linear responses from both arable crops and grassland (Velthof *et al.*, 1997; Schils *et al.*, 2008). As in the data presented here McSwiney and Robertson (2005) and Hoben *et al.* (2011) both reported linear and non-linear responses following urea applications at six N rates (0-291 kg N ha<sup>-1</sup>) in experiments on maize carried out over several seasons in the USA. Other non-linear responses have also been reported for maize and spring barley (e.g. Zebrath *et al.*, 2008; Ma *et al.*, 2010). Van Groenigen *et al.* (2004) also reported non-linear responses to single doses of N (calcium ammonium nitrate) applied to silage maize at 0 to 188 kg N ha<sup>-1</sup> (0 to 125% of the recommended N rate). A recent global meta-analysis has reported that N<sub>2</sub>O emissions from crops respond in a non-linear (increasing) pattern to increasing N-fertiliser application rates. This was particularly apparent in soils with more than 1.5% organic carbon, a pH of less than 7 and where the fertiliser had been applied in a single dose (Shcherbak *et al.*, 2014). However, at a commercial N application rate for most UK crops, e.g. 200 kg ha<sup>-1</sup>, the difference between using this non-linear (increasing) model and assuming the IPCC linear relationship was negligible (Shcherbak *et al.*, 2014).

In contrast, Chen *et al.* (2008), reported a linear relationship between N<sub>2</sub>O emission and N (urea) application rate (0-300 kg ha<sup>-1</sup>) from winter wheat grown in China. In a separate Chinese study (Zhang and Han, 2008) a linear relationship between N<sub>2</sub>O emissions and N rate was also measured following the application of urea fertiliser N (0-150 kg ha<sup>-1</sup>) to abandoned cropland and grassland. Linear responses were also reported in three other Chinese studies: one from a winter wheat field and two from a winter wheat-summer maize double-cropping system (Liu *et al.*, 2011; Ji *et al.*, 2012; Qin *et al.*, 2012).

Overall, it is apparent that the relationship between N<sub>2</sub>O emission and N fertiliser rate is not always as indicated by our simple hypothesis (Figure 1a). We therefore examined whether the variation was associated with the time of peak emission. At both the English and Scottish sites in 2012 most emissions occurred soon after fertiliser application, which could cause N<sub>2</sub>O emissions expressed on an area basis to reflect the fertiliser application rate and show a linear response to N rate. Conversely at both the English and Scottish sites in 2010, at the English sites in 2011 and to a lesser extent at the Scottish site in 2011, dry spring soil conditions inhibited N<sub>2</sub>O emissions soon after N fertiliser application so that most peak N<sub>2</sub>O emissions occurred during or after the main phase of crop N uptake. In this case, one might expect that N<sub>2</sub>O emissions expressed on an area basis would show a curved, non-linear response to N rate. However, at the English sites in 2010 and 2011 there were few curved, non-linear responses as illustrated by Example 1. Dry soil conditions not only delayed and reduced N<sub>2</sub>O emissions substantially, but they gave rise to delayed and poor crop N uptake by the winter wheat, which resulted in considerable amounts of mineral N present in the topsoil (0-10 cm) even after harvest in mid-August. From the analysis of the responses of both SMN and N<sub>2</sub>O emissions to N rate it was apparent that, as a result of the dry conditions, the small daily emissions were not sufficiently large to drive the linear shape of the final N<sub>2</sub>O emission response, which was largely determined by small variations in the daily N<sub>2</sub>O data. This was despite initial distinct linear responses and, later in the summer after limited crop N uptake, non-linear (increasing quadratic) responses of SMN to N rate. Even the rainfall induced peak emission measured after the 3<sup>rd</sup> fertiliser application split (Figure 17) was too small (only c.10 g N<sub>2</sub>O-N ha<sup>-1</sup> d<sup>-1</sup>) to confer a non-linear (increasing quadratic) response on the daily N<sub>2</sub>O emissions; the daily N<sub>2</sub>O response was best described with a linear fit.

McSwiney and Robertson (2005) suggested that although it is the pattern of soil mineral N availability that drives the response of N<sub>2</sub>O emissions to fertiliser N application rate, it is only when there is available N *and* enough soil moisture present that large emissions will occur. Furthermore, Hoben *et al.* (2011) also suggested that the size of the N<sub>2</sub>O emission may be a factor in the shape of the response to N rate: a non-linear response may not occur when N<sub>2</sub>O emissions are low or indeed it may not even be measurable. It is possible therefore, in Example 1, that if soil conditions had been favourable (i.e. warm and wet) when there was still substantial SMN available, large non-linear N<sub>2</sub>O emissions would have occurred.

Numerous studies in the literature have shown that N<sub>2</sub>O production increases with temperature and with a rise in soil moisture and vice-versa (Dobbie *et al.*, 1999; Dobbie and Smith, 2003; Smith *et al.*, 2012). The peak N<sub>2</sub>O emissions which were measured in Example 1 were very small due to the lack of rainfall and predominantly dry soils; it was these conditions coupled with cooler temperatures later in the year that also prevented significant emissions from the surplus N available after harvest. Indeed with a WFPS that was substantially lower than the 'threshold' of

60%, substrate availability and bacterial activity would have decreased (Davidson, 1993) and consequently N<sub>2</sub>O emissions would have been reduced.

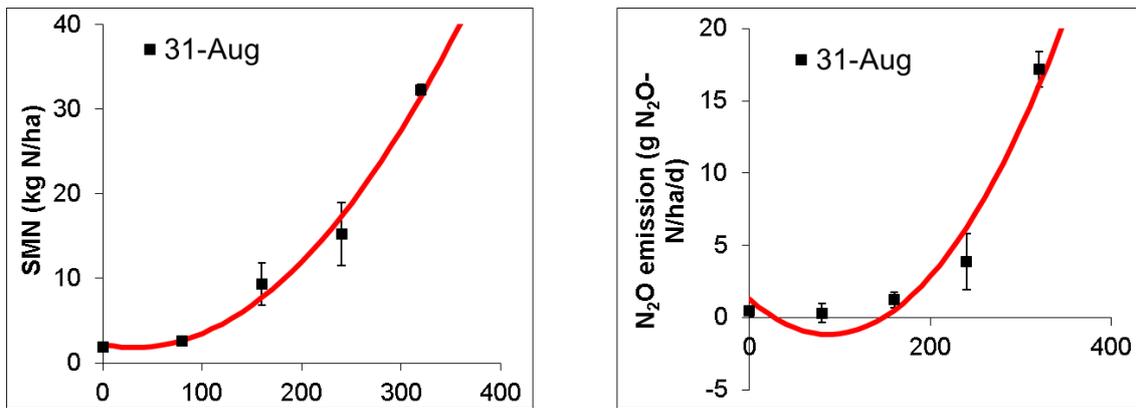
In contrast to Example 1, the linear response of N<sub>2</sub>O emission per hectare to fertiliser N rate seen in Example 2 was coupled with a negligible surplus of N at harvest and a high fertiliser N recovery from all N rates. Soil mineral N data and crop N uptake data (not shown) confirmed that by early summer there was little N in the top soil. Consequently, the N balance between fertiliser N supply and crop N uptake may still have been important in this spring barley experiment, in the fact that there was very little surplus N available as a substrate for use in N<sub>2</sub>O production. However, as with Example 1, despite a non-linear (increasing quadratic) response of SMN to N rate at the time of the largest emission after the 2<sup>nd</sup> split fertiliser application, the emission was so low (c.5 g N<sub>2</sub>O-N ha<sup>-1</sup> d<sup>-1</sup>) that it did not reflect the SMN response. Once again it is probable that the final linear response of N<sub>2</sub>O emission to N rate was the result of small variations in the very small daily N<sub>2</sub>O emissions rather than from a distinct emission event. During the period immediately following the main fertiliser application in late April, even though significant SMN levels were measured, conditions were too dry to induce significant emissions of N<sub>2</sub>O. At this time, the sandy loam texture combined with the lack of rainfall restricted the soil WFPS to 30% or less, indicating that soil nitrification was likely to be the dominant process generating the small amount of N<sub>2</sub>O produced (Davidson, 1993). Indeed, according to the model proposed by Davidson (1991) which describes the relationship between WFPS and the relative emissions of N gases, the production of N<sub>2</sub>O from denitrification would not occur unless the WFPS was >55%. Below this level nitrification would be the dominant source (Davidson, 1991). Bateman and Baggs (2005) showed that even with a heavier textured soil (silt loam) than in Example 2, with a WFPS of 35-60% the main N<sub>2</sub>O production process was nitrification. As it is generally considered that denitrification results in larger emissions of N<sub>2</sub>O than are produced from nitrification (Dobbie and Smith, 2001; Dobbie and Smith, 2003; Smith *et al.*, 2012), the soil would have had to be substantially wetter for large N<sub>2</sub>O emissions to have occurred after fertiliser application.

It is probable, therefore, that the predominance of linear responses of N<sub>2</sub>O emission to N rate at the English sites was the result of dry soil conditions leading to small N<sub>2</sub>O emissions, and despite frequent non-linear (increasing quadratic) responses of SMN to N rate, these were not translated to the response in N<sub>2</sub>O emissions. Conversely, the Scottish site in 2010, despite having a dry spring, had such heavy rainfall in July (Table 20) that daily peak emissions were large enough (>200 g N<sub>2</sub>O-N ha<sup>-1</sup> d<sup>-1</sup> from the winter and spring barley crops and >100 g N<sub>2</sub>O-N ha<sup>-1</sup> d<sup>-1</sup> from the winter wheat crop) when integrated over a long enough time (c.1 month), to dominate the shape of the final N<sub>2</sub>O emission response. Comparable conditions also prevailed at the Scottish site in 2011 and although it was only the month of April which was dry (Table 20), this was sufficient to delay the large (up to c.425 g N<sub>2</sub>O-N ha<sup>-1</sup> d<sup>-1</sup>) peak emissions until early summer (OSR and winter wheat

crops). The wetter soil conditions and higher soil organic matter (SOM) concentrations in Scotland led to particularly high emissions during the summer of 2011. These were exacerbated in circumstances where patchy crop growth encouraged N loss not used by the crop.

Under conditions of extreme wetness it is unlikely that high N<sub>2</sub>O emissions could be avoided. In Scotland in 2012 the annual average rainfall exceeded the LTA by 81%. This resulted in high emissions from both the fertilised and unfertilised treatments. The high emissions associated with the control plots suggests an increase in mineralisation rates during this period associated with the warm and moist soil conditions. The importance of soil wetness in promoting the linkage between mineralisation and N<sub>2</sub>O emissions has been previously reported and linked to conditions where there is a cyclic wetting and drying of the soil (Guo *et al.*, 2014).

The low crop N recoveries shown in Example 1 are not typical of UK crop performance; average fertiliser N recoveries are of the order of 60% and they can exceed 80% (Bloom *et al.*, 1988; Sylvester-Bradley *et al.*, 2008). Indeed, the non-linearity displayed by the response of N<sub>2</sub>O emission to fertiliser N rate in the winter barley crop of Example 3 was probably related to the balance between fertiliser N supply and crop N uptake as assumed by our hypothesis. Following the 2<sup>nd</sup> and final fertiliser N application, the non-linear (increasing quadratic) SMN response to N rate closely mirrored the actual non-linear fertiliser N application pattern whereby the application on the 40% treatment was proportionally smaller than at the higher N rates. This combined with later crop N uptake, particular from the 40% and 80% treatments and to a lesser extent the 120% treatments resulted in a distinctly non-linear (increasing quadratic) response of SMN to N rate at the time of the main N<sub>2</sub>O emissions after harvest in late July (e.g. Figure 23). The crop N uptake data (not shown) supported by SMN indeed indicated that substrate N for N<sub>2</sub>O generation was available in excess, particularly at the top N rate (160% treatment). The extremely dry and/or cold soil in the spring following the N fertiliser applications prohibited significant N<sub>2</sub>O emissions and it was only later, around and after harvest, that rainfall gave conditions favourable for N<sub>2</sub>O production, and because of the greater residual soil nitrate at this time, this especially affected the top N rate (Figure 28). Similarly, in a grassland study, Velthof *et al.* (1997) reported that N<sub>2</sub>O emissions increased after rainfall, particularly from the higher fertiliser N rates. As with the SMN response, the response of daily N<sub>2</sub>O emission to N rate during the main emission period was non-linear (increasing quadratic) and, although the N<sub>2</sub>O emissions were not particularly large, they occurred probably close enough to the end of the 12 month measurement period to drive the shape of the final non-linear (increasing quadratic) N<sub>2</sub>O emission response (Figure 24).



**Figure 28. The response of topsoil total mineral nitrogen (SMN) and daily N<sub>2</sub>O flux to applied N for winter barley in 2011 at Gleadthorpe on the 31<sup>st</sup> August 2011.**

In both England and Scotland in 2012 most N<sub>2</sub>O emissions occurred soon after fertiliser application, suggesting that N<sub>2</sub>O emissions expressed on an area basis might be more likely to reflect the fertiliser N application rate and show a linear response to N rate in the majority of experiments. Indeed, generally the N<sub>2</sub>O emissions did mirror the fertiliser application rate, but because of the artefact of a non-linear fertiliser N application pattern at the last fertiliser split application, the response to N rate was generally non-linear. Example 4 typifies the response of N<sub>2</sub>O emission to N rate. The timing of heavy and sustained rainfall shortly after the final split fertiliser application induced the extremely large N<sub>2</sub>O emissions. The response of SMN at this time was non-linear reflecting the proportionally smaller application of N on the 40% treatment compared to the higher N rates. In the very wet conditions, the non-linearity of the SMN response translated into the N<sub>2</sub>O response. Because the emissions were so large (up to c.300 g N<sub>2</sub>O-N ha<sup>-1</sup> d<sup>-1</sup>) and sustained over approximately 2 weeks the cumulative N<sub>2</sub>O response 'switched' to non-linear (increasing quadratic) and remained as such so that the final response of cumulative N<sub>2</sub>O was non-linear (increasing quadratic) (Figure 27).

Our data indicate that the responses of annual N<sub>2</sub>O emissions per hectare to fertiliser N rates probably reflect the balance between fertiliser N supply and crop N uptake, as well as the amount and timing of rainfall in relation to each fertiliser application split. However, with wet soils in spring (at and following N application), responses of N<sub>2</sub>O emission per hectare to fertiliser N rate were more rather than less non-linear (increasing quadratic), even though N<sub>2</sub>O emissions were large and occurred almost straight away. These non-linear responses are probably an artefact the way that the total spring N application was split between application dates in these experiments, rather than a reflection of biological processes. However, a non-linear response would be consistent with the results of a recent meta-analysis of available global data where only 15% of 26 published datasets exhibited a linear relationship of *direct* N<sub>2</sub>O emission per hectare to fertiliser N input (Kim *et al.*, 2013b). It might be worth examining whether experiments used in this meta-analysis also had confounded differences between rate and timing of applied fertiliser N.

Given the very contrasting moisture conditions in which our experiments were conducted, it will be important to consider how N<sub>2</sub>O emissions in more typical rainfall levels and distributions, such as will occur in other regions and seasons, might be affected (see subsequent sections of this report).

#### **6.6.4. Nitrous oxide emission intensities – N<sub>2</sub>O per tonne**

Given the two hypothesised responses of N<sub>2</sub>O emission to fertiliser N application rate i.e. linear (supply related) or non-linear, increasing quadratic (balance related), and the assumption that the yield response curve would be typical, the expectation was that two possible shapes of emission intensity curves would be observed ('hockey stick' or 'U-shaped' respectively), dependant on the response of N<sub>2</sub>O emission per hectare to N rate (Figure 1). In reality, four relationships were detected, linear, non-linear (increasing quadratic) and 'U-shaped', whilst some experiments showed no significant relationship. The absence of 'hockey stick' relationships may partly reflect the modest levels of precision achieved in these experiments, as well as there being few of these actual responses. However, the 'hockey stick' response shape would be expected with a linear relationship between N<sub>2</sub>O emission and N rate, given the crop yield response to N rate. A lack of additional curvature in emission intensity is illustrated in Example 1 and, as with the other two experiments where this occurred, this probably reflected relatively poor responses in yield to the addition of N, as well as the difficulty in differentiating curve shapes with somewhat variable data. Primarily as a result of drought there was only a small 1.4 t ha<sup>-1</sup> increase in yield of winter wheat (Example 1) from 3.7 t ha<sup>-1</sup> (0N treatment) to 5.1 t ha<sup>-1</sup> (40% treatment) (Figure 18). It is possible that difficulty in detecting curvature was compounded by the use of only 5 fertiliser N rates. Figure 18 shows that the emission intensity corresponding to the 0N treatment was slightly greater than for the 40% treatment; with an increased number of N rates, particularly at the lower end, a 'slightly non-linear' shape might have been detected as statistically significant. In a two-year study in Eastern China, the response of N<sub>2</sub>O emission to urea N rates (0-270 kg ha<sup>-1</sup>) applied to winter wheat was linear, as was the N<sub>2</sub>O emission intensity (kg N<sub>2</sub>O-N t<sup>-1</sup>) response (Ji *et al.*, 2012). In keeping with our Example 1, Ji *et al.* (2012) reported a poor response of yield to N rate with only an increase in yield of 0.8 t ha<sup>-1</sup> from the 0N treatment (3.2 t ha<sup>-1</sup>) to the lowest N rate of 100 kg N ha<sup>-1</sup> (4.0 t ha<sup>-1</sup>). Even at the top N rate (270 kg N ha<sup>-1</sup>) the measured yield was only 4.6 t ha<sup>-1</sup>, just 1.4 t ha<sup>-1</sup> more than from the 0N treatment.

In contrast, Example 2 illustrates the situation observed in two of the experiments when there was a strong effect of fertiliser N on yield and a linear N<sub>2</sub>O response to N rate. The yield of spring barley increased by c.3 t ha<sup>-1</sup> from the 0N treatment to treatments receiving the higher rates of N (Figure 21). This, together with the small cumulative N<sub>2</sub>O emissions and a similar magnitude of N<sub>2</sub>O loss from the 0N and 60% treatments, were likely to have changed the linear relationship between N<sub>2</sub>O emission and N rate to a 'U-shaped' N<sub>2</sub>O emission intensity curve. Likewise, in a

study from northern China, there was a linear response of N<sub>2</sub>O emission to N (as urea) rate and there appeared to be a good response in the combined winter wheat plus maize yield to N rate, (Qin *et al.*, 2012). The resultant emission intensity, although expressed on a crop or grain N basis, was also non-linear (cubic). In a separate study from northern China, crop yield apparently also influenced the shapes of the emission intensity curves in wheat and maize crops (Liu *et al.*, 2011), the linear response of N<sub>2</sub>O emission per hectare leading to an increasing exponential response in emission intensity.

Example 3, as with Example 2, also showed a large 4.3 t ha<sup>-1</sup> increase in yield of winter barley from 2.3 t ha<sup>-1</sup> (0N treatment) to 6.5 t ha<sup>-1</sup> (80% treatment) (Figure 24). This contributed to a 'U-shaped' N<sub>2</sub>O emission intensity curve when combined with the non-linear (increasing quadratic) relationship between N<sub>2</sub>O emission per hectare. However, it appears to be the relative curvature of the two responses to N rate (the yield response and the N<sub>2</sub>O response), rather than the size of the response itself, that governs whether the intensity response is 'U-shaped'. For example, in a Canadian study, a large response in maize yield did not substantially change the shape of the emission intensity curve (Ma *et al.*, 2010); the response of N<sub>2</sub>O emission to fertiliser N rate was non-linear (increasing quadratic) but so was the emission intensity curve. However, as with Example 2, determination of the 'U-shaped' N<sub>2</sub>O emission intensity curve was likely to have related to the similar size of emission measured from the 0N and 40% treatments i.e. if the effect of N rate on N<sub>2</sub>O emissions at the lowest N application rate was small, the response in N<sub>2</sub>O intensity was more likely to be 'U-shaped'.

The size of the N<sub>2</sub>O emissions was also important in determining the shape of the emission intensity curve in Example 4. Despite a distinct non-linear (increasing quadratic) response of N<sub>2</sub>O emission per hectare to N rate, this was also the shape of the emission intensity curve. Together with the poor responses in yield to the addition of N at the lower N rate, the relatively small N<sub>2</sub>O emission from the 0N treatment is likely to have influenced the shape of the emission intensity curve.

In Scotland, the predominance of 'U-shaped' N<sub>2</sub>O emission intensity curves probably related to generally good responses of yield to N rate (from 0N to the lowest N application rate) and (possibly more importantly) to the size of the 0N N<sub>2</sub>O emission. Here, emissions from the 0N 'background' treatment tended to be large and much greater than those measured in England. The larger the emission from the 0N treatment, the greater the emission intensity, hence forming the left half of the 'U-shape'. The unusually wet conditions occurring in Scotland in 2012 were associated with a 'U-shaped' emission intensity curve for both winter wheat and spring barley. This pattern developed in response to wetness throughout the growing season that allowed a crop response to applied fertiliser-N along with significant treatment responses to the quantity of N applied.

In the five OSR experiments, the N<sub>2</sub>O emission intensity curve was never found to be ‘U-shaped’ and it is probable that because of the lower yields of OSR compared to spring or winter cereals, a ‘U-shaped’ N<sub>2</sub>O emission intensity curve would rarely occur with OSR.

The relationship between N<sub>2</sub>O emission intensity and N rate was not significant in our experiments predominantly when emissions were low and when there was not a clear increase in the annual N<sub>2</sub>O emission with each increment in N rate.

It would appear that the response of crop yield to N fertiliser application rate can exert a significant role in determining the shape of the emission intensity curve; in some situations this may even be the dominant factor. However, other very important factors determining the emission intensity shape were also the size of the N<sub>2</sub>O emission from the 0N treatment, and the relative size of emissions from the 0N and 40/60% treatments.

Because of the number of variables involved, it seems well-nigh impossible to predict the shape of the N<sub>2</sub>O emission intensity curve before fertiliser application and crop growth. Thus it appears unlikely that practices could be considered to encourage non-linear ‘U-shape’ intensity curves; these would not only depend on how fertiliser practices effect direct N<sub>2</sub>O emissions but they would also be influenced by the efficiency with which the fertiliser caused a yield response. However, given that the mean EF (for the 120% treatment) from these experiments was much smaller than the IPCC default emission, the shape of the emission intensity curve may be less important than was previously supposed in governing fertiliser practices. In any case, it will be important to develop an N use strategy that minimises N<sub>2</sub>O emissions without compromising food production, so it will be important to consider effects of any mitigation strategies on crop productivity as well as on N<sub>2</sub>O emissions intensity.

## **6.7. Conclusion**

The experiments undertaken within this project have adopted a more detailed sampling strategy and a more comprehensive suite of explanatory measurements (especially SMN) than those used in many previous studies, allowing emissions and EFs to be estimated and interpreted with greater precision. The mean annual *direct* N<sub>2</sub>O EF measured from all 24 experiments at 120% of the recommended fertiliser N application rate was 0.67% of total-N applied, less than the IPCC 2006 default EF of 1.0% used in GHG accounting and the GHG inventory calculations. However, the results were greatly affected by rainfall, so need to be considered in the light of longer-term rainfall data (see subsequent sections).

Although the responses of N<sub>2</sub>O emission to N fertiliser application rate were split between linear and non-linear responses (both increasing and diminishing quadratics) there was no consistent pattern across growing conditions. It is evident that the relationship between N<sub>2</sub>O emission and N fertiliser rate is more complicated and less predictable than assumed in our hypothesis. It is not only an artefact of the fertiliser N application pattern used in these experiments, but also depends particularly on fertiliser N rate and timing, crop N uptake (as affected by the growth of the crop), soil conditions principally influenced by rainfall, and the amount and timing of rainfall in relation to the split fertiliser applications.

Weather and in particular rainfall during the growing season was shown, through its effect on the size of N<sub>2</sub>O emissions and on crop N uptake, to be important in determining the shape of emission intensity curves. However, without accurate medium term weather forecasts at the time of fertiliser application, it will be difficult to use this information to minimise emission intensities.

It is apparent that, along with the size of the N<sub>2</sub>O emission, particularly from the nil N 'background' treatment, the extent and shape of the response of crop yield to N fertiliser application rate is also important in defining the shape of the N<sub>2</sub>O emission intensity curve. Crucially, emission intensities may be minimised without significantly reducing crop productivity even with a linear response of N<sub>2</sub>O emission to N rate if there is a good response of yield to the addition of N. If linear responses of both N<sub>2</sub>O emission and N<sub>2</sub>O emission intensity to fertiliser N rate prove to be the norm in average rainfall conditions and when the effects of splits between N timings are discounted, we may have to acknowledge a trade-off between minimising emission intensities and minimising indirect land use change outside of the UK. However, given that N<sub>2</sub>O emissions were less here than was previously supposed, it may prove acceptable to favour high crop yields rather than minimising *direct* N<sub>2</sub>O emission intensities. In any case, our findings here tend to suggest that there will be benefits from upholding husbandry strategies that maintain good soil conditions (drainage, soil structure, etc.); not only will these minimise soil wetness, and therefore N<sub>2</sub>O emissions, but they will also tend to enhance crop productivity, which will have the added benefit of reducing emission intensities.

## 7. N Fixation and Crop Residue Experiments – WP2B

### 7.1. Introduction

Compared with manufactured N fertilisers, the increased use of biological nitrogen fixation (BNF) from legumes may be a more sustainable way of introducing N into agricultural systems (Fustec *et al.*, 2010) whilst improving biological diversity, soil structure and protection from erosion (Jensen and Hauggaard-Nielsen, 2003). Legumes may also offer an opportunity to reduce GHG emissions, partly due to the avoidance of emissions associated with fertiliser manufacture, and partly due to lower field emissions associated with legume production (Rochette and Janzen 2005).

However, there have been few assessments of N<sub>2</sub>O emissions associated with N fixing crops under temperate conditions. Indeed, there has also been little recent work on quantities of N fixed by field legumes (largely pulses) in temperate conditions. In earlier work estimates of effects on N requirements of subsequent crops were relatively small compared to the estimates of N fixed. Reports of N fixed by *Vicia faba* varied between 45 and 600 kg ha<sup>-1</sup> N (Roughley *et al.*, 1983; Sprent and Bradford, 1977). Most of the N obtained by fixation in peas was removed in the pea grain, with only 25% left in the form of crop residues (Jensen, 1989). Legume haulm with less than 20 g kg<sup>-1</sup> N can immobilise N for the first six weeks after incorporation (Jensen, 1989; Fox *et al.*, 1990). Subsequent mineralisation, although uncertain in absolute rate, was related to the (lignin+polyphenol):N ratio of the material. Residue decomposition appears relatively slow (Haystead, 1983) but there can be also be large losses, perhaps through volatilization, and wind removal of material, in addition to leaching and denitrification (Herridge, 1982). It may be that the root and nodule system is the main source of residual N (Dyke and Prew, 1983). Residual effects of pulses on soil N were most recently reported by Sylvester-Bradley and Cross (1991), with peas leaving c.80 kg ha<sup>-1</sup> residual mineral N in the soil (0-90cm) and beans c.70 kg ha<sup>-1</sup> compared to cereals c.60 kg ha<sup>-1</sup>. However, bean and cereal residues appeared to mineralise further over winter, so that differences between previous crops in soil plus crop N were not statistically significant in the spring. There was no indication that higher yielding legume crops left larger N residues (Haystead, 1983).

The calculation of N<sub>2</sub>O emissions from crop residues was introduced in Section 5.2.3 (see page 32). As stated there, a range of uncertainties and issues overlay the estimation of N<sub>2</sub>O emissions from crop residues, some philosophical, others technical. Key issues are as follows:

- Crop residues are broken down slowly, especially residues with high carbon:nitrogen (C:N) ratios, or low N% concentrations, such as cereal straw, so that N becomes available to the soil only slowly. It is, therefore, questionable whether N<sub>2</sub>O emissions from crop residue N should be treated with the same EF as fertiliser N, which makes large concentrations of ammonium and nitrate available to soil biota over a short time-frame. N<sub>2</sub>O emissions from

crop residues may be more related to N concentration (or C:N ratio) of the residue than the amount of N contained in the residue.

- Crop residues, especially those with high C:N ratios, may not completely breakdown within 12 months of incorporation. Indeed, they can add to SOM levels so may increase the carbon stock of the soil. Whilst simplifying assumptions may be acceptable at a broad scale (for GHG inventory purposes), within a LCA protocol aimed at quantifying the effect of a single season of cropping, it may thus be misleading to count N<sub>2</sub>O emissions that may happen more than a year after harvesting the crop, if at all. It is arguable whether these emissions should be counted as belonging to the original crop, or a subsequent crop for which cultivation has occurred. To an extent, emissions from crop residues can be regarded as giving rise to the background or baseline N<sub>2</sub>O emissions that occur in the absence of N fertiliser; these tend to be exacerbated by soil cultivation and establishment of a crop.
- If N<sub>2</sub>O emissions have to be taken into account, would it not also be consistent to include any net carbon sequestration due to crop residue incorporation?
- Measurement of N<sub>2</sub>O emissions from crop residues is generally difficult, and it is challenging to separate these from N<sub>2</sub>O emissions due to N applications to the growing crop. Inclusion of N<sub>2</sub>O emissions from crop residues within a LCA protocol could be considered as double-counting of emissions from N fertiliser use. Alternatively, it could be considered unfair to include emissions from N cycling through the crop, but not to include emissions from the continual N cycling that occurs in the soil.
- Whether or not the straw is removed has a big impact on N<sub>2</sub>O emissions, as estimated by IPCC-based methodologies. Empirical evidence is lacking that removal of residues reduces N<sub>2</sub>O emissions substantially. By assuming that emissions are reduced due to removing straw, a perverse incentive is created for farmers to remove straw, which otherwise might give small benefits in SOM and carbon sequestration (Nicholson *et al.*, 2014). In any case, under most circumstances, removed straw must be returned to the soil as manure, creating an issue over whether N<sub>2</sub>O emissions from straw (removed and incorporated) should be allocated to the original crop, to the livestock enterprise that uses the straw or to the crop for which the manure is returned to the land.

Given all of these uncertainties, Sections 7.2 and 7.3 describe experimental work aimed at quantifying the N<sub>2</sub>O emissions associated with N fixation by pulse crops, and with the subsequent maturation and decomposition periods of pulse crops, and also with the decomposition periods of residues from winter wheat, OSR and sugar beet, all under contrasting climatic conditions. These experimental results are then discussed, along with the more philosophical issues relating to estimation of N<sub>2</sub>O emissions due to N fixation or crop residues in Sections 7.4 and 7.5.

## 7.2. Materials and methods

Three experiments each running for approximately 24 months were carried out either on a sandy loam at Gleadthorpe in central England, on a clay loam in Edinburgh, south east Scotland (both from September 2010 to October/November 2012), and then on a clay loam at Terrington in eastern England (from October 2011 to November 2013). The sites were on commercially managed fields which had received no manure applications in the previous two years and had a low SMN status of <math>80 \text{ kg ha}^{-1}</math> to 90 cm depth; this and further site details are shown in Table 30.

**Table 30. Mean values (n=3) of background SMN (0-90 cm) and topsoil characteristics (0-15 cm depth)**

Site No. and Name	1. Gleadthorpe	2. Edinburgh	3. Terrington
Soil type	Sandy loam	Clay loam	Clay loam
Background SMN ( $\text{kg ha}^{-1}$ to 90cm)	25	35	40
pH	6.1	6.2	8.1
Phosphorous (Olsen) ( $\text{mg kg}^{-1}$ and index)	33 / 3	29 / 3	21 / 2
Potassium ( $\text{mg kg}^{-1}$ and index)	98 / 1	475 / 4	159 / 2-
Magnesium ( $\text{mg kg}^{-1}$ and index)	101 / 3	264 / 6	157 / 3
Clay (%)	10	21	22
Total nitrogen (%)	0.13	0.16	0.16
Total carbon (%)	1.61	1.9	1.8
Total carbon to nitrogen ratio (C:N)	12.0	11.9	11.3
Initial bulk density	1.23	1.23	1.22

### 7.2.1. Experimental design, treatments and measurements

Experiments were set up to compare four N-fixing pulses (winter beans, spring beans, combinable peas and vining peas) and winter wheat with nil N applied from drilling to harvest. Three additional crops, sugar beet, winter wheat (with N applied) and spring or winter OSR were also grown so that effects of crop residues could be tested on all eight crops. [In the spring, N fertiliser (AN; 34.5% N) was applied to these three crops at the recommended N rate (Anon., 2000) (Table 31)]. These eight crop treatments were replicated (x3) in a blocked design with 24 plots (each 12 x 12 m); for practicalities in harvesting sugar beet and OSR treatments were positioned on the end of a block. Sugar beet and OSR plots were randomised within the end plots and remaining treatments were randomised within the non-end plots. All plots were split so that following harvest half of each plot (6 x 12 m) received the above ground 'removable' crop residue (termed the "*incorporated*" treatment) and the other half did not (termed the "*removed*" treatment). Following harvest and autumn cultivation, all plots were drilled with winter wheat and *direct*  $\text{N}_2\text{O}$  fluxes were measured from all eight crops from both "*removed*" and "*incorporated*" sub-plots; measurements continued for 12 months after autumn cultivation and incorporation of the crop residues. The crop yields and N uptakes of the winter wheat (without N applied) were measured as described in Section 6.4.4.

Table 31. Management details for experiments testing N<sub>2</sub>O emissions from N fixations and from incorporation of removable crop residues.

Site No., Name, and Site	Crop / N applied	N applied (kg ha <sup>-1</sup> )	Drilling	Harvest	Residue returned	Residue incorporated	Next crop drilled	Next crop harvested
<b>1. Gleadthorpe, Central England</b>	Winter beans	0	28 Oct 10	16 Aug 11	16 Aug 11	9 Nov 11	10 Nov 11	14 Sept 12
	Spring beans	0	8 Mar 11	16 Aug 11	16 Aug 11	9 Nov 11	10 Nov 11	14 Sept 12
	Combinable peas	0	8 Mar 11	25 Jul 11	25 Jul 11	9 Nov 11	10 Nov 11	14 Sept 12
	Vining peas	0	8 Mar 11	21 Jun 11	22 Jun 11	9 Nov 11*	10 Nov 11	14 Sept 12
	Winter wheat	0	21 Sept 10	19 Aug 11	19 Aug 11	9 Nov 11	10 Nov 11	14 Sept 12
	Sugar beet	120	1 Apr 11	28 Oct 11	28 Oct 11	9 Nov 11	10 Nov 11	14 Sept 12
	Winter Wheat	160	21 Sept 10	19 Aug 11	19 Aug 11	9 Nov 11	10 Nov 11	14 Sept 12
	Spring OSR	220	18 Apr 11	6 Sept 11	6 Sept 11	9 Nov 11	10 Nov 11	14 Sept 12
<b>2. Edinburgh, Central Scotland</b>	Winter beans	0	30 Sept 10	24 Oct 11	24 Oct 11	26 Oct 11	26 Oct 11	18 Sep 12
	Spring beans	0	7 Mar 11	24 Oct 11	24 Oct 11	26 Oct 11	26 Oct 11	18 Sep 12
	Combinable peas	0	7 Mar 11	8 Sept 11	8 Sept 11	26 Oct 11	26 Oct 11	18 Sep 12
	Vining peas	0	7 Mar 11	25 Jul 11	25 Jul 11	26 Oct 11*	26 Oct 11	18 Sep 12
	Winter wheat	0	28 Sept 10	5 Sept 11	5 Sept 11	26 Oct 11	26 Oct 11	18 Sep 12
	Sugar beet	80	29 Mar 11	24 Oct 11	24 Oct 11	26 Oct 11	26 Oct 11	18 Sep 12
	Winter Wheat	200	28 Sept 10	5 Sept 11	5 Sept 11	26 Oct 11	26 Oct 11	18 Sep 12
	Winter OSR	200	26 Aug 10	16 Aug 11	16 Aug 11	26 Oct 11	26 Oct 11	18 Sep 12
<b>3. Terrington, East England</b>	Winter beans	0	20 Oct 11	22 Aug 12	22 Aug 12	18 Nov 12	18 Nov 12	3 Sept 13
	Spring beans	0	20 Mar 12	24 Aug 12	24 Aug 12	18 Nov 12	18 Nov 12	3 Sept 13
	Combinable peas	0	20 Mar 12	22 Aug 12	22 Aug 12	18 Nov 12	18 Nov 12	3 Sept 13
	Vining peas	0	20 Mar 12	12 Jul 12	13 Jul 12	18 Nov 12*	18 Nov 12	3 Sept 13
	Winter wheat	0	21 Sept 11	3 Sept 12	3 Sept 12	18 Nov 12	18 Nov 12	3 Sept 13
	Sugar beet	120	30 Mar 12	12 Oct 12	12 Oct 12	18 Nov 12	18 Nov 12	3 Sept 13
	Winter Wheat	190	21 Sept 11	3 Sept 12	3 Sept 12	18 Nov 12	18 Nov 12	3 Sept 13
	Winter OSR	190	3 Sept 11	8 Aug 12	8 Aug 12	18 Nov 12	18 Nov 12	3 Sept 13

\*Vining pea residues were incorporated on 23 June 2011 at Gleadthorpe and on 25 July 2012 at Edinburgh, but at Terrington soil conditions were too wet for crop residue incorporation immediately following harvest so they remained on the surface until all crop residues were incorporated in November.

### 7.2.2. Direct N<sub>2</sub>O emission measurements

In order to maintain an undisturbed area for crop sampling, *direct* measurements of N<sub>2</sub>O were made from one third (2 x 12 m) of each of the removed and incorporated sub-plots. Five static flux chambers (40 cm wide x 40 cm long x 25 cm high) covering a total surface area of 0.8 m<sup>2</sup> were placed in regular positions in the N<sub>2</sub>O measurement area of the removed and (following harvest) the incorporated sub-plots. To quantify *direct* N<sub>2</sub>O fluxes due to N-fixation from the pulses (and the winter wheat 0N control) N<sub>2</sub>O emission measurements were taken on the “removed” sub-plot from sowing to the end of the experiment, thereby ensuring a 12 month measurement period from N fixation which was not confounded by ‘removable’ crop residue input. During the ‘fixation’ period (i.e. sowing to harvest) the N<sub>2</sub>O sampling strategy involved taking a sample following sowing of the pulses and winter wheat 0N and then approximately twice a month until incorporation, this resulted in approximately 15 sampling occasions.

After incorporation the sampling strategy was weighted so approximately 50% of the samples were taken within the first 6 weeks following incorporation. This was when N<sub>2</sub>O emissions were expected to be greatest. Measurements were taken on three days each week for the first four weeks following incorporation decreasing to fortnightly and then monthly sampling during winter months when emissions were expected to be lower due to low air temperatures. Sampling frequency increased to fortnightly samples from approximately March to June time when air temperatures were expected to be higher, therefore increasing the potential for mineralisation and subsequent N<sub>2</sub>O emissions; finally decreasing to monthly sampling towards the end of the experiment. This sampling schedule resulted in a total of 35 N<sub>2</sub>O sampling days over 12 months from incorporation. Further details of the N<sub>2</sub>O methodology are given in Section 6.4.1.

For statistical analysis, N<sub>2</sub>O fluxes from the five replicate chambers per sub-plot were averaged (arithmetic mean) and the cumulative N<sub>2</sub>O emission from each treatment (crop x residue) was estimated by interpolating fluxes linearly between sampling points and integrating the daily emissions. Cumulative emissions were calculated over three time periods (according to the dates given in Table 31) as follows:

1. **Sowing to harvest:** for the pulses and winter wheat 0N control. Note that the time period over which N<sub>2</sub>O was integrated differed between crops due to their different sowing and harvest dates.
2. **Harvest to Incorporation:** during this period the ‘removable’ residue was sitting on the surface of incorporated sub-plots; except for vining peas at Gleadthorpe and Edinburgh which (in keeping with commercial practice) was incorporated immediately after harvest. At Terrington residue incorporation was not possible immediately after vining peas’ harvest due to wet soil conditions. Again, the different harvest dates of the crops caused the periods for integrating N<sub>2</sub>O emissions to differ between crops.

3. **12 months from residue incorporation:** both the removable and incorporated sub-plots were cultivated in the autumn ahead of winter wheat drilling. The autumn incorporation date was the same for all crops, therefore cumulative N<sub>2</sub>O was calculated over the same period for all crops.

For each experiment, significant differences ( $P < 0.05$ ) in cumulative emissions and EFs were tested using Analysis of Variance (ANOVA) within GENSTAT version 12 (Lawes Agricultural Trust, 2009).

### **7.2.3. Soil and meteorological measurements**

To enable the effects on N<sub>2</sub>O emissions of soil temperature and soil moisture to be assessed, soil temperature (at a depth of 5 cm) was logged continuously, along with measurements of air temperature and daily rainfall at a meteorological station within 7.5 km of each site. On every N<sub>2</sub>O measurement date, representative soil samples were taken (0-10 cm) from each plot for the determination of gravimetric moisture and SMN (ammonium-N and nitrate-N) content. Further details of the soil sampling methodology are given in Section 6.4.1.

Topsoil dry bulk density measurements (0-5.5 cm) were also taken from all plots one month after sowing, in the spring on winter sown plots, following harvest, after primary and secondary autumn soil cultivations and one month after sowing the subsequent winter wheat crop. The soil bulk density was used to convert the soil gravimetric moisture contents to WFPS. At the start of the experiment, soil pH, extractable P, K and Mg concentrations, particle size distribution and total N and carbon levels were measured from across each block (0-15 cm sampling depth), along with SMN (0-90 cm) (Table 30).

### **7.2.4. Crop measurements during growth and at harvest**

Whole crop samples were taken at intervals during the season and pre-harvest in order to quantify N uptake and measure crop N returns at harvest. This applied to winter wheat (0N) and the pulses as well as the follow-on winter wheat crop. The harvest yield and N recovery were recorded for the 'incorporated' and 'removed' sub-plots; this was to enable accurate estimation of the crop residue N-returned to each sub-plot. The methodology for determining N uptake is detailed in Section 6.4.4; the same approach was used for cereals and pulses.

At Gleadthorpe and Terrington plant counts and root nodule assessments per plant were carried out on all plots of winter beans, spring beans, vining peas and combining peas. Plant populations were counted after establishment when rows were visible and at least two true leaves present (GS 1.01). Root nodules were assessed at the same time on the whole crop samples taken to measure N uptake (Section 6.4.4). The below ground material of 5 plants from each quadrat was washed

and any nodules photographed; the nodules were broken open and the number of white, pink or red nodules recorded (nodule redness was assumed to relate to its fixation activity).

The methods for determining N recovery at harvest and yield of cereals, OSR and sugar beet are detailed in Section 6.4.5. The method for OSR was used for the pulses; however, methods were amended to ensure that crop uptakes were separable into their respective plant components (e.g. stubble, baleable straw, chaff (or pods) and grain or seeds (or sugar beet roots). Also, samples of crop residues were analysed for total C% as well as N%. For the English sites, total N and carbon analyses of plant material were performed using the Dumas combustion method with a Leco FP-228 N Determinator (Leco Corporation, St. Joseph, Michigan). For the Scottish sites total carbon and N analyses were undertaken using a Thermo Scientific Flash 2000 elemental analyser (Cambridge).

Table 32 lists the plant components which remained on the ‘incorporated’ and ‘removed’ sub-plots for each crop following harvest. Quantities of N returned to the soil in these residues were calculated using crop partitioning measurements (proportions of total crop DM in each above-ground component and N contents of each) and the crop yield. Where possible, the DM yield was taken from the combine harvest rather than the pre-harvest quadrat to ensure representativeness.

**Table 32. Residues remaining on the ‘Removed’ and ‘Incorporated’ sub-plots after harvest**

<b>Crop</b>	<b>‘Removed’: parts remaining</b>	<b>‘Incorporated’ parts remaining</b>
<b>Winter beans</b>	Stubble	Stubble, baleable straw and pods
<b>Spring beans</b>	Stubble	Stubble, baleable straw and pods
<b>Combinable peas</b>	Stubble	Stubble, baleable straw and pods
<b>Vining peas</b>	[None]	All haulm, including pod walls
<b>Winter wheat</b>	Stubble and chaff	Stubble, baleable straw and chaff
<b>Sugar beet</b>	Fallen leaves	Fallen leaves and tops
<b>Oilseed rape</b>	Stubble and pod walls	Stubble, baleable straw and pod walls

For cereals, OSR and pulses (except vining peas) the yield at harvest was determined using a plot combine harvester (Section 6.4.6) following an assessment of lodging. Grain or seed yields were corrected to 85% DM for cereals, 91% for OSR and 86% for pulses. Representative grain sub-samples were analysed by near-infrared spectroscopy (MAFF, 1986) for N% (hence protein); for all other crops N% analysis was conducted by the Dumas combustion method.

Sugar beet was harvested and yield calculated as detailed in Section 6.4.6. Vining peas were harvested by hand as whole above-ground plants from 7.2 m<sup>2</sup> and vined with a static viner to remove the peas. Representative samples of peas and haulm were analysed for dry matter and

total N% by the Dumas combustion method. Vining pea yields were expressed on a 'as harvested' fresh weight basis (~20%DM).

## **7.3. Results**

### **7.3.1. Weather**

At Gleadthorpe in 2011 rainfall was considerably less than the LTA and especially low in spring and early summer (Table 20). Following spring drilling, rainfall in March was c.20% and in April c.5% of the LTA, and rainfall remained less than average up to harvest of the pulse crops in August. The exceedingly low rainfall following drilling in 2011 was compounded by an increase in the mean April air temperature of c.2.5°C above the LTA. Subsequently temperatures were similar to the LTA up to harvest (Table 21). The topsoil WFPS generally remained <30%. As the season progressed many crops showed symptoms of water stress, and this greatly impacted on crop yields (Table 33). In 2012, early spring conditions were relatively dry, rainfall in February was c.25% of LTA and in March c.64% of LTA; April was then extremely wet with rainfall >3-fold greater than the LTA. Rainfall again was greater than the LTA in the summer months ranging from c.150% to 175% of the LTA. Correspondingly, WFPS in April was c.43% and in the summer months c.40%. Further details of rainfall and air temperature from 2010-2012 are described in Section 3.5.1.

In contrast to Gleadthorpe, rainfall at Edinburgh in 2011 was high in spring and summer, exceeding the LTA in January to March and May to July, with only April being relatively dry – 70% of the LTA (Table 20). April was also warm being c.2°C above the LTA, but the rest of the year was cooler than average (Table 21). In 2012, conditions were extremely wet, rainfall was +c.80% of LTA (Table 20). Early spring 2012 was generally warm with the rest of the year remaining cool. Further details of rainfall and air temperature from 2010-2012 are described in Section 3.5.1.

At Terrington, following drilling of the winter wheat 0N and winter beans in autumn 2011, monthly cumulative rainfall was 55% and 45% of the LTA in September and October respectively. In 2012 monthly cumulative rainfall remained below the LTA until the spring, while in April, June and July rainfall was c.2.5 to 3-fold greater than the LTA (Table 20). Mean monthly temperature from September 2011 to May 2012 typically ranged from 1 to 3°C above the LTA (Table 21). For all pulses and for 0N winter wheat, WFPS remained between 50-60% from drilling to harvest, and on just one sampling occasion at the end of May fell to c.30% following 10 days of no rainfall and air temperatures of up to 18°C. In 2013 (data not presented here), aside from March which was 3°C cooler than average, temperatures approximated to the LTA. Annual rainfall in 2013 was 75% of the LTA; June was particularly dry with just 10% of the LTA (data not presented here).

### 7.3.2. Crop performance

Crop yields and total N-uptake for the 3 experiments are presented in Table 33. Crop growth at Gleadthorpe in 2011 was severely hampered by the dry conditions and was reflected in the low yields. Winter beans at Gleadthorpe experienced severe frost and snow damage, consequently growth across the plot was very patchy and it was not possible to take a yield measurement. However, there were winter bean plants growing within the N<sub>2</sub>O chambers and it was therefore still possible to measure N<sub>2</sub>O fluxes. Also at Gleadthorpe the winter OSR crop failed due to severe frost, so this crop was replaced with spring OSR. Across all three sites, the highest pulse yields were recorded at Terrington whereas winter wheat and OSR were most successful at Edinburgh.

**Table 33. Crop yield (t ha<sup>-1</sup>) and total nitrogen uptake (kg ha<sup>-1</sup>) in the N fixation and crop residue experiments for each crop at each site.**

Treatment	Gleadthorpe, Notts	Edinburgh, Central Scotland	Terrington, Norfolk
<i>Crop yield at standard moisture (t ha<sup>-1</sup>)</i>			
Winter beans	ND	6.2	7.6
Spring beans	1.5	6.3	7.0
Combining peas	2.8	4.4	6.3
Vining peas	5.4	8.6	9.0
Winter wheat 0N	0.6	4.3	1.4
Winter wheat +N	3.0	11.2	3.8
OSR +N	3.0	5.5	4.2
Sugar Beet (dry)	17.0	16.0	18.0
<i>Total N uptake (kg ha<sup>-1</sup>)</i>			
Winter beans	ND	333	480
Spring beans	70	281	370
Combining peas	83	690*	334
Vining peas	168	433*	186
Winter wheat 0N	14	64	60
Winter wheat +N	93	210	152
Winter OSR +N	162	207	218
Sugar Beet	200	319	187

\* N uptake by peas at Edinburgh was exceptionally high, due to persistence of green material until harvest. This must be regarded as atypical.

### 7.3.3. Legume root nodule assessments

At Gleadthorpe, throughout the season nodules were numerous, but small and white in colour. It is likely that the dry soil conditions inhibited rhizobial N-fixation activity. At Terrington, nodule count data (Figure 29) demonstrated that active (red or pink) N-fixing nodules were most frequent on winter beans followed by combinable peas with little difference between spring beans and vining peas. The data show that active N-fixing nodules were most frequent in all legume species from

April to May. Notably the density of root nodules on winter beans rapidly decreased by 80% between assessments carried out on the 26<sup>th</sup> June (c.2500 nodules m<sup>2</sup>) and the 6<sup>th</sup> August (c.500 nodules m<sup>2</sup>).

### 7.3.4. N<sub>2</sub>O emissions from drilling to harvest – the N fixation period

At Gleadthorpe cumulative N<sub>2</sub>O emissions from winter wheat 0N between drilling and harvest were significantly greater than pulses ( $P < 0.05$ ; LSD = 252) at c.500 g N<sub>2</sub>O N ha<sup>-1</sup> (Table 34; measured over 332 days). However, this was due to a peak in N<sub>2</sub>O emissions following cultivation (Figure 33); emissions from winter beans (c. 235 g N<sub>2</sub>O N ha<sup>-1</sup>; measured over 292 days) and all spring pulses (c.15-80 g N<sub>2</sub>O-N ha<sup>-1</sup> measured over 104 to 160 days) were smaller because they did not include this emission event. There were no significant differences between any of the pulse crops ( $P > 0.05$ ), or between these and the cumulative N<sub>2</sub>O emission for winter wheat 0N calculated from the first spring sowing date of pulses (c.75 g N<sub>2</sub>O-N ha<sup>-1</sup> over 163 days).

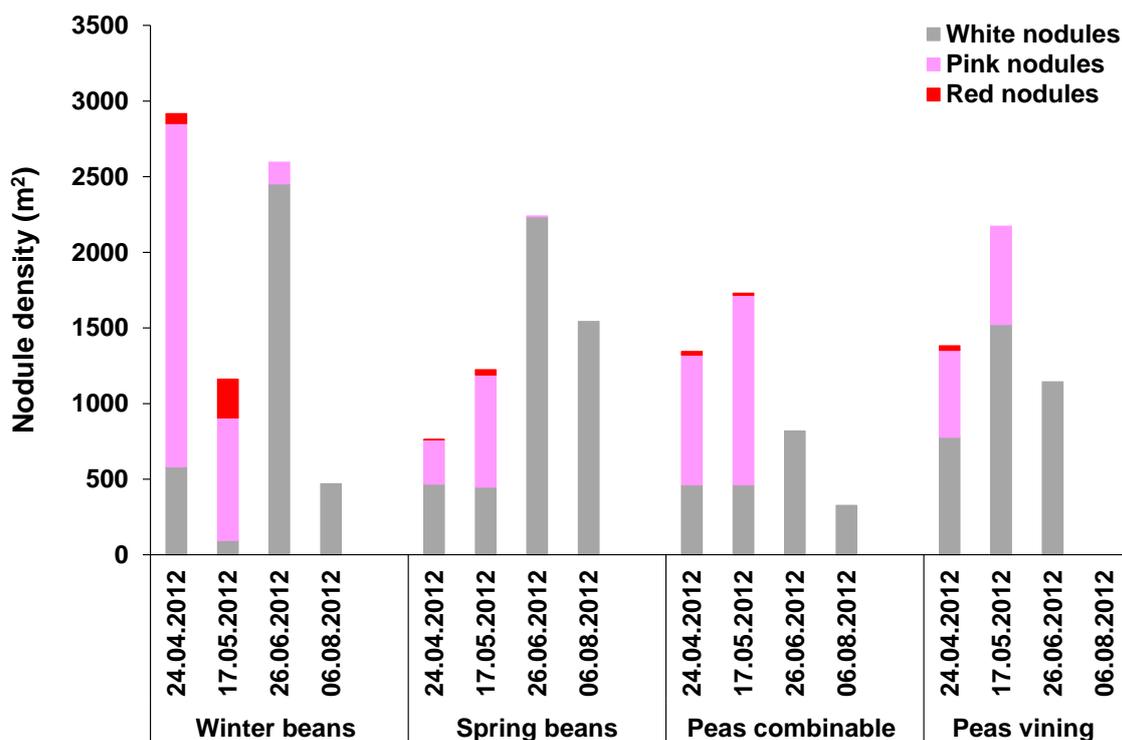


Figure 29. Root nodule density (m<sup>2</sup>) and colour, measured four times through the growing season at Terrington, for winter beans, spring beans combinable peas and vining peas. Nodule redness was assumed to relate to N fixing activity.

At Terrington in 2012 cumulative N<sub>2</sub>O emissions were generally larger than at Gleadthorpe in 2011 (Table 34); the emission from winter beans at c.830 g N<sub>2</sub>O-N ha<sup>-1</sup> was significantly greater ( $P < 0.001$ ; LSD = 138) than winter wheat 0N and all other pulses. This was possibly due to the breakdown of nodules because most emissions occurred towards the end of the season when rapid growth had ceased (Figure 31) this corresponded to an increase in topsoil nitrate content of c.10 kg ha<sup>-1</sup> greater than the control winter wheat 0N (Figure 30).

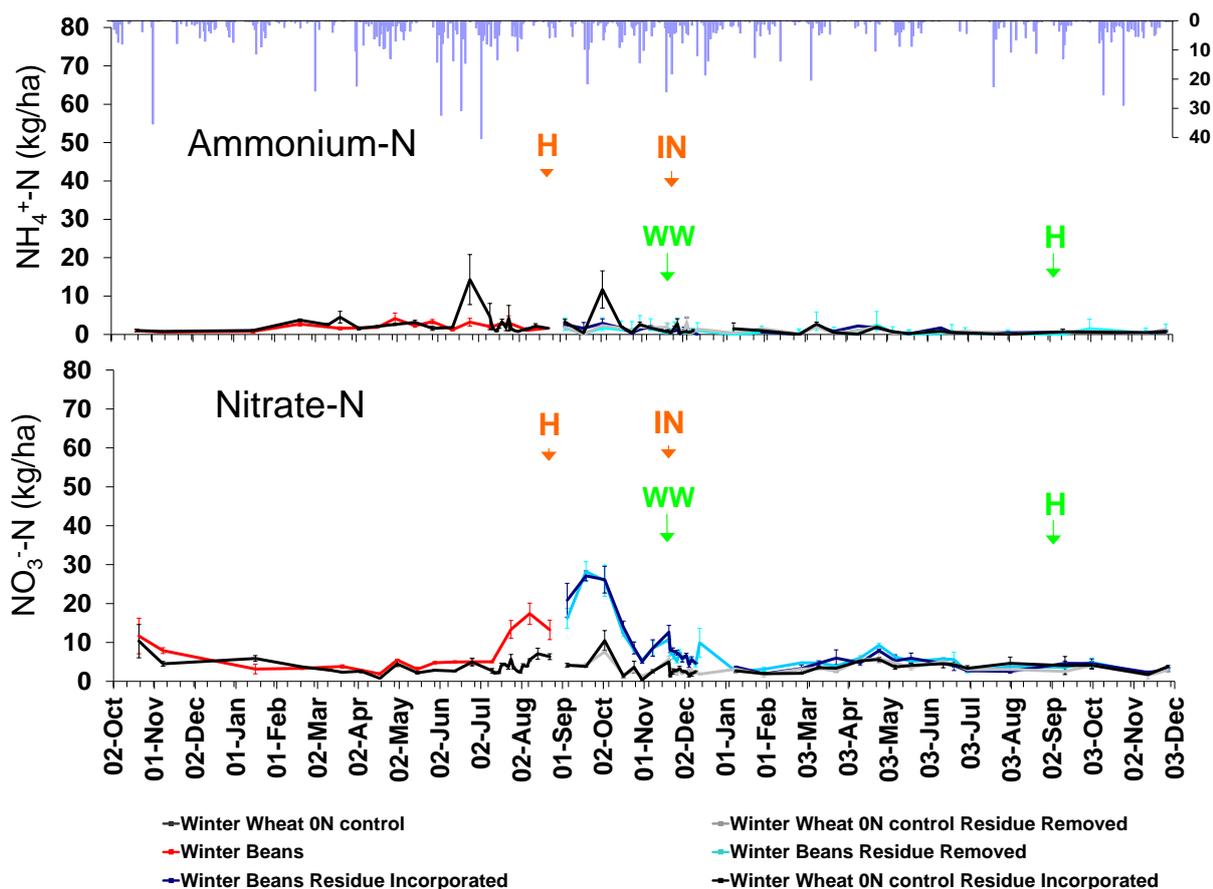


Figure 30. SMN (0-10 cm) presented as mean ammonium-N ( $\text{NH}_4^+\text{-N}$ ) (top) and nitrate-N ( $\text{NO}_3^-\text{-N}$ ) (below) for winter beans and wheat (0N) harvested at Terrington in 2012; error bars represent  $2 \times \text{SE}$  of the mean. The letter 'H' denotes dates of harvest, 'IN' the date of incorporation, and 'WW' the date of sowing the next wheat. Hanging blue bars represent daily rainfall (mm).

At Edinburgh in 2011 (with the exception of spring beans)  $\text{N}_2\text{O}$  emissions overall were greater than those at either of the English sites. Background emissions (i.e. measured from winter wheat 0N) were high at c.1,300  $\text{g N}_2\text{O-N ha}^{-1}$ . Emissions of  $\text{N}_2\text{O}$  from the fertiliser N rate experiments at Edinburgh were also higher in 2011 than in other years (Table 24). There was a significant crop effect ( $P = 0.003$ ;  $\text{LSD} = 745$ ) with the largest  $\text{N}_2\text{O}$  emissions being from winter beans at 1,985  $\text{g N}_2\text{O-N ha}^{-1}$ . However, there was no significant difference between this and  $\text{N}_2\text{O}$  emissions from winter wheat 0N ( $P > 0.05$ ). Emissions from winter beans (measured over 342 days) were significantly greater than from spring pulses ( $P = 0.003$ ;  $\text{LSD} = 745$ ) but this can be explained by the longer measurement period for winter beans rather than by root and nodule breakdown as reported at Terrington. When comparing emissions associated with spring pulses to those for winter wheat 0N for the equivalent period, there was a significant crop effect ( $P = 0.024$ ;  $\text{LSD} = 581$ ). However, the background emission was also high. The highest  $\text{N}_2\text{O}$  emissions were measured from combinable peas at c.1000  $\text{g N}_2\text{O-N ha}^{-1}$  followed by winter wheat 0N at c.870  $\text{g N}_2\text{O-N ha}^{-1}$ ; however, there were no significant differences between these crops ( $P > 0.05$ ). Only

spring beans emitted significantly less than either winter wheat 0N or combinable peas at c.70 g N<sub>2</sub>O-N ha<sup>-1</sup> (P <0.05).

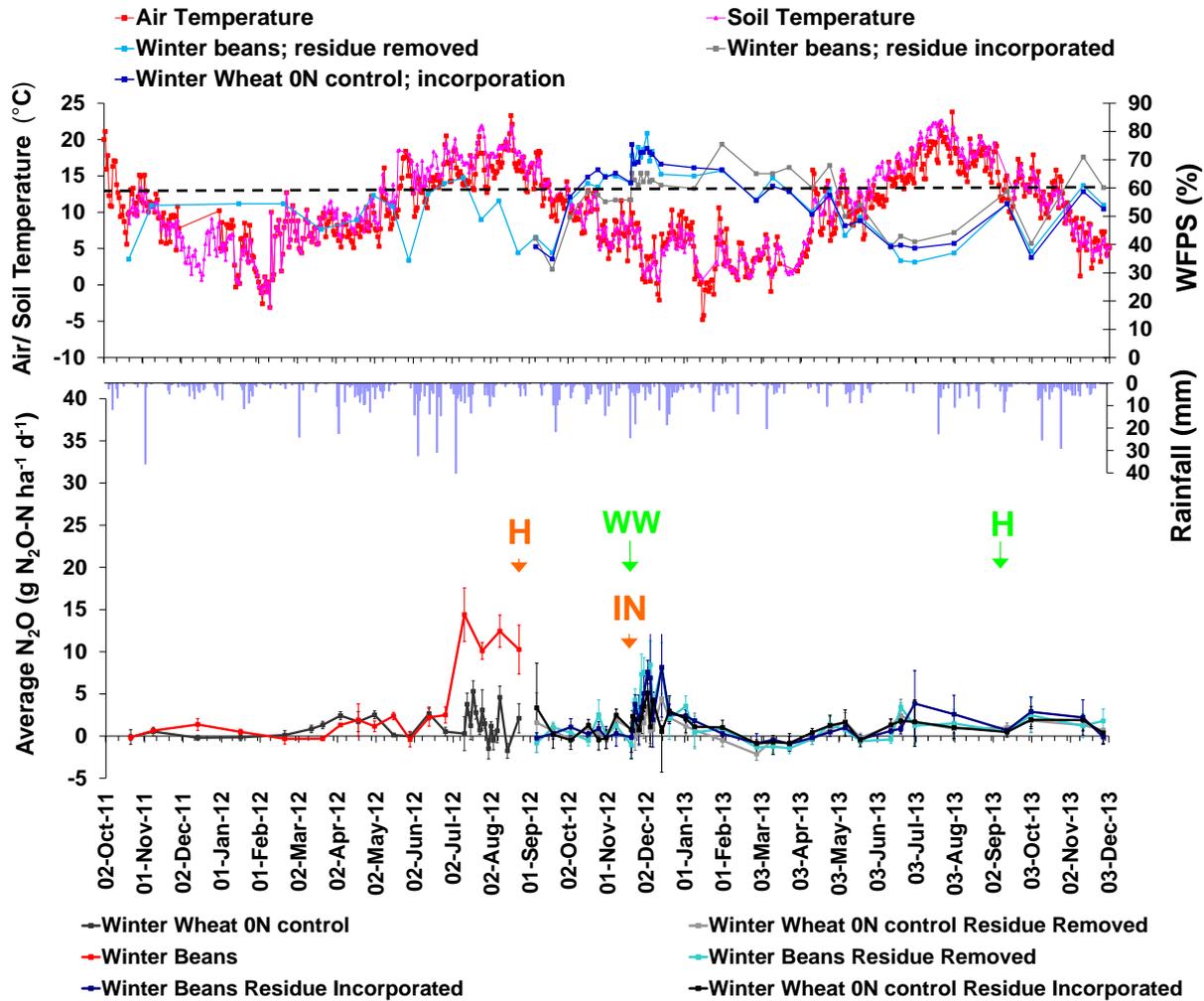


Figure 31. Daily air and soil temperatures, percentage WFPS (Top) and mean daily N<sub>2</sub>O emissions (below) for winter beans and wheat (0N) at Terrington in 2012. Letter ‘H’ denotes dates of harvest, ‘IN’ is the date of incorporation, and ‘WW’ the sowing date for the next wheat. Hanging blue bars show daily rainfall (mm). The dotted line represents 60% WFPS.

### 7.3.5. Nitrous oxide emissions from harvest until autumn residue incorporation

Cumulative N<sub>2</sub>O emissions from harvest of each crop to autumn incorporation (

Table 35) were for different periods, depending on the date of harvest of that crop. Following commercial practice, vining peas at Gleadthorpe and Edinburgh had a primary shallow incorporation immediately following harvest, but this was not possible at Terrington due to wet conditions; thus here, pea residues on the “incorporated” side of the plot remained on the surface until autumn incorporation.

### **Gleadthorpe in 2011**

Only pulses and winter wheat 0N were measured at Gleadthorpe in this period; overall N<sub>2</sub>O emissions here were small, reflecting the dry soil conditions with WFPS around 30% or less. There was a significant crop effect ( $P=0.010$ ; LSD = 61) but no residue effect ( $P>0.05$ ) on N<sub>2</sub>O emissions. When comparing the crop effect (i.e. mean of incorporated and removed sub-plots), the largest emission was from vining peas at c.140 g N<sub>2</sub>O-N ha<sup>-1</sup>; winter beans were next most at c.85 g N<sub>2</sub>O-N ha<sup>-1</sup> (note that although this crop failed on most of the plot, there were winter beans growing in the N<sub>2</sub>O sample chambers). Emissions from vining peas were significantly greater ( $P<0.05$ ) than from spring beans, winter wheat 0N or combinable peas, probably because of the additional incorporation and the physical disruption of the soil and below-ground crop biomass, which may have stimulated N<sub>2</sub>O emissions from decomposition of roots and nodules (Figure 33).

### **Edinburgh in 2011**

Again at Edinburgh in 2011 there was a significant crop effect ( $P<0.01$ ; LSD = 356) but no residue effect ( $P>0.05$ ). Winter wheat (0N) emitted significantly more N<sub>2</sub>O than all other crops ( $P<0.05$ ), its mean cumulative emission being c.1200 g N<sub>2</sub>O-N ha<sup>-1</sup>. There was no difference in cumulative N<sub>2</sub>O emissions between the pulses ( $P>0.05$ ) possibly due to the large background N<sub>2</sub>O emissions, but also due to variability across the site, e.g. there was a significant block effect ( $P<0.05$ ).

**Table 34. Mean cumulative N<sub>2</sub>O emissions from drilling to harvest at three sites in the UK. Note that periods differ between crops (see Table 31). Emissions for the winter wheat (0N) control are presented from autumn drilling (1) and from drilling of the spring pulses (2).**

<b>Crop</b>	<b>Gleadthorpe, Notts. 2011</b>	<b>Edinburgh, Scotland 2011</b>	<b>Terrington, Norfolk 2012</b>
	<i>Cumulative N<sub>2</sub>O-N, g ha<sup>-1</sup></i>		
Winter wheat 0N (1)	502	1311	251
Winter beans	235	1985	829
Winter wheat 0N (2)	75	874	212
Spring beans	79	73	333
Combinable peas	14	1081	499
Vining peas	115	593	227
	<i>N uptake by N-fixing crops, kg ha<sup>-1</sup> (=F<sub>BN</sub>; IPCC 1996)</i>		
Winter beans	70	319	368
Spring beans	70	323	332
Combinable peas	136	226	280
Vining peas	70	108	108
	<i>Cumulative N<sub>2</sub>O-N, % N uptake</i>		
Winter beans	0.32	0.62	0.23
Spring beans	0.11	0.02	0.10
Combinable peas	0.01	0.48	0.18
Vining peas	0.18	0.55	0.23

**Table 35. Mean cumulative N<sub>2</sub>O-N emitted between harvest and autumn incorporation at all three sites**

<b>Crop</b>	<b>Removable Residue</b>	<b>Gleadthorpe</b>	<b>Edinburgh</b>	<b>Terrington</b>
<i>Mean cumulative N<sub>2</sub>O-N (g ha<sup>-1</sup>)</i>				
Winter beans	Incorporated	81	490	104
	Removed	92	672	114
Spring beans	Incorporated	70	494	159
	Removed	74	616	147
Combinable peas	Incorporated	15	734	124
	Removed	16	717	78
Vining peas	Incorporated	167	903	472
	Removed	108	913	497
Winter wheat 0N	Incorporated	74	1060	68
	Removed	51	1399	57
Winter Wheat +N	Incorporated	n/a	721	26
	Removed	n/a	656	56
OSR +N	Incorporated	n/a	235	75
	Removed	n/a	322	76
Sugar beet +N	Incorporated	n/a	492	36
	Removed	n/a	407	59
All crop mean	Incorporated	81	641	133
	Removed	68	713	136

### ***Terrington in 2012***

N<sub>2</sub>O emissions were measured from harvest for all crops because wet conditions in autumn 2012 delayed cultivation until November; omission of any N<sub>2</sub>O emissions from sugar beet (+N), OSR (+N) or winter wheat (+N) was avoided by commencing sampling before incorporation. As at Gleadthorpe in 2011, overall N<sub>2</sub>O emissions were low and there was a significant crop effect ( $P < 0.01$ ; LSD = 113) but no residue effect ( $P > 0.05$ ). Emissions were greater than at Gleadthorpe 2011, probably because soil moisture conditions were wet during this period (mean WFPS c.45%; range 30-65%; Figure 35).

In the wet conditions following vining pea harvest, with the removable residue remaining on the surface, there were clear differences between the removed and incorporated plots in the soil nitrate (NO<sub>3</sub>-N kg ha<sup>-1</sup>) content in the top 0-10 cm (up to c.30 kg NO<sub>3</sub>-N ha<sup>-1</sup>; Figure 34); however, this did not translate into differences in N<sub>2</sub>O emissions (Figure 35). The cumulative N<sub>2</sub>O emission of c.485 g N<sub>2</sub>O-N ha<sup>-1</sup> from vining peas was significantly greater ( $P < 0.05$ ) than all other crops, with daily emissions peaking up to 25 g N<sub>2</sub>O-N ha<sup>-1</sup> d<sup>-1</sup>, however, the magnitude of these emissions was small, being similar to emissions following the application of fertiliser N in a dry spring (e.g. Figure 17).

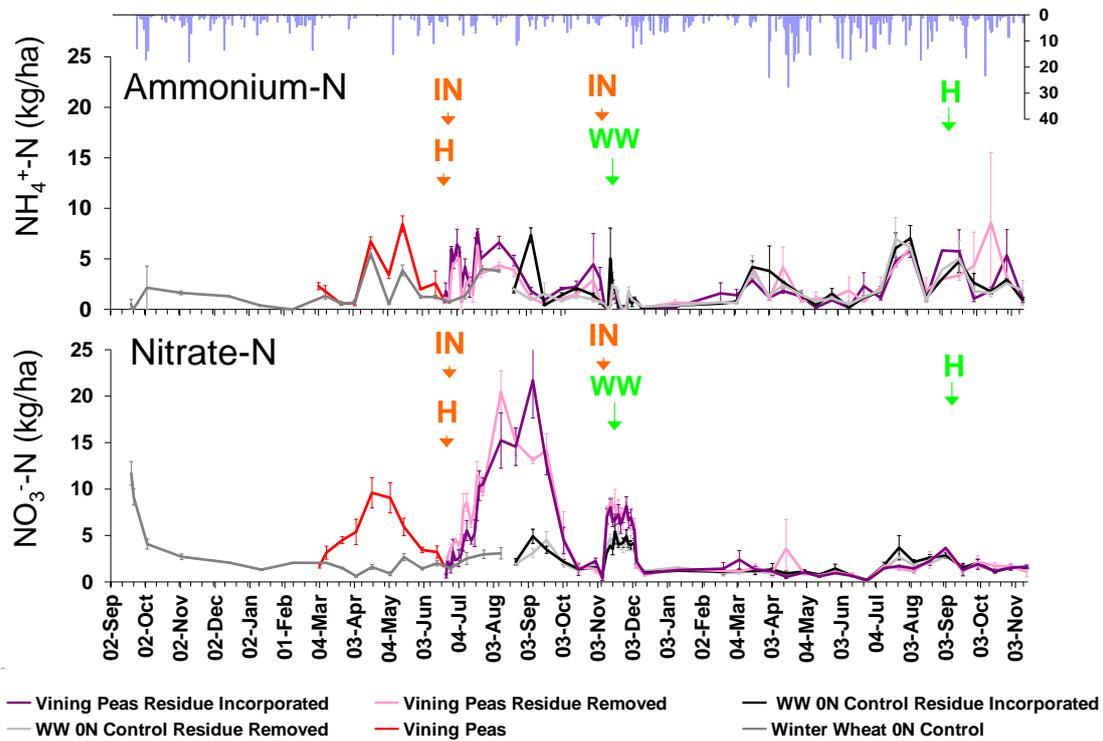


Figure 32. SMN (0-10 cm) presented as mean ammonium-N ( $\text{NH}_4^+\text{-N}$ ) (top) and nitrate-N ( $\text{NO}_3^-\text{-N}$ ) (below) for vining peas at Gleadthorpe in 2011. Error bars represent 2 x SE of the mean. The letter 'H' denotes the date of vining pea harvest, 'IN' the date of incorporation, 'WW' the date of sowing the subsequent winter wheat crop and 'H' its harvest. Hanging blue bars show total daily rainfall (mm).

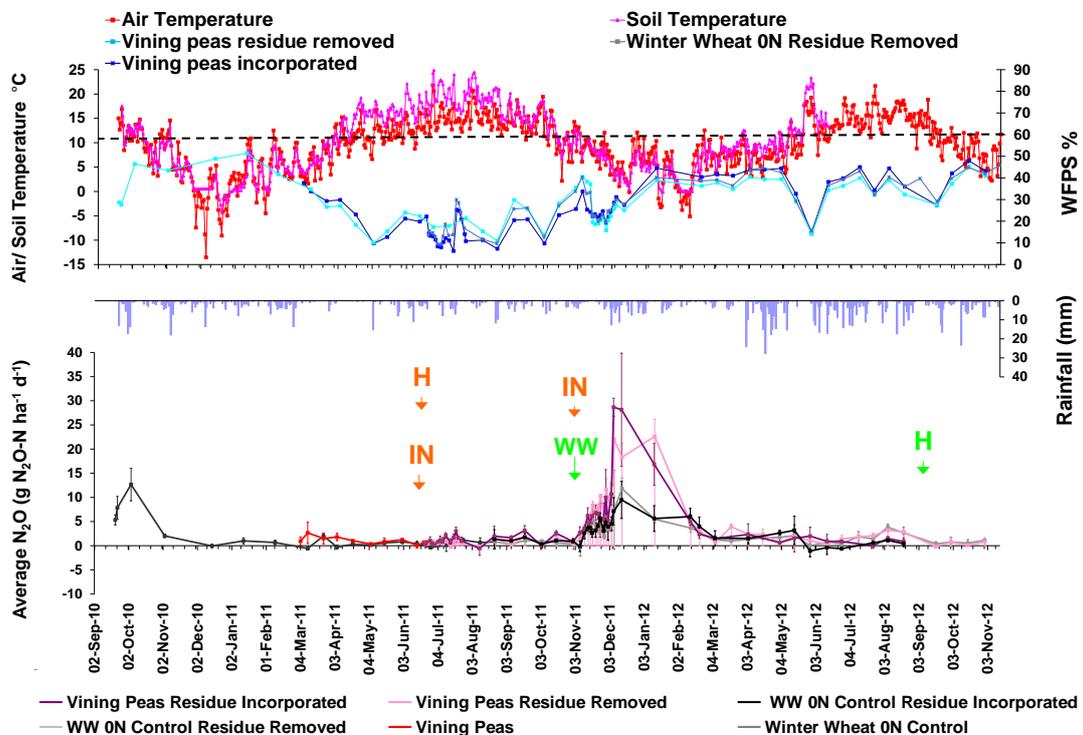


Figure 33. Mean daily air and soil temperature ( $^{\circ}\text{C}$ ), and percentage water filled pore space (WFPS) (top) and mean daily  $\text{N}_2\text{O}$  emissions ( $\text{g N}_2\text{O-N ha}^{-1} \text{d}^{-1}$ ) (below) for vining peas at Gleadthorpe in 2011. The letter 'H' denotes the date of vining pea harvest, 'IN' the date of incorporation, 'WW' the date of sowing the subsequent winter wheat crop, and 'H' its harvest. Hanging blue bars represent total daily rainfall (mm). The dotted line represents 60% WFPS.

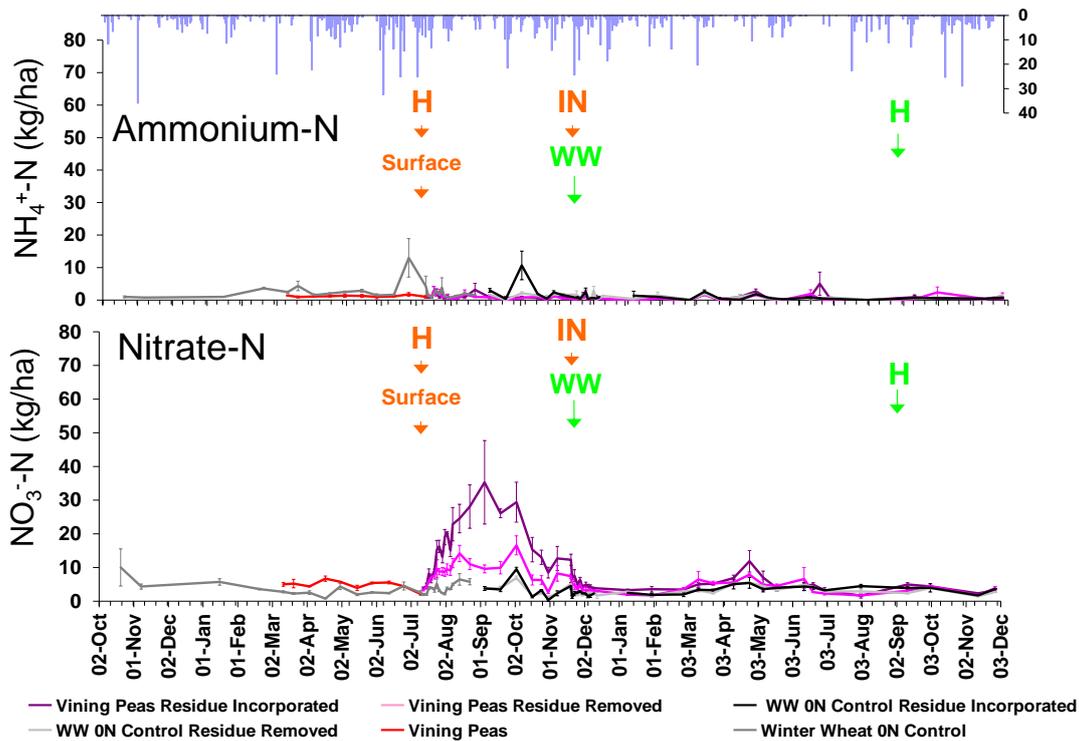


Figure 34. SMN (0-10 cm) presented as mean ammonium-N ( $\text{NH}_4^+\text{-N}$ ) (top) and nitrate-N ( $\text{NO}_3\text{-N}$ ) (below) for vining peas at Terrington in 2012. Error bars represent 2 x SE of the mean. The letter 'H' denotes dates of vining pea harvest, 'surface' indicates residue left on soil surface following harvest, 'IN' the date of incorporation, and 'WW' the date of sowing the next winter wheat crop. Hanging blue bars indicate total daily rainfall (mm).

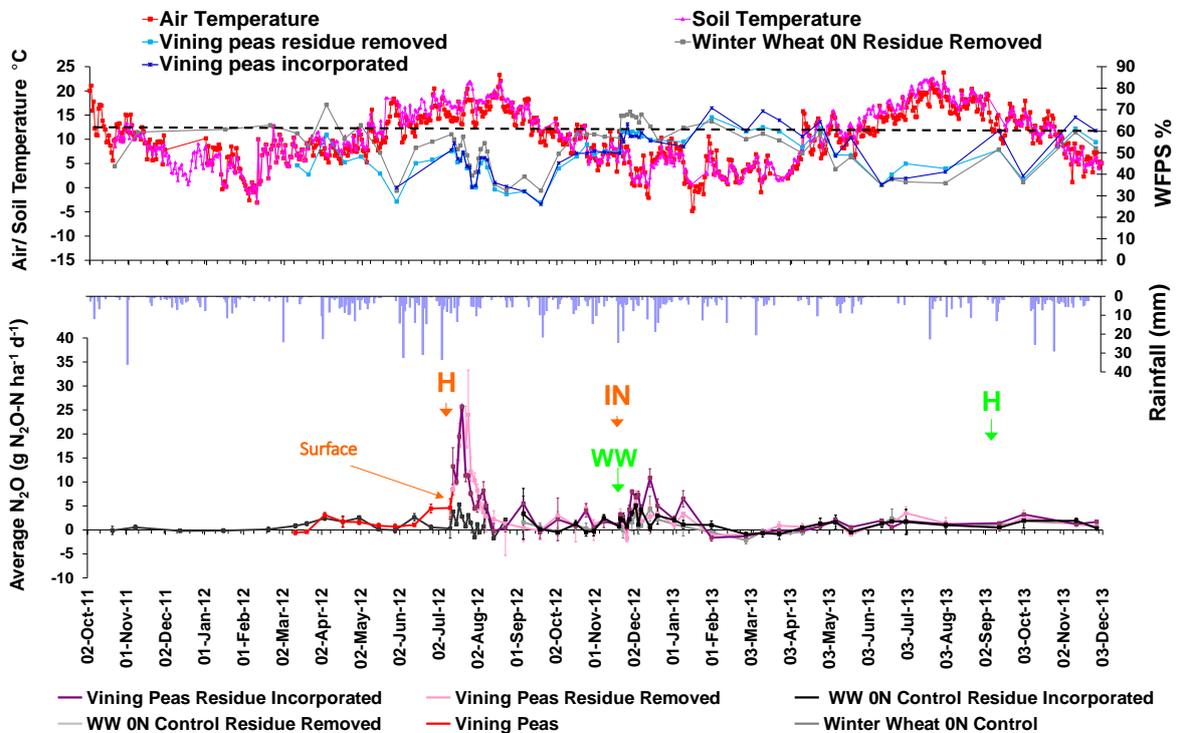


Figure 35. Air and soil temperatures and percentage WFPS (top) and mean daily  $\text{N}_2\text{O}$  emissions (below) for vining peas at Terrington in 2012. The letter 'H' denotes dates of harvest, 'IN' the date of incorporation, and 'WW' the date of sowing the next winter wheat crop. Hanging blue bars represent total daily rainfall (mm). The dotted line shows 60% WFPS.

### 7.3.6. Nitrogen uptake of the follow-on winter wheat crop

Nitrogen captured by the following unfertilised crop of winter wheat can be taken to indicate the extent of the soil N residue left by the preceding crop and its returned residues; this has been termed 'soil N supply', or SNS (Kindred *et al.*, 2012). There were significant effects of both crop and residue removal on SNS in the experiments at Gleadthorpe (crop:  $P < 0.001$ ; LSD = 4.21; residue:  $P < 0.01$ ; LSD = 2.25) and Edinburgh (crop:  $P < 0.001$ ; LSD = 19.73; residue:  $P < 0.01$ ; LSD = 9.86) but only a significant effect of crop at Terrington (crop:  $P < 0.001$ ; LSD = 25.81) (Table 36). There was a significant interaction between crop and residue (crop:  $P < 0.01$ ; LSD = 27.90) removal at Edinburgh, whereby removal increased rather than decreased SNS after most crops. The only crops where this increase did not happen were vining peas, wheat (0N) and sugar beet. This is likely to be explained by patterns of mineralisation, immobilisation and leaching after incorporation. Both the sugar beet leaves and the vining peas at Edinburgh returned large amounts of total N, allowing mineralisation processes to release SMN which would have been available for plant uptake. In general, the pulses resulted in greater SNS than other crops; OSR resulted in more than fertilised wheat or sugar beet (Table 36).

**Table 36. Nitrogen uptake ( $\text{kg ha}^{-1}$ ) by an unfertilised crop of winter wheat grown after experiments comparing a range of arable crops with and without removal of their removable residues (straw, haulm or leaves).**

Crop	Removable Residue	Gleadthorpe 2011	Edinburgh 2011	Terrington 2012	Crop mean
Winter beans	Incorporated	ND*	80	167	127
Winter beans	Removed	ND*	124	138	
Spring beans	Incorporated	41	82	165	97
Spring beans	Removed	38	110	145	
Combinable peas	Incorporated	37	98	139	94
Combinable peas	Removed	35	111	144	
Vining peas	Incorporated	37	94	175	93
Vining peas	Removed	38	84	132	
Winter wheat 0N	Incorporated	34	58	54	51
Winter wheat 0N	Removed	30	59	68	
Sugar beet +N	Incorporated	44	103	66	68
Sugar beet +N	Removed	36	82	75	
Winter Wheat +N	Incorporated	42	67	72	64
Winter Wheat +N	Removed	38	91	74	
OSR +N	Incorporated	45	84	95	87
OSR +N	Removed	43	139	117	
Mean	Incorporated	40	83	117	
	Removed	37	100	112	
Crop $P$ -value		<0.001	<0.001	<0.001	
Crop LSD		4.2	19.7	25.8	
Residue $P$ -value		<0.01	<0.01	0.44	
Residue LSD		2.3	9.9	12.9	

\*crop failure

### 7.3.7. Nitrous oxide emissions over the twelve months after incorporation

The period of integration after residue incorporation was the same for all crops – 12 months. Whilst the assessments of SNS showed crop effects and some residue effects, it should be recognised that SNS is extracted by the wheat crop from all soil depths, and from atmospheric deposition, rather than just from the topsoil where most N<sub>2</sub>O emissions arise. Hence treatment effects on N<sub>2</sub>O emissions over the 12 months after residue incorporation may not directly reflect SNS differences.

#### ***Gleadthorpe 2011***

At Gleadthorpe, after incorporation in autumn 2011 cumulative N<sub>2</sub>O emissions showed a significant effect of crop ( $P < 0.001$ ; LSD = 303), a residue effect ( $P < 0.05$ ; LSD = 152) and an interaction between them ( $P < 0.001$ ; LSD = 429). However, sugar beet was the only crop for which a significant ( $P < 0.05$ ) effect of residue was detected. Mean cumulative N<sub>2</sub>O emissions from sugar beet with tops removed were c.690 g N<sub>2</sub>O-N ha<sup>-1</sup> increasing to a mean of c.2000 g N<sub>2</sub>O-N ha<sup>-1</sup> (Table 37) with incorporation of sugar beet tops (c.80 kg N ha<sup>-1</sup> of N returned), giving an EF of 1.7% (net cumulative N<sub>2</sub>O, incorporated – removed, as a % of removable N-returned). If sugar beet was excluded from the ANOVA there was a significant crop effect ( $P < 0.001$ ; LSD = 244), but the residue effect was small and not significant ( $P > 0.05$ ); residues increased mean emissions by c.55 g N<sub>2</sub>O-N ha<sup>-1</sup> y<sup>-1</sup>. This amounted to about 0.17% of average removable N, but was calculated from very different residue amounts (Table 38) – the amounts of crop residue returned were only significant for vining peas (140 kg N ha<sup>-1</sup>), spring beans (20 kg N ha<sup>-1</sup>) and spring OSR (27 kg N ha<sup>-1</sup>).

**Table 37. Effect of crop type and incorporation of removable residues on (a) cumulative annual N<sub>2</sub>O measured over 12 months after incorporation at three sites; (b) residue effects are shown with EFs (% of removable N returned) in brackets where significant.**

Crop	Removable Residue	Gleadthorpe 2011	Edinburgh 2011	Terrington 2012	Crop Mean
(a)		<i>Cumulative N<sub>2</sub>O (g N<sub>2</sub>O-N ha<sup>-1</sup>)</i>			
Winter beans	Incorporated	816	3,877	545	1,900
	Removed	1,271	4,466	426	
Spring beans	Incorporated	1,089	4,286	387	1,860
	Removed	1,239	3,814	345	
Combinable peas	Incorporated	999	3,949	339	1,691
	Removed	725	3,807	329	
Vining peas	Incorporated	1,866	3,664	602	2,106
	Removed	1,807	4,255	440	
Winter wheat 0N	Incorporated	985	4,456	388	1,899
	Removed	959	4,317	286	
Sugar beet +N	Incorporated	2,038	3,759	869	2,031
	Removed	689	4,447	382	
Winter Wheat +N	Incorporated	1,404	3,728	447	1,827
	Removed	1,315	3,559	511	
OSR +N	Incorporated	1,207	5,005	402	1,819
	Removed	1,179	2,742	377	
(b)		<i>Net cumulative N<sub>2</sub>O due to residue incorporation (g N<sub>2</sub>O-N ha<sup>-1</sup>)</i>			
Winter beans		-454	-589	119	
Spring beans		-150	472	42	
Combinable peas		274	142	11	
Vining peas		60	-591	162	
Winter wheat 0N		26	139	102	
Sugar beet +N		1,349	-688	487	
	(EF % of total N returned)	(1.7%)	not sig.	(0.7%)*	
Winter Wheat +N		89	169	-63	
OSR +N		29	2,263	24	

\* When taking the residue effect to be real at Terrington; residue effect  $P < 0.05$ , however, note crop x residue interaction  $P = 0.07$ .

### **Edinburgh 2011**

There were no significant crop or residue effects on cumulative N<sub>2</sub>O emissions at this site ( $P > 0.05$ ). Background emissions were very large, with mean cumulative N<sub>2</sub>O measured from winter wheat (0N) of c.4400 g N<sub>2</sub>O-N ha<sup>-1</sup> (Table 37). The lack of effect was despite large amounts of removable N being returned and incorporated especially from combinable peas at c.400 kg N ha<sup>-1</sup>

and vining peas at c.350 kg N ha<sup>-1</sup> (Table 38). Wet soil conditions at the Edinburgh site for much of the sampling period were likely to have contributed to the high emissions. It must be concluded that the large background emissions masked any crop residue effects that may have occurred at this site.

**Table 38. Mean amount of N returned (kg ha<sup>-1</sup>) in removable crop residues at each site.**

Crop	Removable Residue	Gleadthorpe 2011	Edinburgh 2011	Terrington 2012
<i>Mean N returned (kg ha<sup>-1</sup>)</i>				
Winter beans	Incorporated	ND	116	206
	Removed	ND	15	18
Spring beans	Incorporated	22	76	114
	Removed	2	9	10
Combinable peas	Incorporated	10	404	124
	Removed	3	28	11
Vining peas	Incorporated	138	349	140
	Removed	0	0	60
Winter wheat 0N	Incorporated	5	11	24
	Removed	3	2	15
Sugar beet +N	Incorporated	83	211	82
	Removed	0	34	10
Winter Wheat +N	Incorporated	26	34	66
	Removed	25	6	28
OSR +N	Incorporated	65	86	108
	Removed	38	13	70

\*nd = not determined

### **Terrington 2012**

Immediately following incorporation at this site soil conditions were extremely wet (WFPS >80%). There was a significant effect of crop ( $P < 0.05$ ; LSD = 166) and residue removal ( $P < 0.05$ ; LSD = 83) on cumulative N<sub>2</sub>O emissions but no significant crop x residue interaction ( $P = 0.07$ ) (Table 37). As at Gleadthorpe 2011 the effect of removable sugar beet residues was large, the mean cumulative emissions from the removed plots being c.380 g N<sub>2</sub>O-N ha<sup>-1</sup> and increasing to 870 g N<sub>2</sub>O-N ha<sup>-1</sup> on the incorporated plots; however the crop x residue interaction ( $P = 0.07$ ) was not quite significant. At Terrington (unlike at Gleadthorpe) the number of fallen beet leaves was estimated to add approximately 10 kg N ha<sup>-1</sup> to both the removed and incorporated treatments; this may have added to the variability. If this residue effect was treated as real, the EF associated with incorporating the sugar beet tops was 0.7% of total N returned. The effects of crop residues other than sugar beet were much smaller at ~55 g ha<sup>-1</sup> yr<sup>-1</sup> N<sub>2</sub>O-N and not significant, so it would not be realistic to estimate an EF for their removable N.

## 7.4. Discussion

### 7.4.1. N<sub>2</sub>O emissions due to N fixation

It is clear that the pulses (particularly those grown at Terrington and Edinburgh) must have fixed large quantities of N during growth; not only did these crops contain large quantities of N at harvest but at Terrington and Edinburgh their N legacies, measured as SNS captured by following unfertilised crops of winter wheat, were greater than those of any of the fertilised crops (Table 37). That these legacies were evident even without the return of removable residues suggests that a significant part of the fixed N was held below ground. Despite fixing large quantities of N, there was no evidence to suggest that the annual pulse crops tested in these experiments emitted significant amounts of N<sub>2</sub>O simultaneously with BNF. At Terrington, active (red or pink) N-fixing nodules were most frequent in April and May but this did not correspond to a peak in N<sub>2</sub>O emissions. This finding is consistent with the conclusions made by Rochette and Janzen (2005) as well as more recent studies that have examined N<sub>2</sub>O fluxes from agricultural leguminous crops in order to assess the N<sub>2</sub>O EF associated with BNF. Rochette and Janzen (2005) concluded that the cumulative emissions from annual legumes was approximately 1 kg N<sub>2</sub>O-N ha<sup>-1</sup> (not much greater than average background emissions) and that N<sub>2</sub>O emissions are most likely to arise due to the N released from root exudates during the growing season and from crop residue decomposition after harvest. In the studies reviewed by Rochette and Janzen (2005) background emissions were not typically measured; instead Rochette and Janzen (2005) used average values of 1 kg N ha<sup>-1</sup> (as proposed by Bouwman, 1996) as a 'background' comparison. However, as shown here in both the residue and fertiliser experiments, background emissions can vary greatly, being influenced by soil moisture, soil type and cropping history (Section 6.6.2). In a pot experiment, Yang and Cai (2005) demonstrated that significant N<sub>2</sub>O emissions from legumes were most likely due to the decomposition of roots and nodules which normally occurs in the later stages of growth. N<sub>2</sub>O emissions were measured up to 117 days post emergence of soybean and compared to emissions from fallow soil (i.e. a measure of background). It was found that soil N<sub>2</sub>O emissions were stimulated after premature plant destruction, but in the full-term crop they were significantly greater (P<0.05) than bare soil only during the final grain filling stage (a 13 day period).

When examining the time-courses of SMN in the top 10cm here, it appears that there was significant mineralisation of fixed N (i.e. greater than observed on the winter wheat 0N control) once the crop had started to senesce (e.g. winter beans at Terrington; Figure 34). The mean trend across all pulse crops at Gleadthorpe and Terrington was for increases in soil NO<sub>3</sub>-N before harvest of up to c.10 and c.20 kg NO<sub>3</sub>-N ha<sup>-1</sup> respectively. Increases in topsoil NH<sub>4</sub><sup>+</sup>-N were also observed at Edinburgh prior to harvest, with peaks up to 20 kg NH<sub>4</sub><sup>+</sup>-N ha<sup>-1</sup> (data not presented).

At Terrington, the daily N<sub>2</sub>O graphs (Figure 33 and Figure 35) reveal that emissions from winter beans increased approximately one month before harvest, corresponding to the increases in

topsoil  $\text{NO}_3\text{-N}$ . Furthermore, root nodule assessments taken approximately three weeks before harvest (22<sup>nd</sup> August 2011) show an 80% decrease in nodule numbers compared to the previous assessment conducted on 26<sup>th</sup> June 2011. These findings suggest that the breakdown of root nodules and the mineralisation of their N-contents is likely to have increased soil  $\text{NO}_3\text{-N}$ ; when combined with soil conditions ideal for  $\text{N}_2\text{O}$  production (i.e. wet and warm) this resulted in cumulative  $\text{N}_2\text{O}$  emissions from winter beans (c.830 g  $\text{N}_2\text{O-N ha}^{-1}$ ) being significantly ( $P < 0.05$ ) greater than background. Root nodule density measured during the growing season maybe a useful tool for predicting the subsequent  $\text{N}_2\text{O}$  emission during crop senescence. For instance, the data show that active (pink and red) nodules were most frequent on winter beans followed by combinable peas. Correspondingly, the  $\text{N}_2\text{O}$  emissions (from drilling to harvest) were greatest from winter beans followed by combinable peas. With vining peas at both Gleadthorpe and Terrington small increases in  $\text{N}_2\text{O}$  emissions occurred just after harvest (Table 34). At Gleadthorpe it is likely that the additional shallow incorporation immediately following vining pea harvest helped to stimulate greater  $\text{N}_2\text{O}$  emissions from root and nodule decomposition, while at Terrington the soil was not cultivated (due to wet conditions).

The experiments conducted here are unique in that they measured  $\text{N}_2\text{O}$  emissions from four different annual leguminous species, and included a measure of background, with different weather conditions and soil types. Thus these experiments have uniquely been able to conclude that both the magnitude and timing of  $\text{N}_2\text{O}$  emissions differs between annual leguminous crop species and also between sites. Overall, these findings support the rather less well substantiated conclusions of Rochette and Janzen (2005), and Yang and Cai (2005) who supposed that, rather than  $\text{N}_2\text{O}$  emissions originating directly from the BNF process of annual legumes, significant  $\text{N}_2\text{O}$  emissions may originate from root exudates during the growing season, and following the decomposition of roots and nodules and the mineralisation of their N-contents.

The timing of  $\text{N}_2\text{O}$  emissions demonstrates that below ground biomass, possibly decomposing root nodules, can act as a source of  $\text{N}_2\text{O}$  and significant emissions can occur before harvest and post-harvest, prior to autumn cultivation. As indicated by the results of winter beans at Terrington, it would appear incorrect to assume zero emissions before harvest (as with IPCC, 2006). Overall, these results argue for the collection of further evidence to work out better ways of characterising  $\text{N}_2\text{O}$  emissions associated with BNF by annual legumes. In particular, they suggest distinguishing between  $\text{N}_2\text{O}$  emissions originating from root exudates, root nodules and the roots themselves. Past studies of protein composition of below ground tissues of leguminous crops such as clover and soybeans (e.g. Wu and Arima, 1993; Gordon and James, 1997) show there to be an extreme contrast in N concentrations between root biomass and nodule biomass from 0.4 to 5-12 mg  $\text{g}^{-1}$  fresh weight (assuming 15%DM, these equate to 0.2% and 3-8%!). Thus a possible hypothesis is that significant leguminous emissions will mainly occur when plant material with a high N% (low

C:N) such as root nodules, decompose in soil, particularly soil which is warm and wet. This suggests that a new study would be worthwhile, focussing on N amounts, N concentrations and longevities of fibrous roots, root nodules, and other leguminous deposits in soil. Most roots and dead haulm hold little N, so N<sub>2</sub>O emissions may largely be associated just with decomposing nodules or green leaves.

#### **7.4.2. N<sub>2</sub>O emissions due to removable crop residues**

When looking at the post-incorporation data it is clear that the only crop to elicit effects of crop residue removal or incorporation was sugar beet. There is little evidence that residue removal / incorporation from other crops caused any detectable effects on N<sub>2</sub>O emissions (Table 37), despite quite large differences in the quantities of N being incorporated or removed (Table 38). An effect of crop was detected at both Gleadthorpe and Terrington with cumulative N<sub>2</sub>O emissions from vining peas being significantly greater ( $P < 0.05$ ) than background. Since there was no difference in emissions between the removed and incorporated subplots these emissions are best attributed to the decomposition of the below-ground plant biomass, as discussed in section 7.4.1. Whilst there may be some effects of other residues 'hidden' within the variability in the large background emissions found at the Scottish site, no differences were significant.

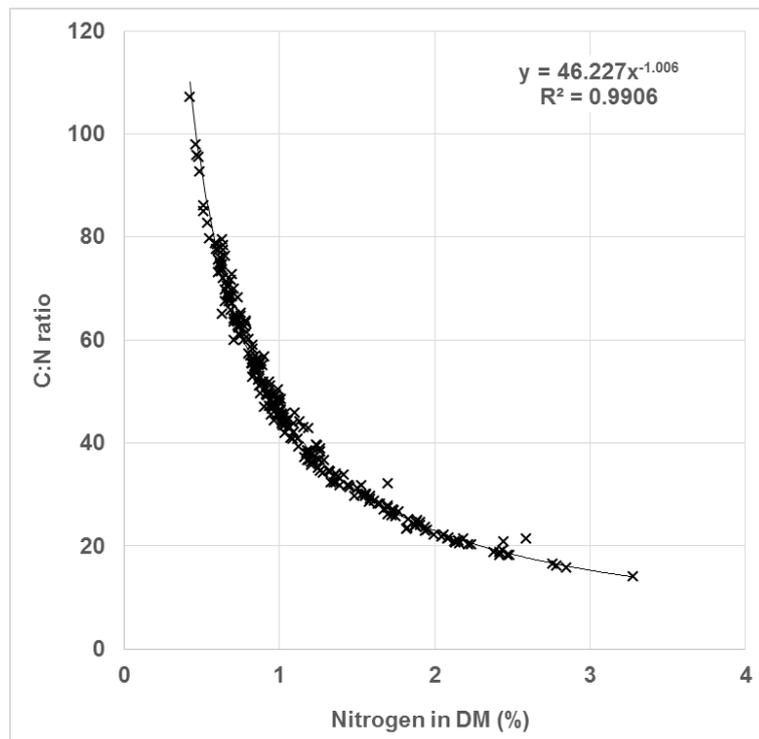
The main practical finding that can be taken from these experiments in relation to removable residues appears to concern sugar beet; however, it may be wrong to consider this as a species-specific effect. Whilst there may be some complex or subtle effects of residue constitution on mineralisation, denitrification or N<sub>2</sub>O emission from sugar beet leaves, such as a low lignin content (Jensen 1989; Fox *et al.* 1990), the simplest explanation of the contrast between sugar beet tops and the other residues is in their N concentration; sugar beet leaves had the greatest N concentration (2.2%) of all the removable residues incorporated here (Table 39) and, other than values for vining pea haulm for which good data did not exist in the 1970s, published tables of tissue N concentrations (MAFF, 1980) tend to support the values observed for all crops here (Table 39).

**Table 39. N concentrations of removable residues from pulses and other non-leguminous crops grown either with (+N; at recommended rates) or without (0N) fertiliser N in experiments at Gleadthorpe and Terrington reported here, or published by MAFF (1980).**

Crop	MAFF RB2087	Gleadthorpe & Terrington	SE
Sugar beet leaves (+N)	2.0	2.21	0.00
Vining pea haulm including pod walls	2.7-3.3*	1.55	0.03
Winter Field Bean straw and pod walls	} 0.83	1.48	ND
Spring Field Bean straw and pod walls		1.29	0.05
Combining Pea straw and pod walls	1.68	1.26	0.47
Oilseed Rape straw (+N)	0.48	0.75	0.01
Winter Wheat straw (+N)	} 0.38	0.67	0.03
Winter Wheat straw (0N)		0.56	0.08

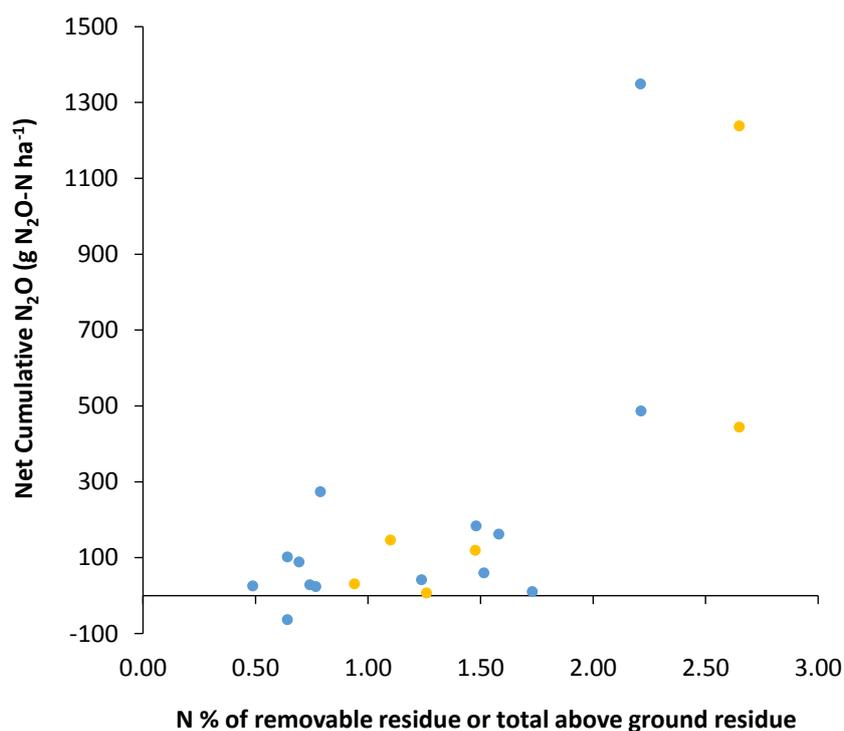
\* Values for 'Pea haulm and pods (canning) ensiled' and 'Peas beginning to flower', respectively

Since N concentration totally dominated the variation in C:N ratio of removable residues tested here (i.e. variation in carbon concentration was almost entirely inconsequential; Figure 36), this high N% is entirely consistent with the suggestion that only residues with small C:N ratios will mineralise their N fast enough to cause detectable N<sub>2</sub>O emissions (Baggs *et al.*, 2000). Baggs *et al.* (2000) in a field experiment conducted in Fife, East Scotland reported the highest emissions from N-rich lettuce residues (C:N = 7.5:1) with 1,100 g N<sub>2</sub>O-N ha<sup>-1</sup> being emitted in the first 14 days following incorporation. An experiment conducted on a sandy loam soil at Stetchworth in East Anglia compared emissions following the incorporation of residues from spring barley, peas, sugar beet and a fallow treatment (Harrison *et al.*, 2002). Although, the results are not fully comparable with those here, due to lack of an equivalent 'removed' treatment, results showed that sugar beet residues (compared to spring barley and pea residues) had the lowest C:N ratio (15-17:1) and the greatest N<sub>2</sub>O emissions (Harrison *et al.*, 2002). N<sub>2</sub>O emissions following the incorporation (in either October or January) of sugar beet tops (and root crowns) in both years of the experiment were significantly greater ( $P < 0.05$ ) compared to a no residue treatment. Consistent with the analyses of sugar beet residue used in this experiment, the N% of the returned sugar beet residue (tops and crowns) in Harrison *et al.* (2002) was greater than 2%, whilst for peas and spring barley the N% of the total residue returned was closer to 1% (ADAS unpublished data).



**Figure 36. Relationship between C:N ratio of removable crop residues from experiments at Gleadthorpe and Terrington, and their N%. The fitted line indicates that, for estimation of C:N ratio, carbon content of a wide range of residues can be assumed to be 46.23%, obviating the need for carbon analysis.**

Figure 37 shows the results from Gleadthorpe and Terrington (in blue), for net (incorporated minus removed) cumulative emissions ( $\text{g N}_2\text{O-N ha}^{-1}$ ) plotted against the N% of removable residue. Results from Harrison *et al.* (2002) are also included (in orange) to show the net cumulative  $\text{N}_2\text{O}$  emissions from the incorporation (in October) of spring barley, peas and sugar beet at Stetchworth minus emissions from the fallow treatment; these are plotted against the N% of the total above ground residue. Figure 37 demonstrates firstly the consistency of results from Gleadthorpe and Terrington with those reported by Harrison *et al.* (2002), in that the highest  $\text{N}_2\text{O}$  emissions are associated with the residues having an N concentration greater than 2%. Secondly Figure 37 confirms that there is large variability in the net amount of  $\text{N}_2\text{O}$  emitted following incorporation of sugar beet tops, most likely reflecting the impacts of residue mixing, soil type and soil conditions (moisture and temperature) on  $\text{N}_2\text{O}$  emissions. Pooled together, the data from Harrison *et al.* (2002) and from these experiments also demonstrate a gap in available data with no measurements after incorporation of residues with an N% between 1.7% and 2.2%.



**Figure 37. Relationship between net cumulative N<sub>2</sub>O emissions and N% of removable residue at Gleadthorpe and Terrington (blue points) or N% of total above ground residue at Stetchworth, UK reported by Harrison et al. (2002).**

It should be noted that although C:N ratio has been advocated as the appropriate predictor of potential mineralisation, for practical purposes N% is cheaper and easier to measure, and much more widely understood by the agricultural industry. It would thus seem that N% should be adopted as the preferred predictor of potential N<sub>2</sub>O emissions from crop residues, with a threshold of about 2%. However, without further investigation, it is unclear whether the high N% content of sugar beet leaves was interacting with other chemical characteristics of the residue (such as availability of carbon) resulted in significantly higher N<sub>2</sub>O emissions or not. For instance Baggs *et al.* (2000) suggested that high water content of a residue with a low C:N content might create anaerobic microsites which might exacerbate N<sub>2</sub>O emissions following its incorporation.

However, based only on evidence available to date, we believe the initial N% of crop residues (as measured at harvest) should be taken as the most useful predictor of potential N<sub>2</sub>O emissions; we would distinguish this from N concentrations of partly decomposed biomass, since N% is likely to increase as decomposition proceeds (due to organic carbon being degraded). Kriauciūnienė (2012) utilised litter bags of 10 different crop residues with initial C:N ratios ranging from 25 to 113 in a field experiment in Lithuania, and reported that c.40 to c.90% of the organic carbon was decomposed within the first 14.5 months after autumn incorporation (September); the most rapid decomposition occurred from March to September (months 7.5 to 14.5), coinciding with warm temperatures. Furthermore, it was found that N concentration of residue could increase for up to

14.5 months of decomposition (Kriaučiūnienė, 2008). The experiments conducted here ran for 12 months, similar to the period of rapid residue decomposition reported by Kriaučiūnienė (2012). The fact that significant N<sub>2</sub>O emissions were measured only from sugar beet and not from the other crop residues indicates that increases in N% of residues as decomposition progresses does not necessarily lead to measurable N<sub>2</sub>O emissions.

Evidence so far suggests that removable crop residues with less than 2% N will not cause significant N<sub>2</sub>O emissions in the first 12 months after their incorporation. The ensuing and more difficult question is whether the incorporation of such residues can have a longer-term effect on N<sub>2</sub>O emissions, essentially contributing to 'background' emissions in future years? It is possible that differences between residue management strategies might eventually lead to new equilibria between carbon and N in the soil, hence to different propensities to mineralise and emit N<sub>2</sub>O; however, experience of long term experiments is that such differences can take decades or even centuries to effect, so any questioning of how most crop residue N should be expected to affect N<sub>2</sub>O emissions will best be resolved by philosophy rather than by experiment.

Although there was no effect of removable legume residues on N<sub>2</sub>O emissions, legume effects were evident before harvest, from harvest to incorporation and in the 12 months after incorporation. When summing the N<sub>2</sub>O emissions from harvest through to 12 months from incorporation the results show that vining peas emitted significantly more N<sub>2</sub>O than background (winter wheat 0N) at both Gleadthorpe ( $P < 0.001$ ; LSD = 306) and Terrington ( $P < 0.02$ ; LSD = 340), emissions from vining peas being approximately c.900 and c.600 g N<sub>2</sub>O-N ha<sup>-1</sup> greater than background respectively. These elevated emissions most likely arise from decomposition of below-ground residues, particularly the nodules (section 7.4.1).

The lack of significant differences between other crops demonstrates that, despite the high amounts of N these crop may sequester below ground (section 7.3.6), they do not necessarily lead to significant emissions. IPCC (2006) uses below ground N as a component of its estimates of N<sub>2</sub>O emissions originating from the breakdown of residues. However, there are almost insuperable difficulties in measuring amounts of below-ground N and, as results here demonstrate, even if much N is retained below ground, this does not always lead to significant N<sub>2</sub>O emissions.

#### **7.4.3. Approaches to estimating crop residues and their N<sub>2</sub>O emissions**

IPCC advocates fairly complex methods of estimating N<sub>2</sub>O emissions from crop residues (detailed in Section 5.2.3). Since no significant differences in emissions were detected here that were not related to decomposition of leguminous roots (and nodules) or leafy material with a high N content, and since this seems consistent with wider observations reported in the literature, there seems to be justification for changing and simplifying the methodology considerably. This section considers

possible issues and solutions within the existing approach, whilst Sections 7.5 and then 10.2.4 point towards possible new approaches.

The data collected here allow comparisons of measured crop residues with values estimated using IPCC methodologies. The results shown in Figure 38 are not encouraging, so causes of the various discrepancies can now be considered.

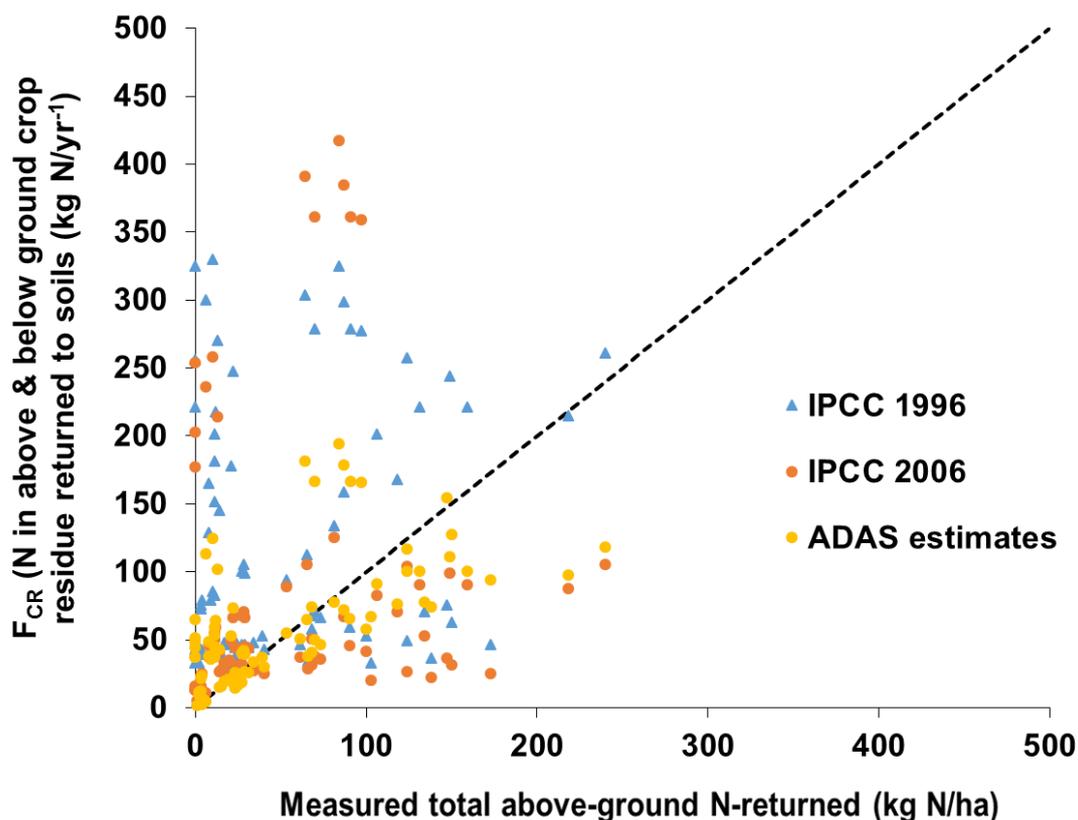


Figure 38. Relationship between total amounts of above ground N returned (excluding harvested N) at Gleadthorpe ( $\blacktriangle$ ), Terrington ( $\blacksquare$ ) and Edinburgh ( $\bullet$ ) and IPCC estimates using (a) 1996, (b) 2006 and (c) 2006 modified by UK evidence (ADAS). IPCC (1996) estimates exclude below-ground residues and employ a default value for the fraction of crop residues removed. IPCC (2006) estimates include above and below ground residues, but experimental data are used for the fraction of crop residues removed.

IPCC has always based its above-ground crop residue estimates on crop yield, but the IPCC (2006) methodology uses a more complex calculation than IPCC (1996) to get from crop yield to crop residues, involving slopes and intercepts of regression relationships between harvestable DM and above-ground residue DM. IPCC (2006) does not provide a default value for the removable fraction of above-ground crop residue N ( $FRAC_R$ ) stating that, if expert data are not available, removed residues should be assumed to be zero, hence ensuring that emissions are not underestimated.

IPCC methods also involve estimating emissions from below-ground residues. The IPCC (2006) approach requires crop specific data for the N contents of the above ground residues, the relationship between yield and residues both above- and below-ground, and the N% of below ground residues. Default values are given for a range of crops as detailed in Table 10. However, not all crops are covered. For instance, the default values for root crops are based on groundnuts, hence are not relevant to crops such as sugar-beet in the UK, and values for OSR are not included.

Initial 'best estimates' for crop residues from wheat, OSR, sugar beet and vining peas were reported in Table 10 based on an IPCC (2006)–like approach but using harvest indices to estimate residue biomass from crop yields. However, in making the estimates presented in Section 5.2.3 we encountered further large uncertainties in other parts of the IPCC approach. For instance, we took N concentrations from standard feed analysis references (MAFF, 1980; Table 39) but these dated from the 1970s when N use on most crops was less than now; these values tended to appear low. They could be improved for modern crops at a modest cost by surveying laboratory analyses of crop residues. In this regard, it is noteworthy that the mean N% of vining pea residues in experiments at Gleadthorpe and Terrington was 1.55%, much less than sugar beet leaves, despite both being green when cut. This result contrasts with the values of 2.7 to 3.3 reported by MAFF (1980), and suggests that a survey of further samples of vining pea haulm would be worthwhile.

Rooting data are very sparse for all crops in the UK (also worldwide) and root DM is almost impossible to measure because it is not feasible to harvest complete root systems of plants grown in the field. In any case, root DM produced throughout the life of a crop tends to exceed markedly the DM present at any one time, since roots (especially fine roots) commonly have a significantly shorter life-span than the crop itself (King *et al.*, 2001). The IPCC (2006) default figures use slightly different ratios for above ground (AG) to below ground (BG) biomass for different crops, and tissue N% also differs. Both the AG:BG ratio and BG tissue N% are *less* for pulses than other crops. As IPCC (2006) estimates of root biomass are directly linked to crop yield, and pulse crop yields tend to be relatively low, final estimates of BG residue N from legumes were typically c.10-20 kg ha<sup>-1</sup> for crops here, whilst for sugar beet the estimate of BG N was greater at c.50 kg ha<sup>-1</sup> (depending on the assumptions used). This is contrary to the evidence from experiments here viz. the higher than background SMN and N<sub>2</sub>O emissions from vining peas at Terrington and Gleadthorpe and winter beans at Terrington (without incorporation of above ground residues).

IPCC (2006) estimates of root biomass are based on crop yield. Although crop yield may indicate the amount of N-fixed by legumes, ultimately N-fixed is determined by interactions between plant growth and the soil N environment; with SMN depressing both nodulation and N<sub>2</sub> fixation (Streeter, 1988; Waterer and Vessey, 1993, Unkovich and Pate, 2000). In addition, the yields of legumes

may not indicate reliably the amounts of N held in below ground biomass. Unkovich and Pate (2000) reported that, while leguminous crops may grow significant amounts of below ground biomass (including both roots and nodules), species vary significantly; of the 10 different legumes tested by Unkovich and Pate (2000) below ground N ranged from c.25% to c.50% of total plant N.

The paucity of good data on root biomass and tissue N% for different crops makes discrimination between crops questionable. Recent data for rooting of wheat and OSR show that root biomass is normally around 1 t ha<sup>-1</sup> for UK crops to 90cm depth (Gregory *et al.*, 2005; White *et al.*, 2014; ADAS unpublished data). We therefore suggest that a common ratio of 0.2 for below ground to above ground biomass would be as much as can be supported by existing evidence. Similarly evidence for N% of root material differing between crops is sparse, so a common figure of say 0.9% N for all crops would be most defensible (Pederson *et al.*, 2002). Evidence of greater SMN and N<sub>2</sub>O emissions at the time of root breakdown of pulses in the experiments here suggests that tissue N% of legumes was substantially greater than other crops. However, IPCC default values for pulses give *lower* root N% concentrations and *lower* root biomass. If we are to fairly account for the N<sub>2</sub>O emissions from growing pulse crops in the UK we need better information on the N% (and size) of legume roots. Such data could be obtained by sampling and analysing roots of commercial crops in a structured survey.

## **7.5. Conclusions**

### **7.5.1. Legumes**

Despite fixing large quantities of N, there is no evidence to suggest that the pulses tested here emitted significant amounts of N<sub>2</sub>O in direct association with their BNF. However, the pulses did cause N<sub>2</sub>O emissions significantly greater than background, and it appears that these originated from below ground during crop senescence (e.g. winter beans at Terrington) both before and after harvest, when decomposition and mineralisation of their below-ground residues must have been taking place (e.g. vining peas at Terrington and Gleadthorpe). Given their extremely high N concentrations, it would appear that nodules are highly likely to give rise to localised concentrations of mineral N in the soil when they start to senesce, lose their integrity and then decompose. This will primarily occur as these crops reach maturity (as with dried peas and beans here), or are defoliated (as with vining peas here, but also with forage legumes such as clovers and vetches). Whilst the detailed dynamics of nodule growth and decomposition (and then N<sub>2</sub>O emission) are difficult to imagine, it is clear that further research to quantify the effects of agricultural practices on legume nodules and N<sub>2</sub>O emissions should prove worthwhile. Given the extent of past research on leguminous species, it will initially be worth reviewing factors affecting nodule biomass production, N concentration, longevity and decomposition. However, studies including experimentation are likely to be required to understand nodule N mineralisation, and then soil transformations (including the evolution of N<sub>2</sub>O) and plant acquisition of this mineralised N. At

present there is little evidence to resolve which legume crops will cause an emission or to estimate how large these emissions might be. Whilst pulse growth was poor at Gleadthorpe, vining peas did give a significant emission both here and at Terrington (of c.0.6 to c.1.0 kg N<sub>2</sub>O-N ha<sup>-1</sup>), whereas dried peas and spring beans did not. However, it would seem wrong to assume that all pulses which are allowed to mature are able to recover the N mineralised from their nodules since the winter bean crop at Terrington clearly showed a significant emission of c.0.8 kg N<sub>2</sub>O-N ha<sup>-1</sup>. There is no indication here that these emissions related quantitatively to the yield or N taken up by these crops, as is assumed by IPCC. One possible approach might be to assume that defoliated pulses will cause a standard emission of say c.0.8 kg N<sub>2</sub>O-N ha<sup>-1</sup>, whereas pulses allowed to mature 'naturally' will cause no significant emission (NB. the significant effect of winter beans at Terrington was only one of nine dry pulse crops tested here).

### 7.5.2. Crop residues

Following autumn cultivation, it is clear that the only crop to elicit measurable effects of crop residue *removal* or *incorporation* was sugar beet. There was little evidence that residue removal / incorporation with other crops caused any detectable effects on N<sub>2</sub>O emissions, although it is possible that the N concentrations in some of these residues (e.g. vining pea haulm) were atypically low. The simplest explanation of the contrast between sugar beet tops and the other residues tested in this project is in their N concentrations, sugar beet leaves being greatest (2.2%). We suggest that the initial N% content of a residue, rather than its species, should be taken as the best predictor of those crop residues having potential for N<sub>2</sub>O emissions, and that the threshold should be of the order of 2%. In the short term, since only residues with >2% N had an effect on N<sub>2</sub>O emissions, and that the only two EFs measured were 0.7% and 1.7% of N returned, we suggest that GHG accounting systems should continue to use the IPCC EF of 1% for these materials.

On the other hand where crop residues have significantly less than 2% N, such as from cereals, oilseeds, and pulses harvested mature, the indications here are that N<sub>2</sub>O accounting systems should ideally treat any emissions from such residues (if they are deemed to occur) as being small and long term, perhaps through ascribing them to an overall 'background' emission. In addition, since most below-ground residues originate from roots and root materials, and since IPCC (2006) defaults themselves show these to contain significantly less than 2%N (0.8%, 0.9% or 1.4% for pulses, cereals and groundnuts respectively), it seems that most below-ground materials should also be treated differently in N<sub>2</sub>O accounting procedures. Some recent evidence suggests that decomposition of grassland root residues does differ between species (Smith *et al.*, 2014), however rates are slow nevertheless, and we know of no such evidence for arable crops. [We will not debate here whether low N% materials (both above- and below-ground) should be considered as causing soil-carbon sequestration and therefore should be included as net contributors within

broader GHG accounting systems.] Thus in future if estimation of crop residue N is to be used within UK GHG accounting systems, it would seem important that the calculations are simplified and rationalised so that they focus on crops and cropping systems that involve soil-incorporation of green leafy materials.

In addition to the haulm of vining peas and the tops of sugar beet plants, the main occasions when agriculture involves destroying green tissues and / or incorporating them into the soil include:

- Destruction of green cover crops or catch crops such as rape, kale, Quinoa and fodder radish,
- Pre-harvest destruction of the green haulm of potatoes, carrots and other root crops,
- Harvest of brassicas and other vegetable crops, where leafy material is left in the field e.g. cauliflowers, cabbages and green salads,
- Occasional frost damage of sensitive tissues of species such as oats, barley, clover, lettuce, potatoes, asparagus, celery, cauliflowers, and broccoli,
- Composting or field disposal of leafy materials e.g. lawn mowings, or residues from glasshouse crops,
- When clover-containing swards are cut, their nodules are likely to senesce and create potential emissions,
- When making silage, particularly from leaves high in nitrate (such as occur in droughted or dull conditions) 'silage gas' is emitted which principally contains NO<sub>2</sub> and CO<sub>2</sub> (and can create a lethal health hazard to farm operatives). However some N<sub>2</sub>O emissions are also possible (Heiniger and Dunphy, 2013), and
- Ploughing down of leafy grass leys (Ball *et al.*, 2007).

In each of these cases, some investigation into N<sub>2</sub>O emissions and, if positive, some attempt at quantifying those emissions would seem to be worthwhile.

## **8. Model validation – WP3**

### **8.1. Validating UK-DNDC with UK experimental data**

#### **8.1.1. Introduction**

The purpose of the site level modelling was to validate UK-DNDC with the MIN-NO (and other UK) experimental data. When the project proposal was written, and during the early stages of the project, the plan was then to use UK-DNDC in Work-Package 4 to ‘upscale’ from the experimental level to the levels of regions and products, and then to prioritise proposed measures for N<sub>2</sub>O mitigation in arable agriculture in the UK (in Work-Package 5). However, the results of model validations conducted here dictated that this and subsequent Work-Packages should achieve their objectives by a somewhat different course, as will be explained in this section.

DNDC and UK-DNDC are dynamic and deterministic simulation models of carbon and N biogeochemistry in agro-ecosystems with daily time steps (Li, 2000). These two models both simulate the same core processes, the key differences being that UK-DNDC was developed from DNDC to link to the UK databases that are required to run the model in this region. DNDC simulates several processes related to the N and carbon cycle (Smith *et al.*, 2010); it uses locality-specific weather data, and input data on carbon and N biogeochemistry, and predicts crop growth, soil carbon dynamics, N leaching, and emissions of trace gases including N<sub>2</sub>O (Li, 2000). Initially the model was developed for predicting carbon sequestration and trace gas emissions from upland agro-ecosystems in the United States (Li *et al.*, 1992). However, the model has been used much more widely for agricultural research and it now allows consideration of crop yields, rotations, tillage, fertiliser additions, and biomass removals under a wide range of agricultural conditions. Coding of UK-DNDC (as well as DNDC) has remained the responsibility of the model author (Li, 2000) during the MIN-NO project; thus any changes or clarifications concerning model operation that were required during the work reported here, involved interaction with the model author in the US.

#### **8.1.2. Approach**

The model validations here were focussed on assessing the ability of the model to predict N<sub>2</sub>O emissions from winter wheat experiments, based on site-level data. In order to stabilise the initial soil pools, the model was run for three years, with the crop of interest sown in the autumn of the second year. Field measurements of water-filled pore space (WFPS) and grain yield were used to guide manual calibration of the input values for WFPS at field capacity and permanent wilting point and the partitioning of carbon and N between grain and the above- and below-ground plant biomass. The model was validated against measured SMN contents and N<sub>2</sub>O emissions.

### 8.1.3. Sites and crops modelled

Runs of UK-DNDC concentrated on winter wheat crops grown at sites around England viz. Boxworth, Cambs. (harvest years: 2007, 2008, 2010, 2011, 2012), Terrington, Norfolk (2010, 2011), Gleadthorpe, Notts. (2008, 2012) and Debathe, Devon (2007, 2008). Data relating to the 2007 and 2008 harvests (from Defra Project AC0101) were also modelled to extend the available datasets and weather conditions. In addition to the work in the MIN-NO project, concurrent work was assessing DNDC and UK-DNDC in the Defra funded project AC0114. This focussed on assessing emissions across the country, based on MORECS squares (Hough and Jones, 1999), for grassland under standard management receiving no fertiliser and the standard level of fertiliser.

### 8.1.4. Results

It was found that WFPS could be modelled with a high degree of confidence (Figure 39), as long as site-specific and year-specific input values for WFPS at field capacity and permanent wilting point were used as inputs to the model. The model was able to simulate the yield responses to the different amounts of fertiliser, with the majority of points falling within the 95% prediction interval (Figure 40a). Similarly there were no site or year-related biases. However, modelling of annual N<sub>2</sub>O emissions showed a profound year-related bias, whereby only the emissions relating to harvests 2010 and 2011 were predicted accurately (Figure 40b). Deficiencies in the modelling of soil N concentrations were also detected. Specifically, soil N was over-estimated as harvest approached, but fell rapidly at harvest. This inadequacy within the model was more pronounced when the larger N applications (i.e. 200 kg N and greater) were modelled (Figure 41).

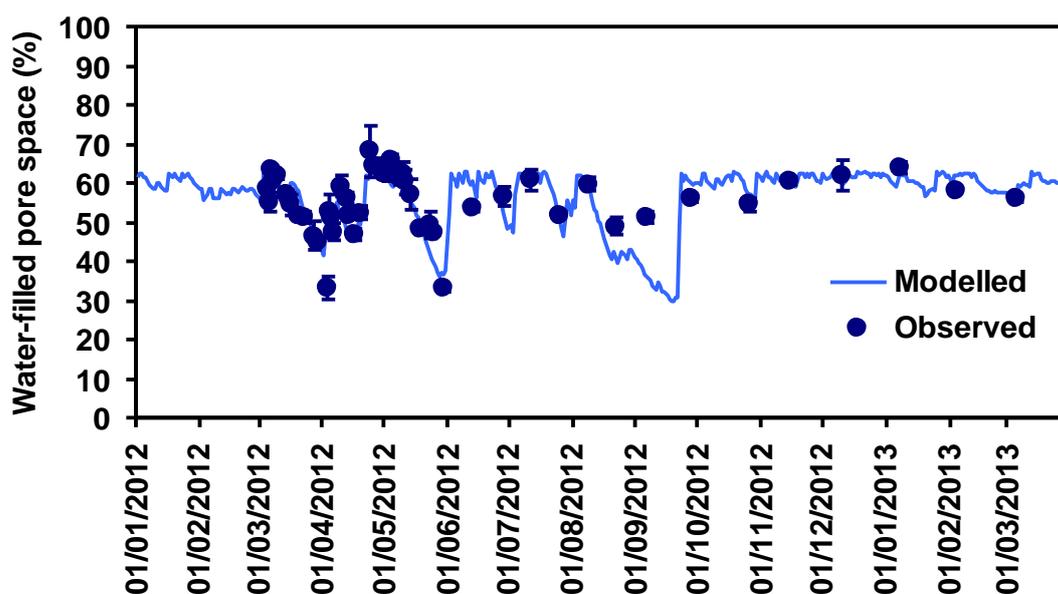


Figure 39. Water-filled pore space as measured in the MIN-NO experiment or modelled with DNDC for a winter wheat crop grown at Boxworth and harvested in 2012 (Section 6.5). Measured values are the means of three replicates and error bars show 2 standard deviations.

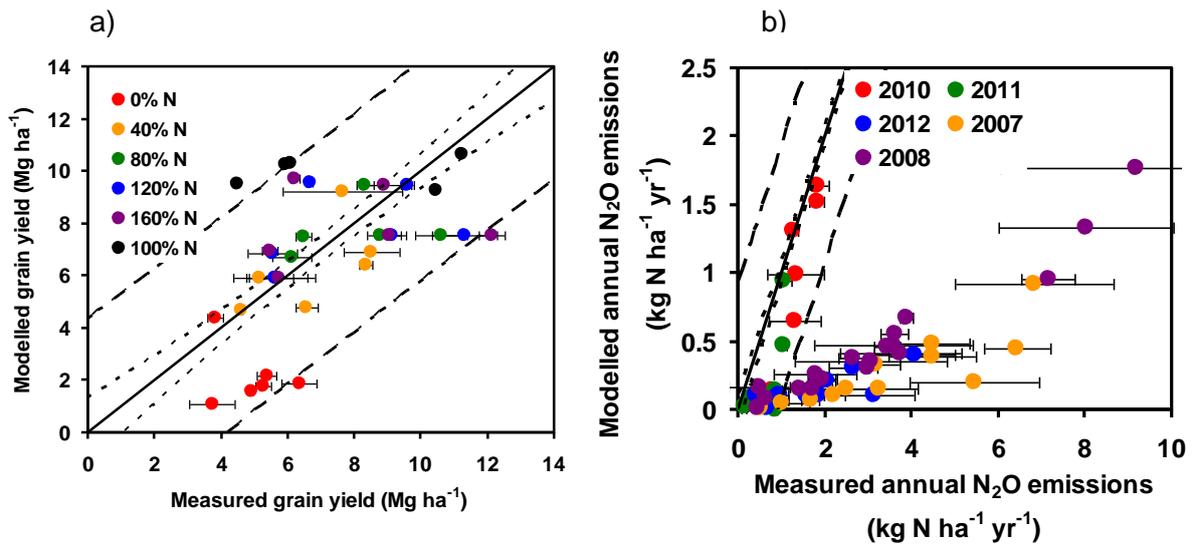


Figure 40. Parity plots for (a) grain yield and (b) N<sub>2</sub>O emissions for five harvests of winter wheat grown at the English sites. Measured values are the means of three replicates and errors bars show 2 standard deviations. Dotted boundary lines indicate the 95% confidence interval, dashed boundary lines indicate the 95% prediction interval.

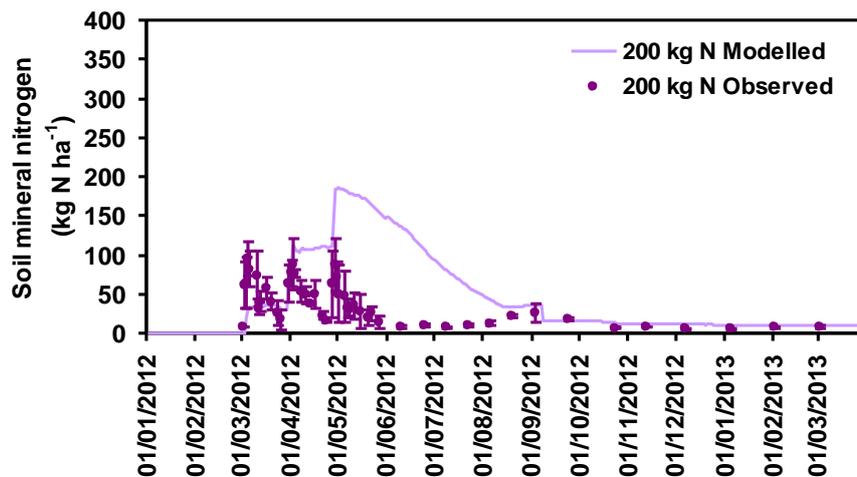
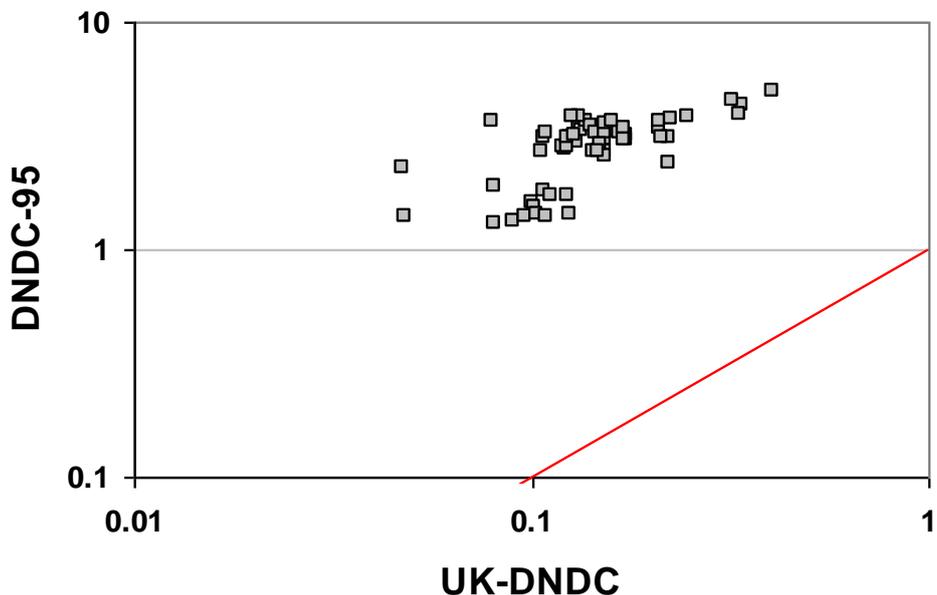
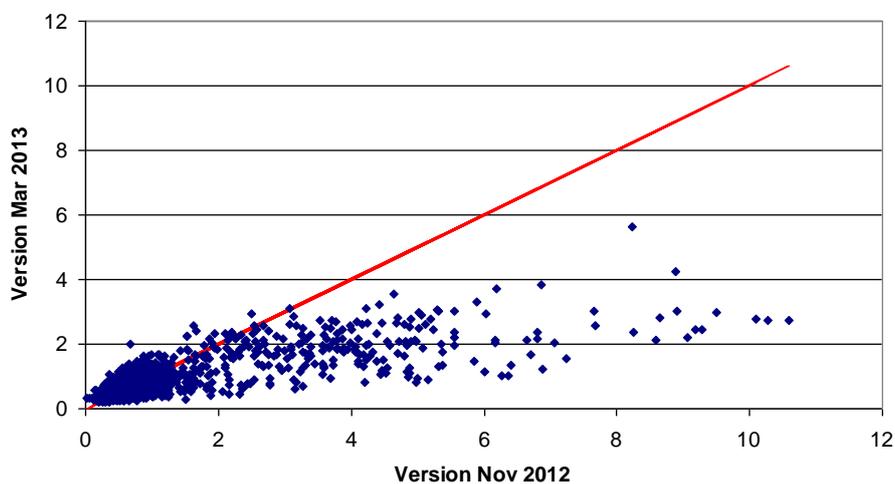


Figure 41. Total SMN to 10cm as measured in the MIN-NO experiment or modelled with DNDC for a winter wheat crop grown at Boxworth and harvested in 2012. Measured values are the means of three replicates; error bars show 2 standard deviations.

During the testing of DNDC and UK\_DNDC in the Defra AC0114 project, concerns were raised over differences between model outputs from different model versions. Examples of this are shown in Figure 42 and Figure 43, and causes are discussed below.



**Figure 42.** Disparity between annual N<sub>2</sub>O emissions (log transformed kg ha<sup>-1</sup> N<sub>2</sub>O-N) modelled for the same input dataset with the version DNDC95, and the version developed for use in the UK.



**Figure 43.** Disparity between annual N<sub>2</sub>O emissions (kg ha<sup>-1</sup> N<sub>2</sub>O-N) modelled for the same input dataset (riparian grass, 4% organic soil carbon, 150 kg ha<sup>-1</sup> N applied, pH 6.5) with versions of UK-DNDC provided in November 2012 and March 2013.

### **8.1.5. Conclusions**

The yield was modelled for the experimental sites with a degree of confidence; for N<sub>2</sub>O, there was a tendency for the model to under-predict emissions when measured emissions were high. However, the simulation of the soil mineral N was poor. This may be because the model has historically been validated against soil water, yield and N<sub>2</sub>O emissions, but without the generous availability of soil mineral N data that is provided by the MIN-NO dataset.

In addition to these findings in WP3, the testing that was carried out as part of the Defra Inventory project (AC0114) raised major concerns through further large discrepancies between predictions from different model versions. Version control was also poor: when minor changes were made to the model, a version history and documentation of the changes were not provided, preventing interpretation of the large differences in N<sub>2</sub>O emissions. In addition, having obtained the computer code for the model, analysis revealed equations with unresolved and serious errors. These findings were discussed with Defra and it was decided that work with the model should be discontinued until UK researchers could obtain closer control of the model code. Recognising that the broad principles simulated by the model are representative of conventional understanding, it was agreed that further modelling with DNDC might employ an independent version of the model called Landscape-DNDC (developed by the Institute of Meteorology and Climate Research, at Garmisch in Germany). [A modelling platform (GRAMP) was also established in collaboration with the Global Research Alliance on Agricultural GHGs to track model development and implement version control, but the benefits of that work were not available within the timeframe of this project.] In the short-term it was agreed that the MIN-NO project should terminate work with DNDC and use the remaining resource to develop statistical models of the annual emissions measured in the MIN-NO experiments so as to assess the effects of soil and weather on the emissions, and possibly to allow extrapolation regionally and nationally.

## **8.2. Statistical modelling**

### **8.2.1. Approach**

Statistical analysis of the 24 field experiments (see Section 6) was performed using REML regression (in Genstat version 16) to assess the relationships between annual cumulative annual N<sub>2</sub>O emissions (g N<sub>2</sub>O-N ha<sup>-1</sup> year<sup>-1</sup>), the fertiliser N applied, the weather and the soil properties. This analysis augments the analysis of individual results summarised in Table 24 and discussed in Sections 6.5 and 6.6 by more formally and objectively identifying and quantifying the main factors that account for most of the variation across the full dataset. However, this analysis does not address intra-seasonal dynamics of N<sub>2</sub>O emission, or the relationships between daily N<sub>2</sub>O emissions and soil variables such as soil mineral N and WFPS.

Many of the annual N<sub>2</sub>O emission values were close to zero so, to normalise their variance, as required for REML analysis, the N<sub>2</sub>O emission values were transformed using natural logarithms. The random factors included in the model were Site, Crop and Block, and the twenty-four experiments were also specified as a random factor. Negative annual emissions of approximately -700 g N<sub>2</sub>O-N ha<sup>-1</sup> year<sup>-1</sup> were observed for the winter OSR trial at SRUC in 2012, so 710 was added to all the values prior to log transformation. Although WFPS is known to be a key determinant of N<sub>2</sub>O emissions, it is not readily known on farms, so the proxy variables of rainfall and clay content (as a proportion) were adopted. Soil organic carbon, bulk density and pH were amongst the additional explanatory variables investigated. Soil characteristics for each experiment are shown in Table 19.

Having removed outliers, a range of models was fitted to the observations of cumulative annual N<sub>2</sub>O, using both multiple regression and REML, and REML models were tested with factors successively added in, or successively removed. The simplest, credible model was then selected.

### 8.2.2. Results

The experiment on winter OSR at Bush Estate in 2012 was excluded from the final analysis because residual plots identified data points from all treatments and blocks as extreme outliers. This included a large negative emission (less than -700 g N<sub>2</sub>O-N ha<sup>-1</sup> year<sup>-1</sup>). In addition, an outlier of -580 g N<sub>2</sub>O-N ha<sup>-1</sup> y<sup>-1</sup> was removed, relating to the control plot on block 2 from the spring barley at Bush Estate in 2011. [Soil N<sub>2</sub>O fixation of this magnitude is unknown.]

A range of models was fitted to the remaining 344 observations of cumulative annual N<sub>2</sub>O-N, using both multiple regression and REML (with factors successively added or removed). Model performance was assessed by inspection of scatter-graphs of predicted versus observed values, criteria being the precision of the predictions, lack of bias in any part of the observed range, and simplicity. The ultimate chosen model was as follows:

$$\ln(N_2O + 710) = 5.567(\pm 0.3490) + 0.002402(\pm 0.0001062) * N + 0.002349(\pm 0.0005046) * Rain + 0.02344(\pm 0.009793) * Clay\% - 0.003096(\pm 0.0014161) * Rain * Clay\%/100$$

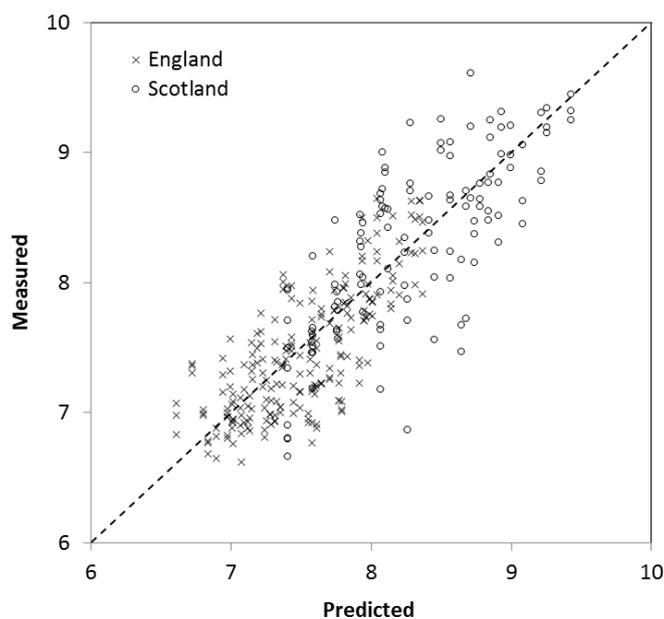
Equation 8

where *N<sub>2</sub>O* is the cumulative annual N<sub>2</sub>O-N emission (g ha<sup>-1</sup>), 710 is added to avoid any negative values, *N* is the rate (kg ha<sup>-1</sup>) at which fertiliser N is applied, *Rain* is the annual rainfall (mm, January to December) and *Clay%* is the percentage of clay in the soil.

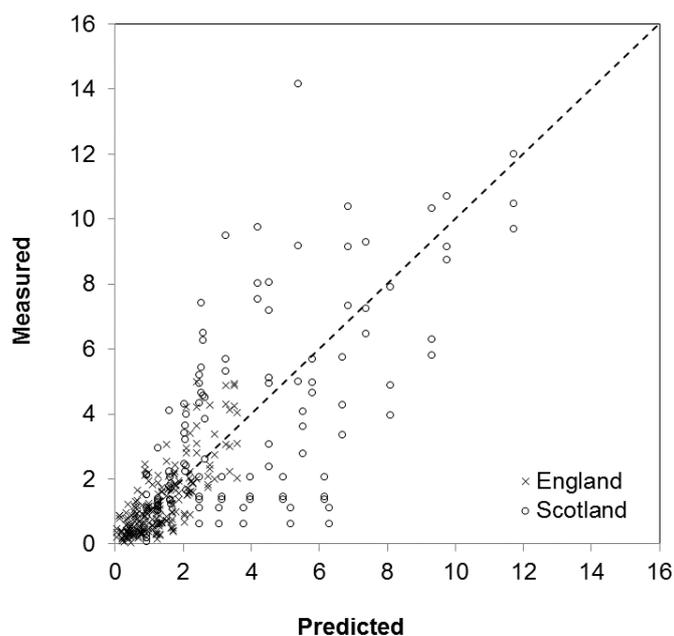
It was found that soil organic carbon and pH did not add to model performance, so these were excluded. Similarly a range of rainfall criteria was considered before annual rainfall was used as

the main indicator of wetness and thus soil anaerobism; this choice was made having observed that the main emission events occurred at any time through the year, and that the rainfall distribution through the year was inconsistent. Some judgement was used in the factors to include in the final chosen model because the performance of several models did not differ greatly, even though some were considerably more sophisticated than others. The chosen model was derived by REML. It was possible to derive a simpler model (by multiple regression), which included N and rainfall but not clay, and which performed slightly better than the chosen model (as judged by the correlation of predicted and observed values), but in working with the daily data (Section 6) it was concluded that soil clay content was influential. It may be that REML is identifying a clay effect, whilst regression is not, because REML is able to take out of the analysis random effects due to 'year', 'experiment' and 'block'. Conversely, it was possible with REML to derive a slightly better performing model than the chosen model, which included crop type and crop type x N rate, but this was rejected because the improvement was slight, and again, in working with the data, no justification was apparent from the detailed analysis of the experiments in Section 6 for believing that (as predicted by the REML model) emission from winter barley was more than from winter wheat, or even that cereals and sugar beet differ significantly. These effects probably arose due to the unbalanced inclusion of crop type and site in the experiments. One advantage of the chosen model over the regression model is that it appears to avoid some under-prediction of high English N<sub>2</sub>O-N emission values.

Plots of predicted versus measured values (Figure 44 and Figure 45) show clearly how both the measured and modelled emissions from the wetter Scottish experiments were larger than those from the drier sites in the arable areas of England; there is also a tendency for the predictions for the Scottish site to exhibit a larger scatter than those in England.



**Figure 44. Comparison of measured N<sub>2</sub>O emissions (g N<sub>2</sub>O-N ha<sup>-1</sup> year<sup>-1</sup>, log transformed after adding 710) with emissions predicted by the chosen MIN-NO model.**



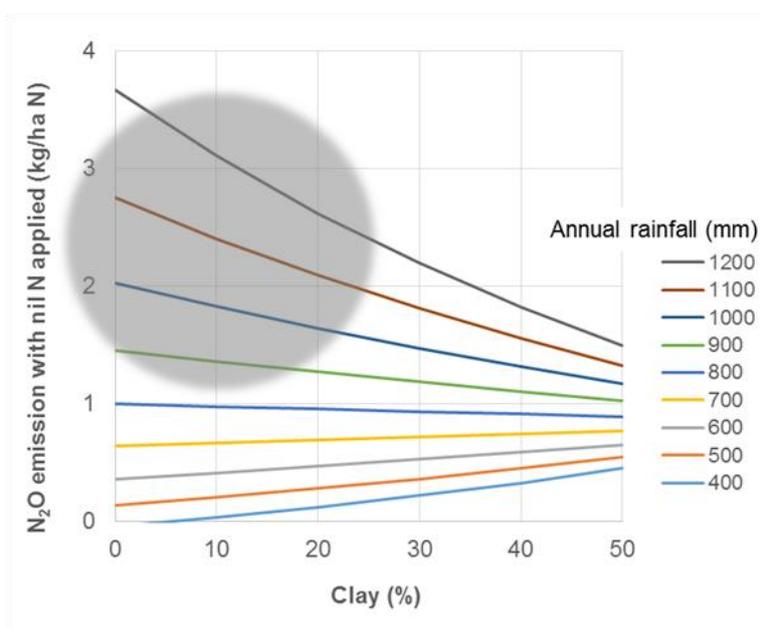
**Figure 45. Comparison of measured N<sub>2</sub>O emissions (g N<sub>2</sub>O-N ha<sup>-1</sup> year<sup>-1</sup>) with emissions predicted by the chosen MIN-NO model.**

The adopted model, which shall subsequently be termed the ‘MIN-NO model’, predicts that,

- At low clay contents emissions, with no fertiliser N applied there is a wide range of emissions from nil to >2 kg ha<sup>-1</sup> year<sup>-1</sup> N<sub>2</sub>O-N, depending on annual rainfall, but at higher clay contents, the range is at least halved (Figure 47). The origin and relevance of these emissions need to be addressed in subsequent sections, and discussed in relation to the findings in the crop residue experiments (Section 7) that there were little or no N<sub>2</sub>O

emissions associated with incorporation of crop residue N in the first year after incorporation. It is possible that this intercept partly arises from the mineralisation of crop residue (and other) N that has accumulated in the soil over the longer term.

- Emissions are approximately proportional to total amounts of N applied, however the proportion increases slightly as quantities of N increase, and it increases markedly as quantities of annual rainfall increase (Figure 47).
- Nevertheless, the interaction between clay content and annual rainfall would suggest that at high rainfall, a high clay content soil will have a lower EF for fertiliser N than a soil with less clay (Figure 48), and this is reversed at low annual rainfalls.



**Figure 46. Effects of rainfall and soil clay content on cumulative annual N<sub>2</sub>O-N emissions for soils with nil fertiliser N applied, as predicted by the MIN-NO model. The grey area indicates the main combinations of rainfall and clay content that were not tested in the MIN-NO experiments. Based on long-term average rainfall, these would comprise <5% of UK arable land.**

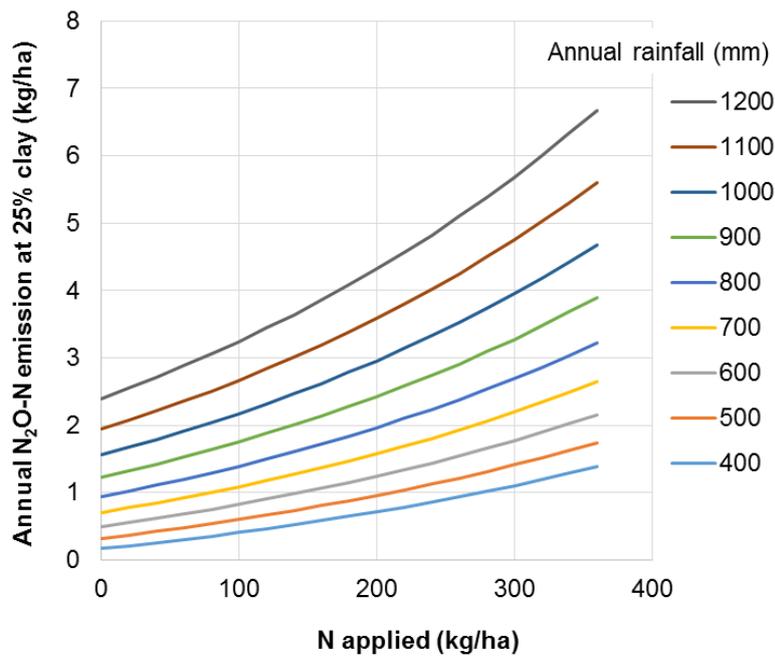


Figure 47. Effects of rainfall and applied N on cumulative annual  $N_2O-N$  emissions as predicted by the MIN-NO model for soils with 25% clay.

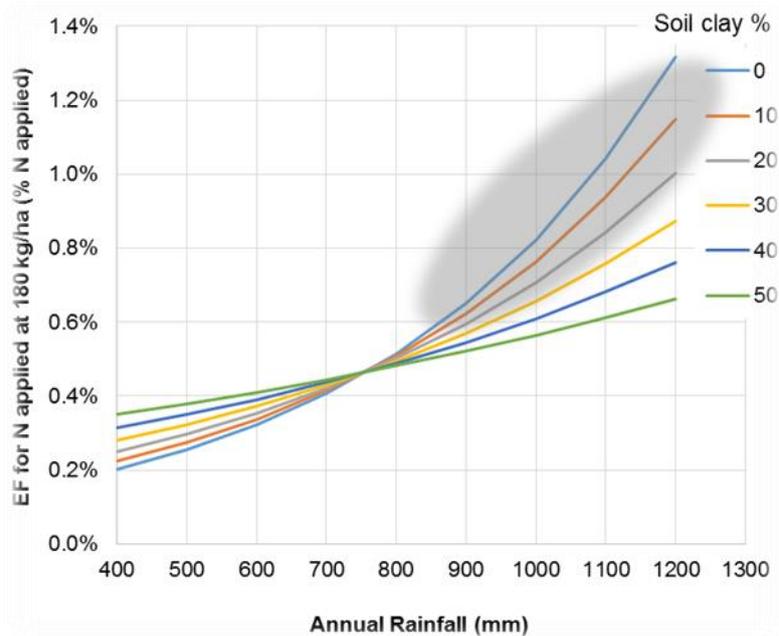


Figure 48. Effects of annual rainfall and clay content on the EF for cumulative annual  $N_2O-N$  emissions with 180 kg ha<sup>-1</sup> N applied, as predicted by the MIN-NO model. The grey area indicates the main combinations of rainfall and clay content that were not tested in the MIN-NO experiments. Based on long-term average rainfall, these would comprise <5% of UK arable land.

### 8.3. Discussion

The use of simulation models in biological research is common-place. However, their use in commercial decision-making or for policy support needs careful scrutiny. There are examples of such applications in reporting of GHG emissions in national inventory reports in the US and also in

irrigation scheduling world-wide but, by and large, decision-making in arable agriculture (variety choice, crop establishment, crop nutrition, crop protection, etc.) is based on simple empirical experimentation and statistically derived or experience-based 'models' directly linked with continuing experimentation, so adoption of the MIN-NO model would be more compatible with this approach than adoption of DNDC.

Simulation models, by their very nature, tend to be too complex to be validated comprehensively (Sylvester-Bradley *et al.*, 2005). Further, they tend to be developed by single authors who commonly invest the majority of their scientific endeavour in one model, hence the fortunes of that model become inextricably linked with the scientific fortunes of the modeller. This may have some advantages in terms of model development, but it serves to inhibit effective and objective evaluation of the model. It is evident that many different models of ecosystem processes have been developed and these exist in a highly competitive academic environment in which alternative approaches can be rigorously tested and compared (Abdalla *et al.* 2010; Li *et al.* 2005; Sansoulet *et al.* 2014). However, it can be difficult to validate these ecosystem models comprehensively; it may be possible to evaluate sets of predictions produced by such models by comparison with experimental data (Butterbach-Bahl *et al.* 2001; Rafique *et al.* 2011), but this does not necessarily guarantee their satisfactory performance in a different environment or for a different purpose. In the case of DNDC, the version that was available to the MIN-NO consortium proved to be unfit for or unsuited to practical use in the UK. When this conclusion was reached (in 2013), in conjunction with collaborators in parallel UK projects, it was decided best to revert to the more conventional approach of empirical interpolation between observed values, using statistical fitting, so to enable prediction of N<sub>2</sub>O emissions in support of carbon accounting for UK arable agriculture as a whole, and for arable-based supply chains.

However, modelling experience within this Project may be used to draw some lessons concerning appropriate choice of models for similar applied research projects into the future. At the outset it was assumed that the widespread use of DNDC around the world and the numerous scientific publications relating to it did provide assurance of its potential suitability. This assumption was clearly too optimistic at that time (noting that improvements to DNDC continue). It is unfortunate that the refereeing process employed by journals that publish descriptions and 'validations' of simulation models has not come to include checks that would diagnose and highlight 'bugs', misrepresentations and uncertainties (as they do with experimental results). There would seem to be an opportunity here for an element of the scientific press to institute such checks, so as to more clearly provide assurance of model credentials, and to minimise the chances of undue investment in inappropriate models by third-parties in future. Meanwhile, it will be well for policy makers and practitioners to exercise great care and caution before employing simulation models for practical purposes, especially where these have considerable economic or social significance.

With regard to the MIN-NO model, it is important to register the constraints that arise from its largely empirical origin, and its derivation from annual summary data. The cross-over interaction between rainfall and clay content (Figure 48) may merely arise because, at lower rainfall levels, light soils emitted less than heavier soils, whilst with modest rainfall (700 to 900mm) there was no clear soil effect. Note that the experiments did not test low clay content soils with high annual rain so (in Figure 48) this combination is all extrapolated beyond the available data (i.e. less than 30% clay with more than 900mm annual rain). Also it should be recognised that there was very little overlap in annual rainfall between England and Scotland; the driest year in Scotland (2010: 810 mm) was only just drier than the wettest site-year in England (2012: 890 mm), so the data include some confounding of rainfall with region (hence with soil characteristics).

Despite these reservations, results of the statistical modelling may have some value beyond mere practical application; for understanding of soil processes they provide evidence of the importance of interactions between controlling variables in influencing N<sub>2</sub>O emissions. Although fertiliser N input is known to play an important role in controlling emissions, the statistical results demonstrate very clearly that, when N fertilisers are applied in dry soil conditions, emissions are much less than those occurring in wetter soils. This interaction is effectively captured in the model developed here, and should enable regional predictions of current emissions (Section 9), and estimation of the potential for mitigation through different management interventions in different climatic regions of the UK (Section 10). However, in using the MIN-NO model to interpolate between arable cropping conditions within the UK, it will be important to recognise that, whilst the model is derived from a very wide range of annual rainfall (<400 to >1700 mm), widely differing soil types (3% to 49% clay) and extremes of fertiliser N application (nil to 360 kg ha<sup>-1</sup>), some combinations of these factors were not tested, and further validation and explanation are required. (Note that further validation will take place within the Defra GHG Platform Programme, after completion of the MIN-NO project.)

In pondering possible explanations of the cross-over effect in Figure 48, it is a tenable hypothesis that the combination of high rainfall and high clay content would create very low oxygen status soils (WFPS >70%) which the literature indicates might encourage denitrification to reduce NO<sub>3</sub> (and NO<sub>2</sub> and N<sub>2</sub>O) through to N<sub>2</sub>, rather than just to N<sub>2</sub>O. Conversely heavy rain and low clay soils might still cause WFPS to be moderate, so might cause more N<sub>2</sub>O emission than a higher clay soil. Noting that the MIN-NO model predicts light soils with high rainfall amounts (Figure 48) to give the greatest N<sub>2</sub>O emissions (Figure 46 and Figure 47), further work would seem necessary on light soils, maybe combined with tests of whether there is a disproportionate effect of rainfall on N<sub>2</sub> generation on heavy soils. Nevertheless, it is important to note that the model still predicts high total N<sub>2</sub>O emissions with high rainfall whatever the soil type.

## 9. Upscaling and UK inventory reporting – WP4

### 9.1. Introduction

The objectives of this work package were to deduce geographically extrapolated EFs (EF) for direct N<sub>2</sub>O emissions from N fertiliser applications used to grow arable crops, as studied in this project, and to assess the responsiveness of these to features of environment and management. The EFs derived from this study were then to be compared with the current default EF used in the UK inventory of GHG emissions from agriculture, and an assessment was to be made of the impact of any changes on the total agricultural emission estimate for the UK. The derived EFs would also be compared with those derived from previous mechanistic modelling (UK\_DNDC) conducted as part of Defra project AC0101.

### 9.2. Methods

#### 9.2.1. Model description

As part of Work Package 3, an empirical model (the MIN-NO model; Section 8) was developed based on the results of the field experiments conducted under Work Package 2a (Eq. 1).

$$\ln(N_2O + 710) = \text{Constant} + \text{NRate} + \text{SumAnnRain} + \text{Clay} + \text{ClayAnnRain} \quad \text{Equation 9}$$

whereby the natural logarithm of the cumulative annual emission of N<sub>2</sub>O-N (g ha<sup>-1</sup>; plus 710 to avoid negative values) is predicted from *NRate* (the total amount of fertiliser N applied; kg ha<sup>-1</sup>), *SumAnnRain* (the cumulative annual rainfall for the year in which the fertiliser N was applied; mm), *Clay* (the clay content of the soil; %) and *ClayAnnRain* (the interaction between clay and rainfall expressed as rain.clay/100). The values for the corresponding model coefficients were:

Constant:	5.567	(s.e. 0.3490)
NRate:	0.002402	(s.e. 0.0001062)
SumAnnRain:	0.002349	(s.e. 0.0005046)
Clay:	0.023440	(s.e. 0.009793)
ClayAnnRain:	-0.003096	(s.e. 0.0014161)

#### 9.2.2. Model runs

The model (Eq. 1) was used to estimate emissions at two spatial scales: county and 5 x 5 km grid. The county scale was as described in Cardenas *et al.* (2013) whereby 72 counties were represented for the UK (with 46 counties located in England, 8 in Wales, 6 in Northern Ireland and 12 in Scotland). County boundaries were the same as those in Cardenas *et al.* (2013) corresponding to the year 2000. Northern Ireland was excluded from the 5 km grid scale runs because not all required data were available with this distribution.

Crop areas at the county scale for the year 2010 were obtained from Defra and Devolved Administration statistics (see Appendix, Table 61). The 2010 crop areas for the 5 km grid runs for Great Britain were obtained from EDINA agcensus (<http://edina.ac.uk/agcensus/>). Some crop category survey elements differ between the Devolved Administrations, and the level of detail (i.e. number of different crop categories) differed between national statistics and the EDINA data reported at the 5 km grid scale. For England and Scotland, the area reported as winter OSR was used for OSR. For Wales, the area of OSR in each 5 km grid square (not reported explicitly) was estimated from the ratio of the areas of OSR and 'other crops' at the national level. Similarly, the area of winter and spring barley for Wales at the 5 km grid scale (reported together as barley) was estimated from the ratio of total areas at the national level.

At the county scale, the clay content data for each of the three dominant soil types for each county were used, classified by a combination of hydrology of soil types (HOST) class and texture. Dominant soils were selected on the basis of their area coverage of agricultural land in each county. Data were supplied from the databases of Soil Survey and Land Research Centre (SSLRC) for England and Wales, Macaulay Land Use Research Institute (now James Hutton Institute) for Scotland and the Department of Agriculture and Rural Development, for Northern Ireland. On average the three dominant soil types accounted for 60% of the total agricultural area of each county (excluding common rough grazing areas). At the 5 km grid scale, clay content data were provided by the National Soil Resources Institute (Cranfield). Data for Scotland were only available at the 10 km grid scale (from the National Soils Inventory of Scotland – point dataset collected by the James Hutton Institute); therefore the clay content of each 10 km grid was applied to the 5 km grids covering the same area. Note that the soils data were supplied under licence, and are not reproduced in this report.

Fertiliser application rate data for each of the 5 crops were derived from the British Survey of Fertiliser Practice (BSFP) 2010 survey data (Tables EW1.1 and EW1.5 for England and Wales and SC1.1 and SC1.5 for Scotland for overall application rate and the distribution of application rates, respectively; publication available from [www.defra.gov.uk](http://www.defra.gov.uk)). For OSR, application rates corresponding to winter OSR were used; for sugar beet there were no fertiliser application rate data for Scotland, so values from the England and Wales tables were also used for Scotland. Data used in the model runs are given in the Appendix (Table 60).

Total cumulative annual rainfall data for each 5 km grid were derived from data provided by the Meteorological Office for the thirty years 1981 to 2010. For county scale model runs, the rainfall data were weighted towards the arable cropped areas within each county.

Model runs were carried out to derive the direct N<sub>2</sub>O EFs for fertiliser N application for the crops included in the MIN-NO experiments (Sections 6 and 8) and, particularly, to assess the importance of representing the fertiliser application rate distribution, the inter-annual variability in total annual rainfall and the spatial scale. Model runs were conducted at both county and 5 km grid scales. At the county scale, model output represented the weighted average of the outputs corresponding to each of the three soil types used for each county. Each model run (at either scale) produced output based on the weighted fertiliser application rate distribution and based on the overall fertiliser rate for each crop type. A model run was conducted for each of the thirty annual rainfall values (1981-2010), from which an average output was derived. A further model run using the 30-year mean cumulative annual rainfall was also conducted.

To derive EFs from each model run, runs were conducted using a zero N application rate in addition to the BSFP overall and distributed application rates. Total N<sub>2</sub>O-N emissions from the 'zero' run were then deducted to produce 'net' N<sub>2</sub>O-N emission values for each run at either the county or 5 km grid scale. The corresponding EF were calculated as follows:

$$EF_{ij} = (N2O_{fij} - N2O_{zero_{ij}}) * 100 / N_{input_{ij}} \quad \text{Equation 10}$$

where  $EF_{ij}$  is the EF for county or 5 km grid  $i$  and crop  $j$ ;  $N2O_{fij}$  is the total emission from N fertiliser application for county or 5 km grid  $i$  and crop  $j$ ;  $N2O_{zero_{ij}}$  is the background emission (no N added) for county or 5 km grid  $i$  and crop  $j$ ; and  $N_{input_{ij}}$  is the total fertiliser N applied in county or 5 km grid  $i$  to crop  $j$ .

### 9.3. Results and discussion

The mean EF derived from the county and 5 km grid scale model runs for each crop type and across all 5 crop types are given in Table 40 and Table 41, respectively. It is important to note that crop type *per se* is not an explicit factor in the MIN-NO model, so the different EFs for crop types shown in Table 40 and Table 41 are as a result of different fertiliser application rates and, perhaps more importantly, different spatial distributions of the crops (Figure 50).

There was very little difference between the EF derived from the county scale model runs and those from the 5 km grid scale. This implies that the spatial variation in the main driving factors (rainfall and clay content) is unimportant at the small scale; however it becomes more apparent at the regional scale. The weighting of the county rainfall data to the areas within the county where crops are grown was important in this respect as it removed any bias that might have been introduced at the county scale due to high rainfall in upland, non-cropped areas.

**Table 40. Emission factor estimates from county scale runs**

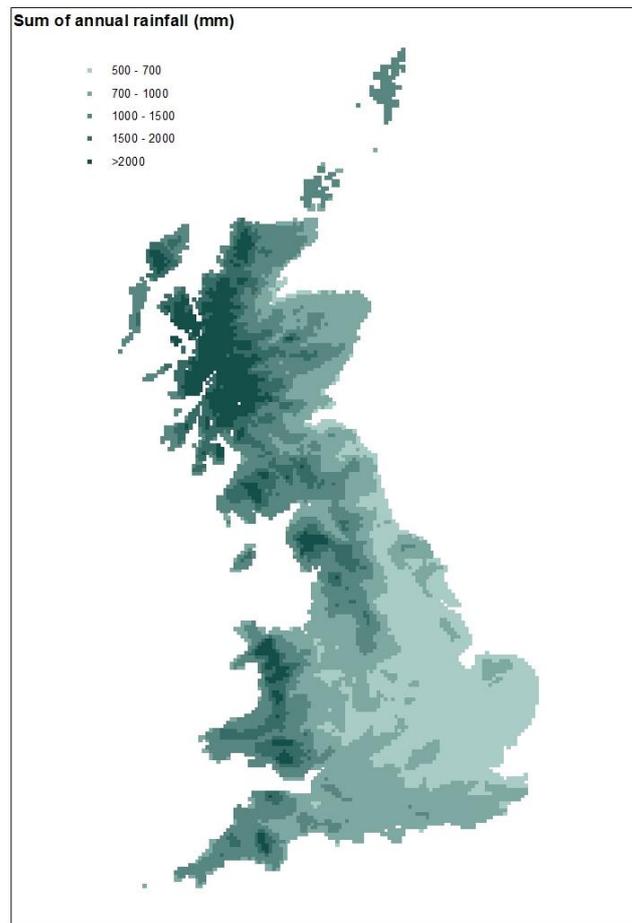
Crop	Fertiliser rate basis	Emission Factor (% of applied N)		
		Mean from 30 runs using year-specific rainfall	Standard deviation	Single run using 30-year mean annual rainfall
Winter Wheat	Overall	0.45	0.07	0.45
Winter Wheat	Weighted distribution	0.46	0.06	0.46
Winter Barley	Overall	0.47	0.07	0.46
Winter Barley	Weighted distribution	0.48	0.07	0.47
Spring Barley	Overall	0.53	0.09	0.51
Spring Barley	Weighted distribution	0.54	0.09	0.52
OSR	Overall	0.45	0.06	0.44
OSR	Weighted distribution	0.46	0.06	0.45
Sugar Beet	Overall	0.35	0.05	0.34
Sugar Beet	Weighted distribution	0.36	0.05	0.35
ALL	Overall	0.46	0.07	0.45
ALL	Weighted distribution	0.47	0.07	0.46

**Table 41. Emission factor estimates from 5 km grid scale runs**

Crop	Fertiliser rate basis	Emission Factor (% of applied N)		
		Mean from 30 runs using year-specific rainfall	Standard deviation	Single run using 30-year mean annual rainfall
Winter Wheat	Overall	0.45	0.06	0.44
Winter Wheat	Weighted distribution	0.46	0.06	0.45
Winter Barley	Overall	0.47	0.07	0.46
Winter Barley	Weighted distribution	0.48	0.07	0.47
Spring Barley	Overall	0.54	0.10	0.52
Spring Barley	Weighted distribution	0.55	0.10	0.53
OSR	Overall	0.45	0.06	0.44
OSR	Weighted distribution	0.46	0.06	0.45
Sugar Beet	Overall	0.35	0.05	0.34
Sugar Beet	Weighted distribution	0.35	0.05	0.35
ALL	Overall	0.46	0.06	0.45
ALL	Weighted distribution	0.47	0.07	0.46

Accounting for the distribution of fertiliser application rates for each crop type made very little difference in comparison to using the overall crop rate (the EF was marginally greater from the calculation with distributed rates). This indicates that the non-linearity in the function relating emissions to fertiliser rate is relatively small over the rates typically applied to these crops. Experimental results from Defra study AC0101 showed highly non-linear response of N<sub>2</sub>O emission to application rates for fertiliser applications to grazed grassland (reported by Cardenas *et al.*, 2010), but predominantly linear response for emissions from arable sites. Accounting for inter-annual variability in total annual rainfall also made very little difference to the EF estimates. In

future it would therefore seem quite reasonable and much more straight-forward to derive EFs based on overall fertiliser application rates and 30-year average weather data (Figure 49).

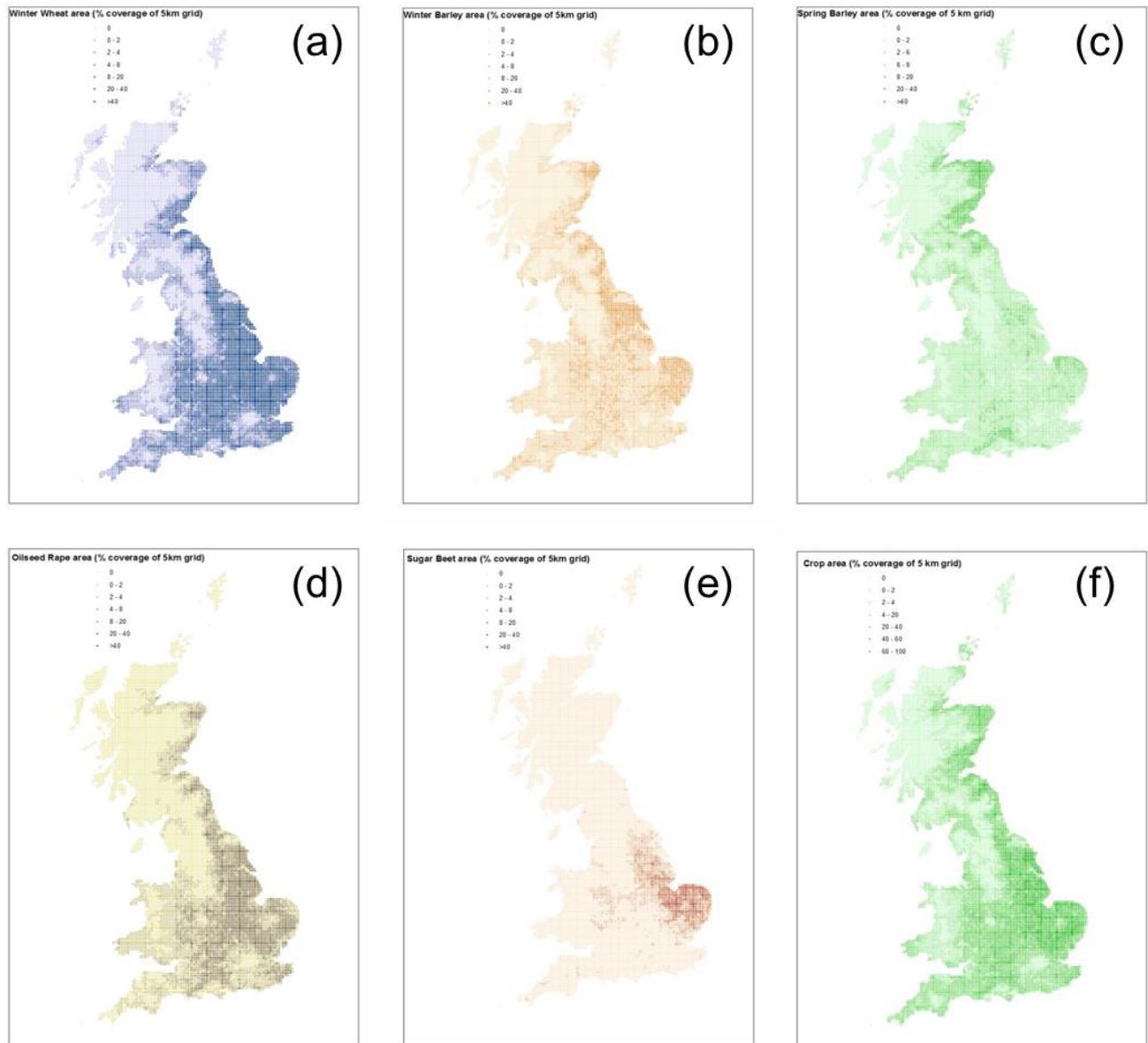


**Figure 49. Distribution of cumulative annual rainfall (30-year mean, 1980-2010) across Great Britain; categories range from 500-700 mm (light), through 700-1000 mm, 1000-1500 mm, and 1500-2000 mm, to >2000 mm (dark).**

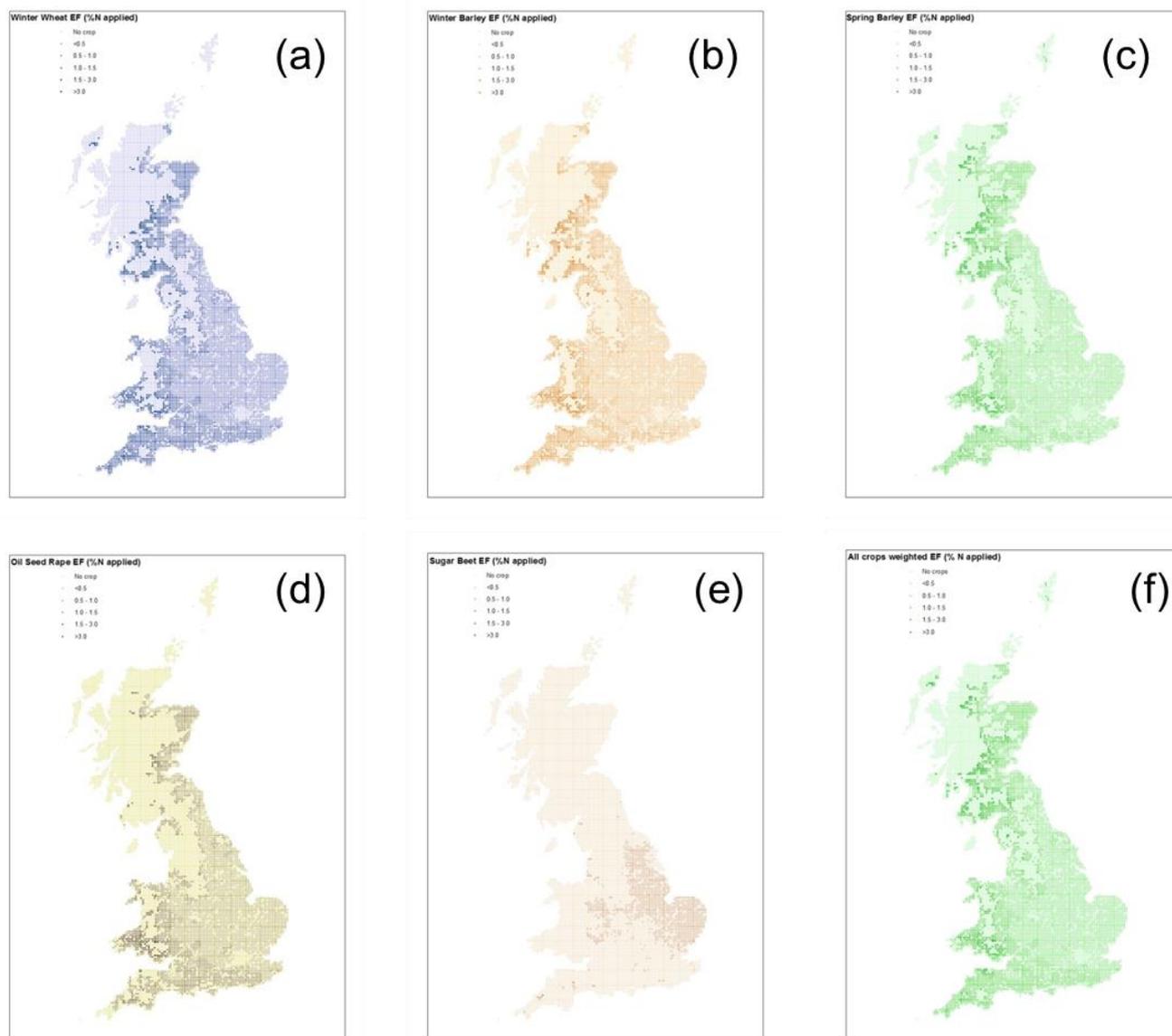
Comparing mapped crop areas with estimated EFs (at the 5 km grid scale, using 30-year average rainfall and overall fertiliser application rates for each crop type) it is evident that the arable crops are predominantly grown in the areas associated with lower EFs (Figure 50 and Figure 51). Major drivers of the higher EFs in the west of the country are the spatial distribution of annual rainfall (Figure 49) and the clay content of the soils, rather than the inter-annual variation in rainfall for a given location. Sugar beet, which has the most localised distribution of the crop types studied with a very eastern-biased distribution, is associated with the lowest EF (Table 40 and Table 41). In contrast, spring barley which has a more marginal distribution is associated with the greatest EF.

This study focussed on the major crop types grown in the UK (winter wheat, winter and spring barley, oil seed rape and sugar beet), which account for 82% of total UK arable crop area and 92% of fertiliser N applied to arable crops (2012 values). The most recently submitted UK inventory of

GHG emissions (1990-2012), which was compiled according to IPCC 1996/2000 Guidelines, estimated the total emission of N<sub>2</sub>O-N from UK agriculture to be 61.1 kt. Of this, direct N<sub>2</sub>O emissions from total fertiliser use accounted for 12.6 kt N<sub>2</sub>O-N (21%), with direct emissions from fertiliser applied to arable crops estimated at 7.7 kt N<sub>2</sub>O-N (13%).



**Figure 50. Coverage of a 5 km grid by the major arable crops across Great Britain: (a) winter wheat, (b) winter barley, (c) spring barley, (d) OSR, (e) sugar beet, and (f) all of these crops. Shades scale from nil (unshaded), through 0-2%, 2-4%, 4-8%, 8-20% & 20-40% (& 40-60% for all crops) to >40% (or >60% for all crops; darkest shading) of total land area.**



**Figure 51. Estimated N<sub>2</sub>O-N EFs (% of N applied) for fertiliser N applications to the major arable crops in Great Britain: (a) winter wheat, (b) winter barley, (c) spring barley, (d) OSR, (e) sugar beet, and (f) all these five crops. Shades scale from no crop (unshaded), through <0.5%, 0.5-1.0%, 1.0-1.5% & 1.5-3.0% to >3.0% of N applied (darkest shading).**

The default IPCC 1996/2000 Guidelines EF for direct emissions from N fertiliser applications is 1.25% of the applied N after accounting for losses via volatilisation. The UK, in common with most other countries, will be adopting the IPCC 2006 Guidelines for inventory reporting for the next submission (1990-2013), in which the EF for direct emissions from N fertiliser applications has been revised to 1% of applied N, with no correction for losses via volatilisation. Estimates using the IPCC 2006 Guidelines for total N<sub>2</sub>O-N emission from UK agriculture (2012), for direct emission from all fertiliser N and for direct emission from fertiliser N applied to arable crops are 40.1, 11.2 and 6.9 kt, N<sub>2</sub>O-N respectively. The mean EF found here for direct N<sub>2</sub>O-N emissions from fertiliser N applied to UK arable crops is 0.46% (not accounting for any volatilisation losses), i.e. less than half the IPCC 2006 Guidelines default of 1%. Implementing this EF gives an estimate of direct

N<sub>2</sub>O-N emissions from fertiliser N applied to arable land (2012 values) of 3.2 kt i.e. a decrease in the emission estimate of 3.7 kt N<sub>2</sub>O-N, which is almost a 10% reduction in the estimate of total N<sub>2</sub>O-N from UK agriculture.

However, some caution must be exercised in applying or extrapolating the EF derived from this study. The model derived from the MIN-NO experimental dataset requires validation against other UK experimental datasets of N<sub>2</sub>O emissions from fertilised arable crops. In particular, evidence is required for combinations of rainfall and clay content not represented in the MIN-NO experimental dataset (e.g. high rainfall with low clay content, where the model currently predicts high emissions; Figure 48). This study has demonstrated how a relatively simple model can be combined with available spatially disaggregated national datasets to produce an improved national estimate of emission with a reasonable level of confidence. Clearly such a model can better represent country-specific environmental and management factors than the current IPCC Tier 1 approach.

## 10. Integration and economic analysis of N<sub>2</sub>O mitigations – WP5

### 10.1. Introduction

The aims of Work Package 5, described in this section are to

- i. propose revised approaches to estimating N<sub>2</sub>O emissions, by means of ‘smart’ EFs based on findings from WP1 through to WP4 (Section 10.2), and
- ii. assess their effects on the UK national GHG inventory and on estimated GHG intensities of crops and their products (Section 10.3).

and then, using the smart EFs, it aims to

- iii. devise feasible strategies for mitigating N<sub>2</sub>O emissions associated with arable agriculture (Section 10.3.3), and
- iv. prioritise these mitigations, based on their estimated impacts on the GHG intensities of crops and their products, and their estimated economic effects (Section 10.5).

At the outset it is crucial to distinguish the two main contexts in which agricultural GHG emissions are assessed in the UK, and to acknowledge the different criteria, practicalities and uncertainties pertaining to each: (i) national inventory reporting and (ii) LCA (or ‘carbon footprinting’) of farm, food and fuel products. It is even the case that, within the UK, different communities have been studying and administering GHG emissions in these two contexts and, since this project has engaged both communities, this section reflects both common and some distinct views of the project’s findings.

The first main context for assessment of UK GHG emissions is at a national scale and concerns preparation of the UK national GHG inventory. The inventory is compiled according to IPCC guidelines to represent average annual GHG emissions from anthropogenic sources, including agricultural activity, within UK territories; it is primarily used to meet international obligations to report on national GHG emissions at decadal intervals. However, the inventory is also sometimes used to inform national climate change mitigation policy for agriculture, as a specific sector of the economy.

IPCC guidelines for the preparation of national GHG inventories allow a choice between three different methods for estimating soil N<sub>2</sub>O emissions, referred to as the Tier 1, 2 and 3 methods (Eggleston *et al*, 2006). The Tiers offer increasing degrees of accuracy through increasing levels of sophistication, but they therefore require increasing levels of investment for the countries that adopt them. The Tier 1 method involves applying relatively few, internationally agreed default EFs that relate direct N<sub>2</sub>O emissions from soil and indirect N<sub>2</sub>O emissions from volatilisation and atmospheric deposition, and from leached nitrate to aggregated national quantities of fertiliser N use, and estimated returns of organic manure N and crop residue N to land. The Tier 1 EFs are

intended to be generalised approximations which do not take into account, explicitly, the effects of particular factors which might cause soil N<sub>2</sub>O emissions to vary locally e.g. crop type, soil type, soil management and climate. The IPCC guidelines provide default values for Tier 1 EFs which have been adopted for determination of national GHG inventories by most countries; Tier 1 EFs do not have to be validated locally. With the Tier 2 method, more detailed country-specific EFs are necessary; these can be locally determined and can relate to influential factors which, with suitable spatial activity data, can be used to attempt more accurate national GHG inventories. The Tier 3 method uses simulation modelling to evaluate soil N<sub>2</sub>O emissions, rather than using EFs. Adoption of the Tier 3 method depends on a model being validated against local experimental measurements. Alternatively, it is possible to use modelling as well as experimental measurements to derive national alternatives to the international default EF values for Tier 1, or indeed to derive EF values for Tier 2.

The second main context for assessment of agricultural GHG emissions is in support of LCA, i.e. carbon footprinting; usually this is at the finer scale of a product, a farm or an enterprise, and usually this includes consideration of GHG emissions associated with manufacture of the fertiliser used. (In the national GHG inventory context, emissions from fertiliser manufacturing are considered within the industrial sector, which is separate from the agricultural sector.) Estimation of soil N<sub>2</sub>O emissions at this finer scale is somewhat more complicated and confused than at the national inventory scale. This is mainly because there is no single methodology for GHG emissions calculation. However, it is also partly because the purposes of the many different calculation methodologies and the associated tools that are available are not adequately specified or even fully articulated, and because many studies either do not state or do not justify the approaches which they adopt. Currently, there is only one mandatory GHG emission calculation methodology in force in the UK, and this is the RED methodology for reporting GHG intensities of biofuels and bioliquids for regulatory purposes (European Commission, 2009). However, in terms of estimating soil N<sub>2</sub>O emissions associated with the cultivation of crops for biofuels and bioliquids, the RED methodology states that any one of the IPCC Tier 1, 2 or 3 methods can be used (European Commission, 2010). As a result, current RED approaches used within the EU are apparently based on a mixture of IPCC Tier 1 EFs and IPCC Tier 3 modelling, whilst proposed new RED default values for EU crops (Edwards *et al.*, 2013) are derived from the crop- and site-specific Global Nitrous Oxide emission Calculator (GNOC; Koeble, 2014). Thus even mandatory schemes can be equivocal.

In most LCA, calculation methodologies, tools, and studies of GHG emissions adopt IPCC Tier 1 EFs, probably because they require no local or specific justification. This is appropriate if the purpose of the LCA is to determine generalised GHG intensities that are comparable across crops, across products and across countries. However, GHG intensities are being used increasingly as a

means of driving reductions in GHG emissions, as part of climate change mitigation. In particular, this is the purpose behind application in the UK of the PAS2050 approach to carbon footprinting of products and services, officially certified by the Carbon Trust (CT). Within this certification scheme, companies are expected to achieve year-on-year reductions in the GHG intensities of their products and services. Hence, LCA under PAS2050 encourages companies to apply strategies which reduce GHG emissions through their supply chains. Companies with products reliant on crop cultivation thus become interested in mitigation options which affect major sources of GHG emissions, including fertiliser use and soil N<sub>2</sub>O emissions. In addition to encouraging mitigation strategies on farms, this approach raises possibilities (rightly or wrongly; see Section 10.4.5) of reducing soil N<sub>2</sub>O emissions by selectively sourcing specific crops from locations with more favourable soil types and climates. Hence, although many calculation tools, including the CT's own carbon footprinting software<sup>3</sup>, incorporate IPCC Tier 1 EFs, pressure should be expected to adopt IPCC Tier 2 EFs which can reflect national evidence on GHG emissions and can recognise factors such as soil type and climate. However, any significant expansion or sophistication of carbon footprinting in the UK, for instance through adopting IPCC Tier 2 EFs at farm- or enterprise-scale, will depend on policies and financial incentives to support crops and products with lower GHG intensities.

In summary, it should be recognised throughout this section that the two most important contexts for GHG accounting in the UK have quite distinct scales and aims, but they are also connected: LCA depends strongly on development of the national GHG inventory, because it commonly adopts the same procedures and EFs. This dependence is almost inevitable because LCA practitioners and their clients have fewer resources than the governments responsible for GHG inventories, and yet they generally seek the same degrees of veracity and validity for the procedures they use. Thus it will be necessary in the ensuing analysis and discussion of GHG accounting procedures and EFs to recognise that GHG inventories and LCA are both distinct and interdependent.

## **10.2. Deriving 'smart' emission factors**

The results of this project, as well as new findings published in the literature over the period of the MIN-NO project, enable us to propose country-specific replacement values for IPCC Tier 1 default EFs for soil N<sub>2</sub>O emissions, along with IPCC Tier 2 EFs for evaluating national GHG inventory and farm-scale GHG intensities for crops and their products. For convenience, these will be referred to here as 'smart EFs'. For completeness in relation to GHG emissions associated with the UK agricultural sector and GHG intensities for crops and their products, the smart EFs encompass GHG emissions from fertiliser manufacture in addition to direct and indirect soil N<sub>2</sub>O emissions. It

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<sup>3</sup> <http://www.carbontrust.com/client-services/footprinting/footprint-measurement/carbon-footprint-software/>

should be noted that these smart EFs are proposed in the knowledge that there will be subsequent wider consideration before a new set of EFs are adopted for the UK, for example, in the reporting of Defra's GHG Projects AC0114 and AC0116 in 2015-6. Thus the set of smart EFs developed here are not intended for direct adoption, either for national inventory reporting or for LCA; they are merely derived as the best summary of UK-related evidence on arable N<sub>2</sub>O emissions, and to explore the scope for and impacts of mitigations.

### **10.2.1. Fertiliser manufacturing emissions**

Fertiliser manufacturers in the UK and across Europe have made substantial investments in recent years to abate N<sub>2</sub>O emissions from AN production plants. At the same time, work co-ordinated by the European fertiliser industry, represented by the European Fertiliser Manufacturers Association (EFMA), now 'Fertilizers Europe', and the AIC ran concurrently with the MIN-NO project to update estimates of GHG intensities of N fertiliser products manufactured in European facilities (Fertilizers Europe, 2014). This work describes the situation in 2011 and updates work which described the situation in 2006 (Brentrup and Pallière, 2008). Table 2 shows how the manufacturing EF for AN has reduced from 6.31 to 3.52 kg CO<sub>2</sub>e kg<sup>-1</sup> N, whilst the correction of previous inaccuracies in the urea estimate have increased its manufacturing EF from 3.18 to 3.57 kg CO<sub>2</sub>e kg<sup>-1</sup> N.

### **10.2.2. Direct N<sub>2</sub>O emissions due to fertiliser use**

The UK national GHG inventory is moving towards adoption of a default value for the 2006 IPCC Tier 1 EF for direct N<sub>2</sub>O emissions from N fertiliser application, which assumes that 1% of applied N will be lost as N<sub>2</sub>O-N; this reduces from the 1.25% EF for applied N (minus 10% NH<sub>3</sub>-N loss) adopted in 1996 as the IPCC Tier 1 default value. [Note that the IPCC (2006) guidelines were adopted for the UK's inventory submission in 2015.] However, the large experimental programme in WP2 and the subsequent modelling and upscaling in WP3 and WP4 have provided a sufficient body of direct evidence of emissions for further UK-specific revision, along with a relatively robust means of interpolation between conditions for fertiliser applications to arable crops. Whilst there are caveats applied to use of this evidence, it indicates that the current best estimate of the IPCC Tier 1 EF for direct soil N<sub>2</sub>O-N emissions from use of fertiliser N in arable cropping at a national UK level is approximately 0.46%, just less than half the default value for the 2006 IPCC Tier 1 EF. Furthermore, the finding of variation with soil type and rainfall offers scope for adjustment according to location within the UK, through new smart EFs derived for feasible use at IPCC Tier 2 level.

IPCC Tier 2 smart EFs could be produced at various scales: 5 km square (as in WP4; Section 9), county or region (e.g. NUTS2). In practice the unit of accounting is likely to depend on the availability of appropriate activity data (i.e. N fertiliser use, rainfall and soil type). If data are not available at a fine scale, it will be necessary either to broaden the scale at which the accounting is

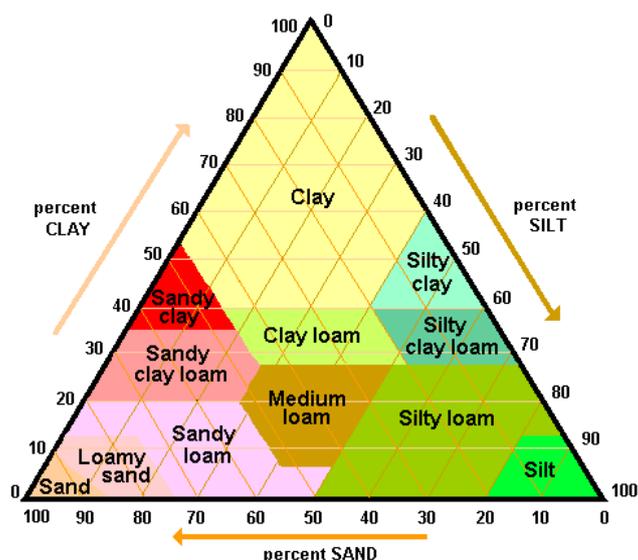
conducted, or to use default activity data that have been set to represent the *likely* N fertiliser application rate, rainfall and soil type, rainfall which would apply to an average crop with a given yield.

To explore variation due to location in direct fertiliser N-related N<sub>2</sub>O emissions across UK arable land, nine scenarios were chosen here with differing annual rainfall and soil clay contents, as shown in Table 42. It should be noted that, in interpreting the results from these scenarios, one in particular, scenario G, has been extrapolated beyond the conditions tested by the MIN-NO experimental programme. Therefore, the subsequent result for scenario G is subject to the greatest uncertainty. Based on distribution of soils, arable cropping and average annual rainfall over the last 30 years, this scenario would represent a small proportion (<2%) of UK arable crops.

**Table 42. Scenarios tested in WP5 using the MIN-NO model. Scenario G has least certainty.**

Annual Rainfall (mm)	Clay content %		
	5	25	45
500	A	B	C
700	D	E	F
1000	G	H	I

The relevance of these scenarios to arable farming regions within the UK can be judged from the annual rainfalls shown in Figure 49 and the texture diagram shown in Figure 52. Almost all arable land in the UK has annual average rainfall in the range 500 to 1000 mm, and has a soil clay content of up to 50%. (Of course, individual years have rainfall values outside this range, but any agricultural strategy must depend on LTAs and, as shown in Section 9, use of LTAs creates no bias.) High rainfall areas are in the west and north; least rain falls in the East and South, with the driest locations being around Essex and Cambridgeshire. Arable farming, particularly of more diverse spring sown crops, commonly takes place on loamy sands, sandy loams and silty loams which often have low soil clay contents. On the other hand, a significant area of arable land in Eastern UK has top-soils with a clay content >25%, and tends to specialise in autumn sown cropping.



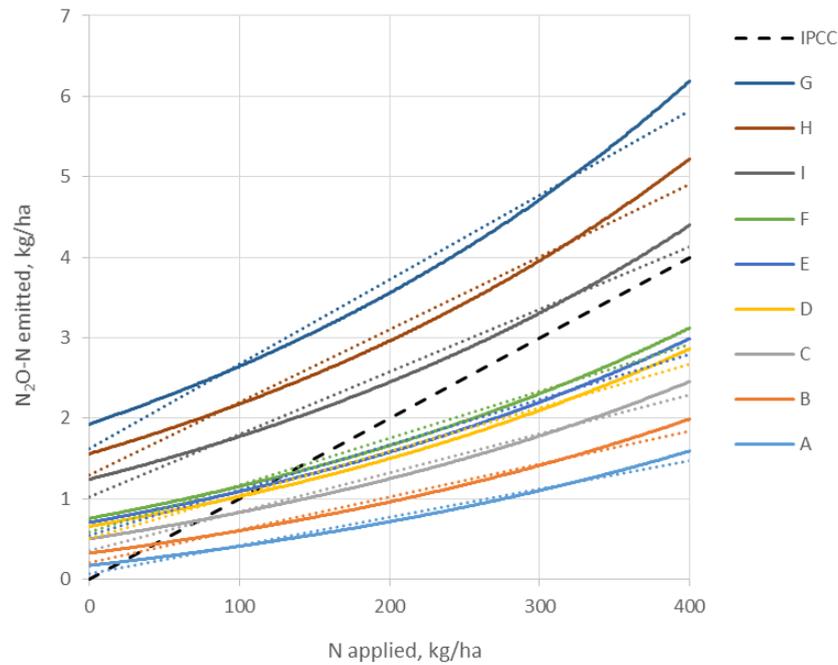
**Figure 52. The triangle used to define soil textures in the UK. Equal percentages of clay are represented by horizontal lines on the diagram. Textures with <50% clay include almost all textures found in top-soils.**

The direct N<sub>2</sub>O emissions estimated with the MIN-NO model for each of the above scenarios across the full range of N fertiliser application rates are shown in Table 43. Both the zero-N emission and the proportion of fertiliser N emitted as N<sub>2</sub>O are predicted to increase as rainfall increases. With low rainfall, clay increases emissions, whereas with high rainfall the effect of clay is uncertain, and possibly negative; only with (the most uncertain) Scenario G does the proportion of N emitted (with 400 kg ha<sup>-1</sup> N applied) exceed the 1% factor assumed in the default value for the 2006 IPCC Tier 1 EF. However, as indicated in Figure 53, the effect of the N fertiliser application rate on direct N<sub>2</sub>O emissions is non-linear.

### **Addressing non-linearity of direct N<sub>2</sub>O emissions**

As hypothesised at the outset of the MIN-NO project, about half of the 24 experiments in WP2A showed non-linear responses of direct N<sub>2</sub>O emissions to the N fertiliser application rate (see Table 25), emissions increasing disproportionately with increasing fertiliser N application rate. In that instance, the best models derived from the full dataset (in WP3; section 8.2) were found to require a non-linear function to account for the effect of the N fertiliser application rate, albeit with only slight curvature. Thus, relationships predicted by the MIN-NO model between N<sub>2</sub>O-N emitted directly and fertiliser N applied are *all* curved (Figure 53), with emissions increasing disproportionately at higher N rates. However, little information appears to be lost by describing the N effects with linear slopes (dotted lines in Figure 53). Given the greater ease with which linear relationships in GHG emissions accounting can be communicated and used compared to ‘curved’ models, it is suggested that average EFs can be ascribed to each of these scenarios, as is used by IPCC. In taking this approach, it must be acknowledged that the intercepts and/or slopes do depend, to some extent, on the range of fertiliser N application rates considered.

Figure 53 shows that, if N rates between 0 and 400 kg ha<sup>-1</sup> are averaged, then the emissions at ~200 kg ha<sup>-1</sup> N are over-estimated and emissions near zero-N are under-estimated as a consequence of approximating the derived curves by linear relationships. To provide an EF which is accurate at near-normal rates for most crops, it is suggested that the fitted slope at 200 kg ha<sup>-1</sup> is used, provided that the intercepts predicted by the MINN-NO model from Figure 53 are also adopted. The resultant over-estimation of total emissions at around 200 kg ha<sup>-1</sup> will be minor compared to locational variation.



**Figure 53. Response of direct N<sub>2</sub>O emissions to N fertiliser applied using the MIN-NO model, with scenarios for soil clay content and annual rainfall defined as in Table 42. Linear relationships across N rates 0 to 400 kg ha<sup>-1</sup> are also shown (dotted lines). The dashed black line shows the direct N<sub>2</sub>O EF for fertiliser N use, as adopted as the default value by 2006 IPCC, for comparison with the modelled slopes (the IPCC intercept is not comparable).**

**Table 43. Background emissions (with nil N applied) and EFs at two N rates derived for the 9 scenarios. Scenario G has least certainty.**

Scenario	Emission, or EF		
	at nil kg N ha <sup>-1</sup> <i>kg N<sub>2</sub>O-N ha<sup>-1</sup></i>	at 200 kg N ha <sup>-1</sup> <i>kg N<sub>2</sub>O-N kg<sup>-1</sup> N applied</i>	at 400 kg ha <sup>-1</sup> N
A	0.17	0.0027	0.0035
B	0.32	0.0032	0.0041
C	0.50	0.0037	0.0048
D	0.66	0.0042	0.0054
E	0.71	0.0044	0.0056
F	0.76	0.0045	0.0058
<b>G</b>	<b>1.93</b>	<b>0.0081</b>	<b>0.0105</b>
H	1.56	0.0070	0.0090
I	1.24	0.0060	0.0078

### 10.2.3. Indirect emissions

The MIN-NO project has not explicitly measured or reviewed indirect N<sub>2</sub>O emissions, but strong evidence from elsewhere (see below) suggests that 2006 IPCC Tier 1 default value probably over-estimates NH<sub>3</sub> emissions from UK arable agriculture. IPCC Tier 1 default values for indirect EFs assume that the fraction of applied fertiliser N lost as NH<sub>3</sub> through volatilisation is 10%. A detailed programme of experiments in the UK from 2003 to 2005 showed that NH<sub>3</sub> emissions from AN, the majority form of manufactured N fertiliser used in the UK, are typically around 3%, and urea treated with a urease inhibitor emits at similar levels (Chadwick *et al.*, 2005; Chambers *et al.*, 2009); only urea (untreated with an inhibitor) caused emissions of 10% or more (Chadwick *et al.*, 2005; Chambers *et al.*, 2009). In addition, given the smaller EF found here for direct soil N<sub>2</sub>O emissions due to fertiliser N application, the same smaller EF (0.46%) would seem appropriate for indirect N<sub>2</sub>O emissions due to re-deposited NH<sub>3</sub>, on the basis that climate and soil conditions affecting UK arable land are responsible for the difference between the average UK EF of 0.46% derived here and the 1% EF derived by IPCC for global use, and that climate and soil conditions are applicable to both applied fertiliser N and re-deposited NH<sub>3</sub> N.

Turning to NO<sub>3</sub>, modelling of leached NO<sub>3</sub> by Cardenas *et al.* (2013) has recently assessed the appropriateness of the 30% fraction of N that is assumed to be lost as NO<sub>3</sub> by leaching (0.75% of which is subsequently assumed to be emitted as N<sub>2</sub>O-N; IPCC, 2006). The fraction of fertiliser N leached was shown to vary considerably across the country due to rainfall and soil type, and (was only 9% for grass but) on average was 28% for UK arable agriculture, very similar to the assumption in the 2006 IPCC Tier 1 default value for indirect N<sub>2</sub>O emissions from leaching. Thus

the IPCC-assumed proportion of applied N lost as NO<sub>3</sub> may be appropriate for estimation of arable emissions at a large scale, e.g. for the UK national GHG inventory.

However, this approach appears less applicable to estimating GHG intensities for individual crops and their products, where applied N use may vary across a wide range, but leaching may not. The extensive work on NO<sub>3</sub> leaching over recent decades (e.g. Addiscott *et al.*, 1991; Lord, 1992; Lord *et al.*, 1999; Stopes *et al.*, 2002; ADAS, 2007) has shown a markedly non-linear relationship with N fertiliser application rate, and in most cases the effect of this N rate on leached N only becomes significant when it markedly exceeds N<sub>opt</sub>; amounts of N leached without fertiliser N are not substantially less than those with N<sub>opt</sub> (Lord and Mitchell, 1998). It should be noted that N fertilisers are predominantly applied in the spring when risk of leaching is minimal, and when the probability of almost complete removal by crop uptake and soil immobilisation is high (King *et al.*, 2001). Thus, it would appear misleading to relate leached NO<sub>3</sub> to the rate of fertiliser N applied at the scales of GHG accounting for a farm enterprise or a crop product.

Furthermore, the same substantial body of evidence (e.g. ADAS, 2007; Cardenas *et al.*, 2013) provides little support for the notion that leached NO<sub>3</sub> relates directly to quantities of crop residue N incorporated in soil, unless these residues have high N concentrations (e.g. Silgram and Harrison, 1998). Therefore, leaving aside residues with high N%, it would seem best, at least for LCA purposes, to disassociate indirect NO<sub>3</sub>-related N<sub>2</sub>O emissions from crop residue N, as well as from fertiliser N, and to relate them (through a re-formulated EF) to best estimates of NO<sub>3</sub>-N leached by crop type. Cardenas *et al.* (2013) modelled N leached according to crop type, and showed the importance of N removed, as well as N applied, but we have not adopted their crop type averages (their Table 6) for the smart EFs explored here because these are confounded with differences in organic manure applications and location (i.e. soil type and average over-winter rainfall).

In addition, a further substantial body of relevant evidence has been amassed (e.g. Dampney, 2000) in support of fertiliser N application rate recommendations (Defra, 2010a; Sinclair *et al.*, 2009; Sylvester-Bradley, 2009), a crucial component of which estimates supplies of soil-available N to crops (Kindred *et al.*, 2012). This indicates that, rather than estimating leached N from the fertiliser N and crop residue N applied to soil, as advised by the 2006 IPCC Tier 1 method, leached N would be best estimated through the three key variables of previous crop type, soil type and over-winter rainfall, i.e. fertiliser N recommendations invoke (explicit or implicit) estimates of leached N, and they are primarily discriminated by crop type. This approach creates quite different estimates from the default values prescribed by in the 2006 IPCC Tier 1 method. For example, if we take the comparison between cereals and legumes, IPCC (2006) currently assumes NO<sub>3</sub> leaching after wheat to be 30% of fertiliser N (176 kg ha<sup>-1</sup>; Table 1) plus crop residue N (104 kg ha<sup>-1</sup>; Table 10) applied i.e. 84 kg ha<sup>-1</sup> in total, and after pulses to be 30% of just crop residue N (53

kg ha<sup>-1</sup>; Table 10) i.e. 16 kg ha<sup>-1</sup> in total. However, work to support use of N fertilisers after pulses (e.g. Sylvester-Bradley and Cross, 1991; Kindred *et al.*, 2012) shows that soil N supplies are similar after peas and after beans, and that they are ~40 kg ha<sup>-1</sup> greater than after cereals (on retentive soils); a difference supported by MIN-NO experiments (Table 36), hence inferring that amounts of leached N are likely to be significantly larger after pulses than after cereals.

Comprehensive predictions of leached N as it affects N<sub>2</sub>O emissions are outside the remit of this project (and will be dealt with through Defra's GHG platform programme) but for the purposes of estimating mitigation impacts here, an overall average smart EF for N leached from arable land in the UK of 60% of SNS has been deduced from (i) the average N leached from conventional (non-organic) arable land estimated by Cardenas *et al.* (2013) as 46 kg ha<sup>-1</sup> N and (ii) an estimate of average leachable N, taken as being equivalent to the area-weighted average soil N supply estimated from national areas of each crop type and values of SNS for retentive soils in low-rainfall seasons given in the fertiliser recommendations (Defra, 2010; Section 3, Table A); see Table 44. [This approach avoids the confounding between crop type, location and manure use which affected individual crop type estimates made by Cardenas *et al.* (2013).]

Having estimated the national average N leached as a proportion of SNS, indirect emissions of N<sub>2</sub>O-N from N leached and their GHG equivalents could then be estimated according to each crop type, and for each combination of annual rainfall and soil clay content used to define the locational scenarios in Table 42. These are given in Table 44 and were then used in Section 10.3.3 to examine the impacts on GHG intensities of crop products of sourcing crop produce according to the location of its production.

Note that effects of tillage on N mineralisation (Silgram and Shepherd, 1999) and NO<sub>3</sub> leaching (e.g. Lord *et al.*, 1999) are much clearer than effects on N<sub>2</sub>O emissions. Thus a further consideration should be whether to adjust an indirect EF for leached NO<sub>3</sub> according to the timing (autumn versus spring) and intensity (direct drilling, minimum cultivation, or ploughing) of cultivations. Further deliberation and research may be required here.

**Table 44. Effects of crop type on autumn soil N supplies [SNS; in italics, from fertiliser recommendations for retentive soils (Defra, 2010, Section 3, Table A)], and hence on N leached over-winter, taking the national average leached N for arable crops in UK conditions (in bold) as 60% of autumn SNS (Cardenas et al., 2013). Also, taking national average conditions as equivalent to Scenario E in Table 42, estimates of the combined effects of crop type and locational scenarios from Table 42 on N leached and on consequent indirect GHG emissions [CO<sub>2</sub>e ha<sup>-1</sup>; N leached x 0.75% x 44/28 x 298].**

<b>Locational Scenario</b>		<b>A</b>	<b>B</b>	<b>C</b>	<b>D</b>	<b>E</b>	<b>F</b>	<b>G</b>	<b>H</b>	<b>I</b>
Annual average rainfall		Dry (500mm)			Moderate (700 mm)			Wet (1,000 mm)		
Soil type	(clay content)	5%	25%	45%	5%	25%	45%	5%	25%	45%
Over-winter leaching of autumn SNS		30%	20%	10%	70%	<b>60%</b>	50%	100%	100%	90%
<b>Benchmark crop type</b>	<i>Autumn SNS (kg ha<sup>-1</sup>)</i>	<b>N leached (kg ha<sup>-1</sup>)</b>								
Wheat – Feed	<i>70</i>	21	14	6	50	<b>42</b>	34	70	70	63
Wheat – Bread	<i>70</i>	21	14	6	50	<b>42</b>	34	70	70	63
Oilseed Rape	<i>100</i>	30	20	9	71	<b>60</b>	49	100	100	90
Sugar Beet - tops returned	<i>50</i>	15	10	4	35	<b>30</b>	25	50	50	45
Sugar Beet - tops removed	<i>40</i>	12	8	3	28	<b>24</b>	20	40	40	36
Vining peas	<i>110</i>	33	21	10	78	<b>66</b>	54	110	110	99
<b>Indirect GHG emitted from N leached (kg CO<sub>2</sub>e ha<sup>-1</sup>)</b>										
Wheat – Feed		74	48	21	174	148	121	246	246	221
Wheat – Bread		74	48	21	174	148	121	246	246	221
Oilseed Rape		106	68	31	249	211	173	351	351	315
Sugar Beet - tops returned		53	34	15	124	105	86	176	176	158
Sugar Beet - tops removed		43	27	12	99	84	69	140	140	126
Vining peas		117	75	34	274	232	190	386	386	347

#### 10.2.4. Crop residues

The default value for the 2006 IPCC Tier 1 method and many extant GHG emissions accounting studies assume that 1% of N in incorporated crop residues will be emitted directly as N<sub>2</sub>O, whilst recent meta-analyses of global data (Shan and Yan, 2013; Chen *et al.*, 2013) reach contradictory conclusions about the appropriateness of this approach.

The limited UK evidence in WP2B (Section 7) showed that direct emissions from the incorporation of removable above-ground crop residues were usually not significant (Figure 37). It is the case that the straw or haulm of combinable crops in the UK is almost always dry and dead at harvest, with no green tissue, so they almost always have a N concentration of <1% and, hence, a C:N ratio >40 before soil incorporation<sup>4</sup>. The extensive worldwide literature on decomposition of dead residues (e.g. Jensen, 1929; Janssen, 1996) supports the conclusion that rapid N mineralisation and then direct N<sub>2</sub>O emissions from incorporation of such above-ground residues is improbable. Furthermore, with the exception of legume nodules, the similarly low tissue N% of root materials (Wu and Arima, 1993; Gordon and James, 1997) indicates that significant emissions from below-ground crop residues from UK arable crops are also unlikely (Section 7.4.1). There is an argument that low N% crop residues might cause such oxygen consumption that soil conditions become more anaerobic and encourage denitrification, especially in soil with low WFPS (Chen *et al.*, 2013); however, this argument must be balanced against the likelihood that, particularly on arable land after harvest when SMN levels tend to be small anyway, low N% residues will also cause further SMN immobilisation, and so will further deplete the substrate for any denitrification. Thus, although direct evidence from MIN-NO experiments is limited and further UK research is clearly required, it is concluded here that over the 12 months after their soil incorporation, negligible N<sub>2</sub>O emissions from crop residues (other than those which are green) is likely to be typical, and that adoption of the IPCC (2006) approach in emission mitigation studies or for short-term LCA purposes may well be inappropriate. Whether the IPCC (2006) approach is appropriate for long-term studies and the generation of accurate national GHG inventories is discussed further below.

The only crop residues expected to cause direct N<sub>2</sub>O emissions in the short-term (within 12 months of soil incorporation) would be those with N contents exceeding ~2% of DM, such as sugar beet leaves or any other fresh green residues such as cover crops; these we will call 'green residues'. MIN-NO results and available UK data are insufficient to prescribe precisely at what point the N content of crop materials causes emissions to become significant. For the present analysis, only a crude assumption can be made that residues become 'emitting' when their N content exceeds 2%, hence almost always these are green rather than dead; however, it should be emphasised here

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<sup>4</sup> Note that the concept of Dead Organic Matter, or DOM, which includes crop residues, is already integral to the IPCC's broader approach to GHG emissions (IPCC, 2006, Chapter 5.2.2), so we will adopt the term 'dead' residues.

that the inclusion or exclusion of such a large emission should not in future depend on such a crudely defined boundary. It is likely that the timescales and extents of N<sub>2</sub>O-N emissions from crop residues will show a gradation across the scale of N contents from dead to lush and green, from ~1% to ~3%, and may depend on features of the residue other than its N content; consequently, this is a matter for further research. As an example it should be noted that, although vining pea residues here were measured at 1.6% N and their incorporation did not result in additional N<sub>2</sub>O emissions, the 'normal' N content of vining pea haulm may often exceed 2% (Table 39), so there is significant uncertainty about the inclusion of N<sub>2</sub>O-N emissions in GHG accounts for vining pea products, and this imposes a large uncertainty on GHG intensities of pea products. Nevertheless residues of the benchmark vining pea crop were 1.7% N (Table 10) so in the analysis below these are assumed *not* to cause direct N<sub>2</sub>O emissions.

As discussed in Section 7.4, given that direct emissions from residues only appear to occur with a minority of UK crops, and that appropriate data are not readily available for application of the IPCC (2006) approach (e.g. root:shoot DM ratios, as discussed in Section 5), the estimation of emissions from crop residues would benefit at least from some simplification, and possibly its whole underlying basis could be reconsidered. The approach explored here (but not advocated for direct adoption) is to regard dead crop residues as simply adding to total SOM, with its long-term propensity to mineralise N and then to sometimes cause associated N<sub>2</sub>O emissions; thus crop residues with low N% might be considered as contributing to 'background' emissions such as are estimated by the intercept of the MIN-NO model (Figure 53), or as are estimated by the Stehfest and Bouwman (2006) model. If background emissions were included in this way, it would be inappropriate to include additional emissions from crop residues with N<2%. However, if no 'background' emissions were included, it would seem inappropriate to exclude crop residue related emissions entirely, even though the extent of these emissions in reality is likely to be much more closely related to soil factors (such as cultivations, organic matter content, moisture and temperature) than to recent residue returns. Possibly there is a dichotomy here between using background-related estimates to meet the more specific requirements of LCA, and to guide mitigations, whilst using residue-related estimates to provide longer-term average emissions for the national GHG inventory.

Therefore, at least when accounting for GHG emissions at an enterprise or product level, subsequent analyses include a smart EF for background N<sub>2</sub>O emissions and exclude N<sub>2</sub>O estimates based on dead residues of combinable crops (with <2%N). This approach helps to avoid estimation of *reduced* emissions where straw has been removed rather than incorporated, for which there is extensive (indirect) evidence to the contrary (e.g. Scott, 1921). For crops where green residues (>2%N) have been incorporated (sugar beet, vining peas, cover crops, carrot tops, etc.; see Section 7.5.2) it seems appropriate for N<sub>2</sub>O emissions to be estimated in relation to the N

in above ground residues, and we propose that these should be considered *additional to any* background emission, and should be subject to the same EF (1% of total crop residue N; Table 37) as is assumed by IPCC (2006). For below-ground crop residues, it does not seem appropriate to include emissions from any crop species, except in the special case of legume crops whose N fixing nodules senesced prematurely (see Section 7.5.1). Adoption of the above approach might be appropriate for the UK GHG Inventory but, given its less common use in mitigation, this is less essential; the issues will be discussed further in relation to background emissions in Section 10.

### ***Leguminous nodules***

Whilst the MIN-NO project has not generated anything like sufficient evidence to generalise about quantities of N<sub>2</sub>O emitted due to decomposition of leguminous root nodules, the emissions observed support suggestions from elsewhere (Rochette and Janzen, 2005; Yang and Cai, 2005) that decomposing nodules may be the main source of N<sub>2</sub>O emissions from legumes, and these may, in some cases, be sufficient to justify inclusion in national GHG inventories and GHG accounts for crops and crop products. The conditions which govern whether nodule N is resorbed by the crop (hence will not cause an emission), or remains in the nodule for mineralisation in the soil (hence may cause an emission) are not clear but the few results discussed in Section 7.4.1 are consistent with the view that emissions largely arise from crops that are still green when harvested, either because they are harvested before plant maturity (as with vining peas) or because their haulm has remained green whilst their seeds have matured (as may have been the case for the winter beans at Terrington). Thus a smart EF of 0.8 kg N<sub>2</sub>O-N ha<sup>-1</sup> is proposed here and is tested below, just relating to vining peas, as suggested in Section 7.5.1. This EF may be compared to an emission of 3.75 kg N<sub>2</sub>O-N ha<sup>-1</sup> estimated for the benchmark crop (Table 10) by the 1996 IPCC (1996) method<sup>5</sup>, or zero by the 2006 IPCC method.

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<sup>5</sup> 10 t ha<sup>-1</sup> biomass x Fra<sub>CNCRBF</sub> (0.03 kg N kg<sup>-1</sup> dry biomass) x EF<sub>1</sub> (0.0125 kg N<sub>2</sub>O-N kg<sup>-1</sup> N input).

### **Crop residue estimation methodology**

Notwithstanding any changes of approach to dead crop residues, whenever direct N<sub>2</sub>O emissions from crop residue incorporation are still being included in evaluation of GHG emissions intensities for crops and their products (i.e. until zero emissions from dead crop residues is adopted, and for any *green* crop residues), it is proposed that above-ground (AG) residue N will be most accurately derived from crop yields via values of harvest index in the following steps:

<i>Calculation (CR: crop residue)</i>	<i>Wheat example (from Table 10):</i>
1. DM Yield = Yield x DM%	= 7.8 x 85% = 6.63 t ha <sup>-1</sup>
2. AG biomass = DM Yield / HI	= 6.63 / 50% = 13.26 t ha <sup>-1</sup>
3. AG CR biomass = AG biomass – DM yield	= 13.26 – 6.63 = 6.63 t ha <sup>-1</sup>
4. AG CR N = AG CR biomass x AG N concentration	= 6.63 x 0.6% x 1,000 = 40 kg ha <sup>-1</sup>
5. Removable CR N = AG CR N x removable proportion	= 40 x 50% = 20 kg ha <sup>-1</sup>
6. BG biomass = AG biomass x BG:AG biomass ratio	= 13.26 x 0.2 = 1.33 t ha <sup>-1</sup>
7. BG CR N = BG biomass x AG N concentration	= 1.33 x 0.9% x 1,000 = 12 kg ha <sup>-1</sup>

Our proposed best estimates for dry matter contents, harvest indices, N% concentrations, ratios of BG to AG and removable portions of AG residues for arable crops were given in Table 10 (right-hand columns). These recognise UK data availability, and avoid using some rather abstruse ratios advocated by IPCC, for which IPCC default values are inaccurate and local data are unavailable e.g. regression relationships between crop yield and AG biomass, over-precise ratios of BG to AG biomass, or N% of BG residues. Note that we do not recommend further research on estimation of BG residues, because the likelihood of significant direct N<sub>2</sub>O emissions from such residues is slight.

#### **10.2.5. Background emissions**

As discussed so far, the MIN-NO experiments have shown (i) significant N<sub>2</sub>O emissions without fertiliser N, these being affected by rainfall and soil clay content (Figure 46) and (ii) negligible emissions over a 12 month period relating to removable above-ground dead crop residues. Yet currently, most GHG accounting methods include estimates relating to the quantity of N in crop residues and not to factors more likely to affect longer term background emissions i.e. those likely to result from historic cultivation decisions, nutrient management and incorporation practices. We now argue that this approach could be reversed, primarily so that more appropriate and effective mitigation practices are encouraged. Again, these arguments possibly apply more to purposes of LCA than to purposes of national GHG inventory estimation.

Whilst general IPCC philosophy is that only anthropogenic emissions should be estimated, IPCC (2006, page 1.5) states that “For the AFOLU Sector, anthropogenic GHG emissions and removals by sinks are defined as *all* those occurring on ‘managed land’” where AFOLU stands for ‘agriculture, forestry and other land use’. However, the IPCC (2006) text dealing with N<sub>2</sub>O

emissions (Chapter 11, page 11.5) is not consistent with this, in that it does not include any estimate of the (often significant) N<sub>2</sub>O emitted from cropped lands *without any recent N inputs*. [NB. IPCC's footnote 7 on page 11.10 says background emissions "are not 'natural' emissions but are mostly due to contributions of N from crop residue."] Thus, in devising smart EFs here, it is necessary to make a choice between approaches.

Kim *et al.* (2013a) recently reviewed extant global data on background N<sub>2</sub>O emissions and, despite the expectation that undisturbed and ungrazed temperate ecosystems generally entail very "tight" nutrient cycling, with high N retention (Stark and Stephen, 1997; Müller *et al.*, 2002; Müller and Stephens, 2004), background emissions were variable and they could not discern a significant difference between agricultural and natural ecosystems. Their median values for emissions measured on a global scale were 0.21 kg N<sub>2</sub>O-N ha<sup>-1</sup> for 'natural' temperate grassland and 0.03 kg N<sub>2</sub>O-N ha<sup>-1</sup> for 'natural' temperate forest. Background emissions predicted by the MIN-NO model (Table 43) were larger, but it is not clear how much of these background emissions should be considered as anthropogenic.

Perhaps the key issue, leaving aside the academic question of past human influence, is the extent to which background emissions can be manipulated in future. Kim *et al.* (2013a) found the global variation in cropland emissions related to soil bulk density and pH, whilst emissions from pastures related to soil organic carbon. Climatic effects were also evident, corroborating the rainfall and soil type effects predicted in the MIN-NO model (Figure 53; Table 43). Thus at least some of the background emissions could be considered subject to long-term management, with land choice, cultivation intensity, liming, and crop residue (C) accumulation being potential considerations in any long-term N<sub>2</sub>O mitigation strategy. At present, these influences on N<sub>2</sub>O emissions are not addressed, yet could cause misleading inaccuracies in GHG accounts.

Thus our suggestion is that (at least) any UK Tier 2 method or enterprise or product-level GHG accounting should aim to estimate *all* N<sub>2</sub>O emissions from croplands; no N<sub>2</sub>O emissions from croplands should be excluded, because the concept of 'natural' emissions from croplands is untenable. Agricultural, and especially arable, land has been so fundamentally altered by man over so many years that all emissions must be deemed anthropogenic. Furthermore, exclusion of any emissions from cropped land is unhelpful to GHG mitigation, given that all emissions from managed land, even if deemed partially 'natural', may to some extent be mitigated. Whilst this approach may create an inconsistency with previous IPCC-based National Inventories such that comparisons become compromised, we propose that this approach is appropriate for GHG footprinting of crop products (where the ultimate purpose is to elicit appropriate mitigation practices), and it is possibly also appropriate for UK IPCC Tier 2 accounting.

Thus the challenge for GHG estimation is to estimate and attribute as accurately and precisely as possible the *total* N<sub>2</sub>O measured as emissions from arable land. If we take our modelled total N<sub>2</sub>O emission for Scenario E (with 25% clay and 700 mm rainfall, with say 200 kg ha<sup>-1</sup> fertiliser N and all straw incorporated) of 1.59 kg ha<sup>-1</sup> N<sub>2</sub>O-N, only 0.88 kg ha<sup>-1</sup> is attributable to the fertiliser, whilst 0.71 kg ha<sup>-1</sup> relates to the control (unfertilised) plots and must be attributed to something else. As described above (leaving aside the small minority of arable land on which green residues or fresh nodules are incorporated) extant evidence fails to relate such emissions to incorporated dead crop residues or roots. However, the concept of equilibrium SOM of long-term arable land (as recognised in IPCC, 2006, Chapter 5) requires that N losses from land must balance the unaccounted N inputs (such as in immobilised fertiliser N and ammonia deposition as well as dead residues). The majority of this N will almost certainly be lost in the UK in the form of leached NO<sub>3</sub>, but it is eminently feasible that a small quantity will be lost as N<sub>2</sub>O. Hence a possible approach that we test through the smart EFs below is the inclusion of the emission represented by the observed intercepts in Figure 46 (weighted UK average: 0.71 kg ha<sup>-1</sup> N<sub>2</sub>O-N, but adjusted by -0.02 kg ha<sup>-1</sup> N<sub>2</sub>O-N to avoid double-counting indirect emissions from NH<sub>3</sub> deposition) and suggest that this new smart EF is termed 'background' to indicate that it relates to observations from cultivated land without N inputs. 'Background' emissions may be considered as 'replacing' the emissions attributed by IPCC to dead crop residues. However, we suggest that it would be misleading to attribute this emission directly to crop residues. It is likely that emissions from control plots in fertiliser experiments derive from N mineralised from a wide range of sources; for instance the fertiliser N immobilised by soil organisms during growth of the last crop, the very variable releases from multifarious types of more recalcitrant SOM, and some recent deposition as NH<sub>3</sub>.

Although soils may have been maintained in arable production for many decades or centuries, there are always seasonal periods of mineralisation that approximately balance periods of immobilisation, and even long-term arable soils are seldom in perfect equilibrium year-on-year as regards SOM levels; small changes in SOM within the range defined as 'mineral soil' (i.e. 0-10%, and even changes too small to detect by repeated soil analysis) can cause large releases of soil mineral N; for example a decrease in SOM of 0.1% relates to ~150 kg ha<sup>-1</sup> mineral N. Mineralisation of SOM is common to all farming systems especially those involving tillage (Silgram and Shepherd, 1999), whether they involve applied N (as fertilisers, residues or manures), or not. Thus background N<sub>2</sub>O emissions may well be substantial, depending on whether recent conditions, particularly influenced by farming practices, effect any decreases in SOM. The Land Use, Land-Use Change and Forestry section of the national inventory (LULUCF; IPCC, 2003) attempts to deal with changes in SOM due to land use changes (e.g. arable to grassland), and in future it may recognise effects of finer differences in land uses (e.g. contrasting tillage practices used on arable land). However, arable tillage practices in the UK are seldom consistent over periods of several years because of its variable climate. For example, it is feasible that the large background

emissions observed at Edinburgh in MIN-NO experiments (Table 22 & Table 37) were due to SOM decreases caused by the combination of SOM, tillage and contrasting weather in 2010-2012. Thus GHG accounting that is to be accurate at a local scale would seem to need a means of recognising variation emissions due to variable SOM mineralisation. However, much more extensive research than has been conducted here will be necessary (i) to devise an accurate means of attributing specific quantities of N<sub>2</sub>O to specific causes of SOM decrease, and (ii) to assess the extent to which these might be mitigated.

We conclude that the 'background' smart EF is best estimated from N<sub>2</sub>O emissions from unfertilised plots, thus from the intercepts in Figure 46 here. However, these intercepts must be corrected for any indirect emissions resulting from atmospheric NH<sub>3</sub> deposition that they are likely to include. The approach for deriving the smart 'background' EF is to correct the intercepts according to IPCC EFs for NH<sub>3</sub> redeposition and the associated N<sub>2</sub>O emission scaled according to recommended N use on adjacent crops (see Section 10.2.3). This then allows indirect emissions still to be addressed explicitly by a bespoke smart EF, as with the IPCC (2006) approach.

Taking scenario 'E' as a 'typical' soil and rainfall combination for UK arable cropping the MIN-NO model predicts the background emission to be 0.71 kg N<sub>2</sub>O-N ha<sup>-1</sup>, or 1.12 kg N<sub>2</sub>O ha<sup>-1</sup>, or 332 kg CO<sub>2</sub>e ha<sup>-1</sup>. Correcting these values for indirect N<sub>2</sub>O emissions from redeposited NH<sub>3</sub>, assuming an average of 142 kg ha<sup>-1</sup> AN-N was applied in the area surrounding the MIN-NO experiments (the average for all tillage land from BSFP, 2013) and assuming smart EFs for NH<sub>3</sub> emission and indirect N<sub>2</sub>O emission derived in the section below, we get 0.69 kg N<sub>2</sub>O-N ha<sup>-1</sup>, or 1.09 kg N<sub>2</sub>O ha<sup>-1</sup>, or 323 kg CO<sub>2</sub>e ha<sup>-1</sup> as the smart EF for background emissions. This compares to N<sub>2</sub>O from residues of a benchmark crop estimated by the IPCC (2006) approach as being equivalent to ~350 kg CO<sub>2</sub>e ha<sup>-1</sup> (including indirect emissions; Table 10). Thus, whilst encouraging more effective decisions about incorporation of low-N crop residues and cultivations, the change from crop residue EFs to a smart EF for background emissions would only make a slight difference if adopted for the UK national N<sub>2</sub>O inventory.

#### **10.2.6. 'Standard' and 'smart' EFs for further analysis**

The full set of smart EFs devised in Sections 7.2.1 to 7.2.5 is shown in Table 45 along with a set of standard EFs, as was used to conduct the sensitivity analysis in WP1 (Section 5); the standard EFs are based on total GHG emissions associated with AN fertiliser manufacturing from Brentrup & Pallière (2008), and default Tier 1 EFs for direct and indirect soil N<sub>2</sub>O emissions from IPCC (2006). These two sets of EFs are then analysed and evaluated in subsequent sections.

By way of introduction, it is important to recognise that the smart EFs have been devised as the best summary of the new evidence from MIN-NO experiments and modelling, and from the

scientific literature which has been reviewed within the resources of the MIN-NO project. However, the smart EFs are supported by very different levels of provenance in terms of (i) numbers of observations, (ii) confidence in the chosen summary values, (iii) relevance of the tested conditions to UK conditions, (iv) credibility of the underpinning science, (v) degree of scrutiny and acceptance by the international scientific community, and (vi) validity for any particular purpose of GHG emissions accounting. Also, continuing UK research will soon add further evidence relevant to several of these smart EFs. Thus, the smart EFs adopted for analysis here should not be taken as suitable for unquestioning adoption for either the national GHG inventory, or GHG emissions accounting for the UK agricultural sector or for LCA of crops and their products at farm- and enterprise-scales. Rather, they should be regarded as best estimates from the MIN-NO project and a basis for further consideration before more formal or widespread adoption.

In an attempt to emphasise the contrasting status of the various EFs being compared in this analysis, all the EFs are presented in Table 45 with their main sources of evidence (as fully set out in Sections 10.2.1 to 10.2.3) and a colour code according to judgements about their provenance for use by the UK arable industry for GHG accounting (more than for the UK national GHG inventory): green indicates availability of adequate UK-relevant evidence; orange indicates limited UK-relevant evidence plus good generic support in the scientific literature; red indicates existence of only limited UK-relevant evidence or just generic supporting scientific literature. Whilst the standard EFs have had a high degree of scientific deliberation and scrutiny, they were largely devised for use in the Tier 1 method of GHG accounting at national and global scales. Indeed, one key motive for establishing the MIN-NO project was that the evidence base for many of these standard EFs relates poorly to UK conditions and circumstances. Thus, the colour code in Table 45 ignores the extent of international scientific deliberation and attempts to summarise the representativeness, the statistical certainty and the scientific credibility attached to each estimate for use by the UK arable industry and its customers in accounting for GHG emissions associated with crops and their products at the level of a farm or enterprise. Thus IPCC (2006) Tier 1 EFs are coloured red, and the smart EFs are coloured from red to green depending on how far it has been judged that their provenance for UK use has been improved. It is hoped that this approach will encourage appropriate evaluation of these smart EFs in subsequent work, and will discourage inappropriate use.

Effects of adopting the smart EFs are considered by successively applying them, in place of the standard EFs and calculating the resultant total GHG emissions and GHG emissions intensities of crops and their products. Effects are examined in five incremental steps, as set out in Table 45.

**Table 45. Summary of EFs and other coefficients used to estimate crop-related GHG emissions, with their ‘standard’ values as at the start of the MIN-NO Project, and their ‘smart’ values derived in Section 7.2. Column one indicates the steps in which ‘standard’ and ‘smart’ values are compared for their effects on total GHG emissions and intensities (through Section 7.3). Evidence sources and cell colours indicate the provenance of each ‘standard’ or ‘smart’ value for application to UK conditions (green, good UK-relevant evidence; orange, limited UK-relevant evidence plus generic supporting science; red, limited UK-relevant evidence or generic supporting science).**

Step	Description of EF or coefficient:	Units	‘Standard’ value	Evidence	‘Smart’ UK value	Evidence
1	Total GHG emissions associated with AN fertiliser manufacturing	<i>kg CO<sub>2</sub>e kg<sup>-1</sup> N</i>	6.31	Brentrup & Palliere (2008)	3.52	Fertilisers Europe (2014)
2	Direct soil N <sub>2</sub> O-N, due to N fertiliser use	<i>% N applied</i>	1%	IPCC (2006)	0.46% <sup>1</sup>	24 MIN-NO exp’ts & model
3	NH <sub>3</sub> volatilised & redeposited, due to AN fertiliser use	<i>% N applied</i>	10%	IPCC (2006)	3%	Chadwick <i>et al.</i> (2005)
	N <sub>2</sub> O-N emitted due NH <sub>3</sub> -N redeposited	<i>% N volatilised</i>	1%	IPCC (2006)	0.46%	By analogy with Step 2
4	N <sub>2</sub> O-N from ‘dead’ crop residue N (where N% <2%)	<i>% N returned</i>	1%	IPCC (2006)	0%	2 MIN-NO experiments and literature
	N <sub>2</sub> O-N from ‘green’ crop residue N (where N% >2%)	<i>% N returned</i>	1%	IPCC (2006)	1%	2 MIN-NO experiments and IPCC (2006)
	Below-ground crop residue N	<i>% N returned</i>	1%	IPCC (2006)	0%	Literature
	N from premature death of legume nodules	<i>kg ha<sup>-1</sup> N<sub>2</sub>O-N</i>	0	IPCC (2006)	0.80 <sup>2</sup>	2 MIN-NO experiments
	Background emission	<i>kg ha<sup>-1</sup> N<sub>2</sub>O-N</i>	0	IPCC (2006)	0.69 <sup>1,3</sup>	24 MIN-NO exp’ts & model
5	NO <sub>3</sub> -N leached due to fertiliser use or crop residue incorporation	<i>% N applied</i>	30%	IPCC (2006)	As in Table 44 <sup>4</sup>	UK Literature
	N <sub>2</sub> O-N emitted due to NO <sub>3</sub> -N leached	<i>%N leached</i>	0.75%	IPCC (2006)	0.75%	IPCC (2006)

<sup>1</sup> These are UK national averages. Additional smart EFs used to explore effects of location were given previously in Table 43.

<sup>2</sup> EF limited to crop practices causing premature death of N fixing nodules (e.g. vining peas).

<sup>3</sup> EF reduced by 0.02 kg ha<sup>-1</sup> N<sub>2</sub>O-N from values given to Table 43 to account for indirect N<sub>2</sub>O emissions from re-deposited atmospheric NH<sub>3</sub>.

<sup>4</sup> Crop-specific EFs for leached N in average UK conditions were adopted in place of EFs related to all N applied (fertilisers and residues) (Table 44) i.e. for 66 kg ha<sup>-1</sup> N after beans & peas, 60 kg ha<sup>-1</sup> N after OSR and potatoes, 42 kg ha<sup>-1</sup> N after any cereal, 30 kg ha<sup>-1</sup> N after sugar beet *without* crop residue (leaf) removal and 24 kg ha<sup>-1</sup> N after sugar beet *with* crop residue removal.

### 10.3. Impacts of smart EFs on GHG estimates

The impacts of the smart EFs above on GHG estimates associated with benchmark crops and arable crop products are now described, by comparing them with the standard GHG EFs for crop products from the benchmark crops already described in Section 5 (WP1; Figure 3 to Figure 5).

#### 10.3.1. National inventory emissions

It should be noted that, other than impacts on the national inventory of the change in the direct soil EF for N<sub>2</sub>O from fertiliser N, as described in Section 9 and summarised in Table 46, it is not possible to summarise impacts of all the smart EFs on the national inventory, since the 2014 inventory submission (reporting years 1990-2012) which was the most recent available to this project was still based on EFs from IPCC (1996), not on IPCC (2006).

**Table 46. The impacts of revised EFs for direct soil emissions after fertiliser use (in brackets), and calculation approaches, on estimated UK annual GHG emissions expressed in different contexts.**

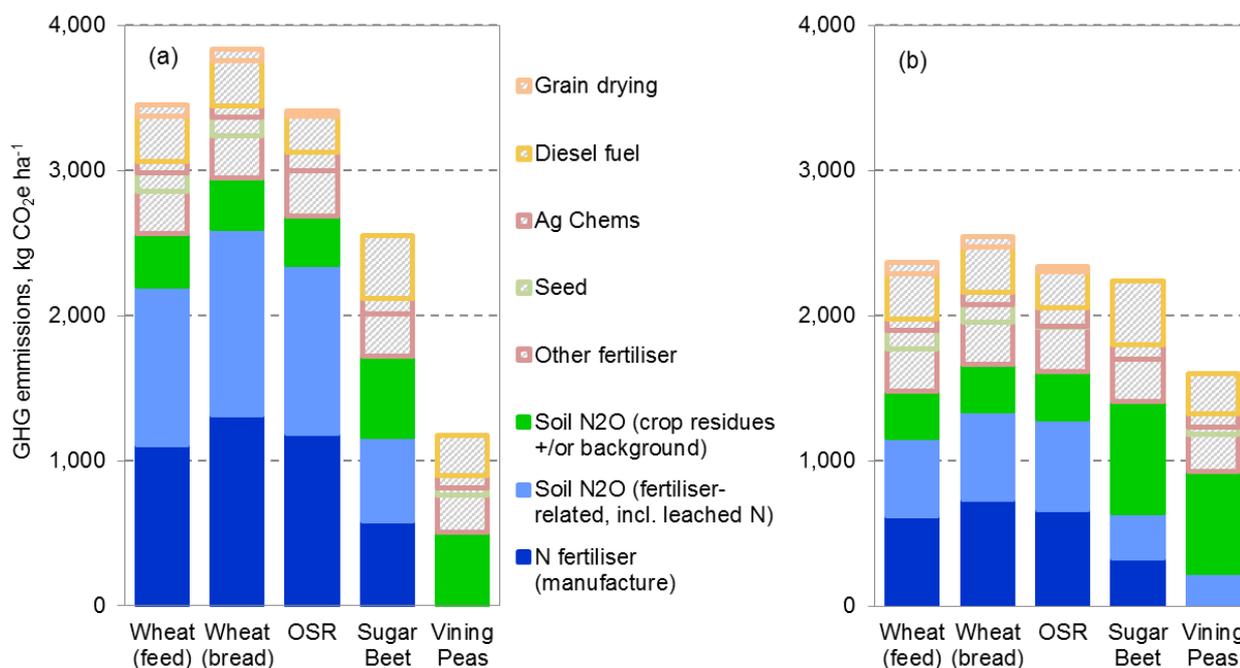
	Current 1996/2000 Inventory (1.25%) <sup>1</sup>	Using 2006 IPCC guidelines (1%)	Using new MIN-NO EF (0.46%)	<i>Reduction of MIN-NO EF cf. IPCC 2006 (or 1996)</i>
	<i>GHG Emissions (Mt CO<sub>2</sub>e year<sup>1</sup>)</i>			
UK total GHG	553			0.3% <sup>2</sup>
UK agricultural GHG	51			3% <sup>2</sup>
UK agricultural N <sub>2</sub> O	28.6	18.7	17.0	9%
UK direct N <sub>2</sub> O due to fertiliser	5.9	5.2	3.5	33%
UK direct N <sub>2</sub> O due to arable fertiliser	3.6	3.2	1.5	53%

<sup>1</sup> This EF applies to total N applied after 10% NH<sub>3</sub> loss. <sup>2</sup> These values relate to IPCC (1996).

It should also be noted that emissions from fertiliser manufacturing are considered in the national inventory as emissions associated with 'industry', rather than for AFOLU, so the mitigation of emissions from fertiliser manufacturing has not been effective in reducing the AFOLU inventory.

Effects on national annual N<sub>2</sub>O emissions for the UK's AFOLU sector of using the smart direct soil EF for fertiliser N were estimated as being 3.7 kt N<sub>2</sub>O-N year<sup>-1</sup>, or 1.7 Mt CO<sub>2</sub>e year<sup>-1</sup> in Section 9. Impacts on UK annual GHG emissions at different scales are shown in Table 46, and depend on whether the comparison is with the current national inventory (using IPCC 1996 methodology), or the approach to be used for the national inventory in 2015 (i.e. with the IPCC 2006 method). It should be noted that moving from IPCC 1996 to 2006 methodologies causes a large reduction in the assumed total annual GHG emissions from UK agriculture due in part to the reduction of indirect emissions associated with leaching following reduction in the EF from 2.5% to 0.75%, as well as the reduction in the EF for direct soil emissions after fertiliser N use from 1.25% to 1% (but

adjusted for NH<sub>3</sub> emissions). Overall, the 1.7 Mt CO<sub>2</sub>e year<sup>-1</sup> change in GHG emissions from adoption of the MIN-NO direct EF for arable crops would reduce UK annual agricultural N<sub>2</sub>O emissions by 9%, UK annual agricultural GHG emissions by ~3%, and total UK annual GHG emissions by ~0.3%. As mentioned earlier, if a background EF were adopted in place of direct emissions from all crop residues, there would be a small further reduction in the estimate of total annual GHG emissions from the AFOLU sector.



**Figure 54. Contributions to total GHG emissions associated with production of benchmark crops as defined in Section 5, estimated either (a) with all standard EFs or (b) with all smart EFs, as defined in Table 45. Contributions shaded grey are the same for (a) and (b).**

As illustrated in Figure 54, the changes in estimated N<sub>2</sub>O emissions (excluding those from fertiliser manufacture) were -33% for wheat, -22% for OSR, +5% for sugar beet and +128% or +156% for vining peas depending on whether the pea haulm was <2%N or >2%N. The overall effect of the smart EFs was to reduce the total estimated crop-related GHG emissions for all crops except vining peas, and to decrease the differences in total estimated GHG emissions between crops (Figure 54). The change in total estimated GHG emissions for wheat and OSR was about -30% and for sugar beet was -12%, whereas the change for vining peas was +37%, and this could have been +68% if incorporated pea haulm was >2%N content (with total AG crop residue N unchanged). The proportion of GHG emissions attributable to fertiliser N or other N<sub>2</sub>O emissions decreased for the fertilised crops but increased for vining peas. The latter effect was partly because the smart EFs assume that vining pea nodules cause N<sub>2</sub>O emissions and partly because they assumed vining peas cause some NO<sub>3</sub> leaching.

### 10.3.2. GHG intensities of crop produce

Table 47 shows the successive (cumulative) effects of using smart EFs (as defined in Table 45) on GHG intensities of arable crop produce, as harvested. The revised fertiliser manufacturing EF reduced total emissions by around 15% for wheat and OSR, and 10% for sugar beet, but it did not affect vining peas. The smart EF for direct emissions from fertiliser N used on UK crops (0.46%) reduced the overall intensities of produce by similar further extents. Effects of additional smart EFs were small for all four fertilised crops. However, for vining peas, whilst reducing NH<sub>3</sub> emissions had no effect, both further smart EFs (i.e. including a background N<sub>2</sub>O emission in place of an emission related to the soil-incorporation of low N% crop residues, and estimating leached NO<sub>3</sub> according to crop type, rather than to N applied) increased the overall emission intensities substantially.

**Table 47. Absolute and relative effects of successively applying smart EFs defined in Table 45 on GHG intensities of produce from benchmark crops estimated with standard EFs. CR = crop residue.**

Step	EFs used for GHG estimation	Feed wheat	Bread wheat	OSR	Sugar beet	Vining peas
<i>GHG emission intensity (kg CO<sub>2</sub>e t<sup>-1</sup>)</i>						
	Benchmark crops with standard EFs, as in WP1	443	518	974	45	294
1	With fertiliser manufacturing EF for AN (2014)	380	439	824	40	294
2	Plus 0.46% direct EF for N applied	323	368	688	36	294
3	Plus reduced EFs for indirect NH <sub>3</sub> emissions	314	357	666	35	294
4	Plus background replacing EF for CR <2%N	308	354	664	41	341
5	Plus leached NO <sub>3</sub> EF by crop, not N applied	303	344	668	39	399
<i>Change compared to standard EFs</i>						
1	With fertiliser manufacturing EF for AN (2014)	-14%	-15%	-15%	-10%	0%
2	Plus 0.46% direct EF for N applied	-27%	-29%	-29%	-19%	0%
3	Plus reduced EFs for indirect NH <sub>3</sub> emissions	-29%	-31%	-32%	-21%	0%
4	Plus background replacing EF for CR <2%N	-30%	-32%	-32%	-9%	+16%
5	Plus leached NO <sub>3</sub> EF by crop, not N applied	-31%	-34%	-31%	-12%	+36%

Adoption of all the smart EFs altogether reduced the GHG intensities of wheat grain and rape seed by about 30%. Sugar beet also showed a reduction of 12%. However, the effect of applying smart EFs were most dramatic for vining peas; inclusion of a direct EF of 0.8 kg N<sub>2</sub>O-N ha<sup>-1</sup> from N-fixing nodules increased total emissions by 17% and inclusion of emissions related to NO<sub>3</sub> leaching increased it by a further 20%. In addition, note that if vining pea haulm was deemed to contain >2% N, direct emissions associated with the benchmark N content of ~80 kg ha<sup>-1</sup> N (Table 10) would increase their GHG intensity by a further 31%, giving an overall increase of 68% compared to their benchmark intensity. Whilst the emission associated with root nodules is based on limited evidence, and the N concentration of their haulm is rather uncertain (Table 39), it is clear from the large potential effects of inclusion or exclusion, that these EFs could markedly alter the GHG footprints of pulse products. Given the amounts of N fixed and contained in the haulm, roots and nodules, given the reputation that legumes have for increasing soil fertility, and given the emissions measured in WP2B (Table 34; Table 35; Table 37) there is a significant likelihood that emissions

exceed those predicted by IPCC (2006), and it is clear that further research into emissions from all temperate pulse crops is required.

**Table 48. The relative impacts of successively applying smart EFs defined in Table 45 on GHG intensities of food products from benchmark crops estimated with standard EFs. CR = crop residue.**

Step	EFs used for GHG estimation	Chicken meat 1 kg	Loaf of bread 800g	Cookin g oil 1 litre	Bag of sugar 1 kg	Frozen peas 1 kg	Bottle whisky 70cl
<i>Change in GHG emissions compared to using standard EFs:</i>							
1	With fert. manufacture EF for AN (2014)	-3%	-7%	-13%	-4%	0%	-3%
2	Plus 0.46% direct EF for N applied	-6%	-13%	-25%	-8%	0%	-5%
3	Plus reduced indirect EFs for NH <sub>3</sub>	-6%	-14%	-27%	-9%	0%	-6%
4	Plus background EF, not for CR <2%N	-6%	-14%	-27%	-4%	+5%	-6%
5	Plus leached NO <sub>3</sub> EF by crop, not fert N	-7%	-15%	-27%	-5%	+10%	-6%

**Table 49. The impacts of successively applying smart EFs defined in Table 45 on GHG intensities of fuel products from benchmark crops estimated with standard EFs. CR = crop residue.**

Step	EFs used for GHG estimation	Wheat bioethanol	OSR biodiesel	Sugar beet bioethanol
<i>Change in GHG emissions compared to using standard EFs:</i>				
1	With fertiliser manufacturing EF for AN (2014)	-11%	-2%	-12%
2	Plus 0.46% direct EF for N applied	-22%	-14%	-16%
3	Plus reduced EFs for indirect NH <sub>3</sub> emissions	-24%	-16%	-16%
4	Plus background replacing EF for CR <2%N	-25%	-16%	-11%
5	Plus leached NO <sub>3</sub> EF by crop, not N applied	-26%	-16%	-13%

### 10.3.3. Effects of smart EFs on GHG intensities of crop products

The successive (cumulative) effects of using smart EFs (as defined in Table 45) on GHG intensities of arable crop products are shown for foods in Table 48 and for biofuels in Table 49. Effects depend directly on the assumed proportion of any product's GHG footprint that was attributable to harvested produce (e.g. grain). These proportions ranged from a minority (15-30%) for whisky, chicken meat and frozen peas, through a larger minority (35-45%) for bread, sugar, and bioethanol from sugar beet, to a large majority (70-85%) for biodiesel from OSR, bioethanol from wheat, and cooking oil. Thus the effects of all smart EFs combined are always smaller than for harvested produce, ranging from -27% for cooking oil to -6% for whisky, with the +37% change for vining peas becoming +10% for frozen peas (Table 48), and between -26% for bioethanol from wheat to -13% for bioethanol from sugar beet (Table 49).

## 10.4. Mitigation of arable N<sub>2</sub>O emissions

Section 5 presented the application of standard EFs to benchmark crops so as to estimate N<sub>2</sub>O emissions and their contribution to total GHG intensities of arable products. Then Section 10.3 showed how the new MIN-NO smart EFs affected these emissions and intensities. This section

now explores possible mitigation options and how these could impact on GHG emissions and intensities, estimated using either standard EFs or smart EFs.

#### **10.4.1. Summary of previous reviews, and mitigation options for analysis**

Four Defra projects over recent years have considered the possible range of mitigation methods for GHG emissions reductions in the agriculture sector (RMP4950 also known as MACC 1, an unpublished update of RMP4950 also known as MACC 2, AC0206 and WQ0106), and the feasibility of these was then reviewed in Defra Project AC0222. In summary, it was concluded that the potential contribution of the feasible options to the Government's GHG emissions targets would be less than first thought (RMP4950).

Clearly the majority of GHG intensities of crop produce and a sizeable part of the intensities of most crop products are N-related, even with the smart EFs rather than the standard EFs (Table 50). Thus, as was addressed in the Introduction (Section 4), reduced use of N fertilisers is an obvious option for N<sub>2</sub>O mitigation. However, as was also underlined at the outset, fertiliser recommendations advocate economically optimal N amounts for on-farm use so, by definition, reducing N use without also reducing the crop's N requirement will inevitably incur an economic penalty to the grower as well as to the fertiliser supply industry; it will also reduce global crop production which may indirectly stimulate replacement production elsewhere. Thus indirect effects of reducing N use are likely to arise, affecting land-use and GHG emissions elsewhere, in addition to the direct effects on farm profitability, and these may out-weigh any effects on global GHG emissions achieved by local N<sub>2</sub>O mitigation.

Hence, whilst it will be worth quantifying the costs and benefits of 'unilaterally' reducing N use, the conclusion (already reached in previous reviews) is that the key mitigation of reduced N use will be best achieved by prioritising and developing crops with inherently low requirements for N. Sugar beet is the clear example amongst the crops tested in the MIN-NO project, since the added N it requires to generate a similar amount of energy (whether for food or fuel products) is around half that of cereals or oilseeds. The example considered elsewhere (Defra Project AC0222) was the adoption of triticale in place of feed wheat but, more significantly, large research programmes in the UK and elsewhere in Europe are currently seeking to develop wheat and OSR varieties with low N requirements (e.g. Foulkes *et al.*, 2009; Gaju *et al.*, 2011; Barraclough *et al.*, 2014; Defra funded [WGIN](#) and [OREGIN](#)). Recent findings (Sylvester-Bradley *et al.*, 2015) already confirmed that triticale can yield as well as the best feed wheat whilst having a reduced N requirement (-30 kg ha<sup>-1</sup>); they also indicated that wheat varieties bred in Denmark (where fertiliser N use has been constrained for many years; Danish Environmental Protection Agency, 2015) yield similarly to the best UK feed varieties but with lower N requirements. As yet the food and fuel markets have inadequate systems to exploit the potential value of mitigating GHG emissions from agriculture in

this way. Even though low N use crops should have positive impacts on farm incomes, uptake may be slow initially because strategic investments are required, such as the development of appropriate infrastructure and trading systems to distinguish low N materials in the supply chain. However, the biofuels industry has invested in current research to test the value of low N crops in their processes; should results prove positive (as was predicted from initial small scale testing; Kindred *et al.*, 2010) there is a prospect of establishing a sizeable demand for low N crop produce e.g. triticale grain, hence of overcoming the initial market obstacles, and perhaps neutralising the discounts that currently tend to apply to low N grain in UK markets.

**Table 50. Portions of total GHG intensities of crop produce, and crop food and fuel products, that derive from fertiliser N or N<sub>2</sub>O, estimated with standard EFs used in WP1 and smart EFs used in WP5.**

Crop produce or product	Standard EFs	Smart EFs
Feed wheat	74%	63%
Breadmaking wheat	77%	65%
OSR	79%	69%
Sugar beet	68%	63%
Vining peas	43%	58%
Chicken	16%	10%
Bread	34%	23%
Cooking oil	68%	56%
Sugar	29%	25%
Frozen peas	12%	20%
Whisky	14%	7%
Wheat bioethanol	57%	47%
OSR biodiesel	59%	49%
Sugar beet bioethanol	27%	25%

Other simple mitigation options considered in recent reviews (above), such as fertiliser and manure planning and timing, were seen generally as being in keeping with implementation of ‘best farm practice’. ‘Normal’ and ‘best’ farm practice tend to differ only slightly so implementation of ‘best farm practice’ offered small GHG mitigation potential and some carried a net cost at a farm level. With economics and regulation being the key drivers for uptake of such mitigation practices, largest uptake was seen where changes were also positive for farm incomes, or where market forces took effect e.g. requirement for fertiliser spreader calibration for arable crop assurance. Increased awareness, and more local evidence of successful adoption, were seen as means of encouraging uptake of such mitigation practices.

Mitigation studies have generally employed Tier 1 EFs which are not location dependent, so they have not addressed the potential for mitigation by the location of crop production. However, adoption of Tier 2 EFs, such as might be derived from results in Table 43 and Table 44, raises the prospect of the food or fuel supply chains favouring produce from specific locations, especially

those with low rainfall and sandy soils. Differences in emissions between these conditions are apparently large, so it is inevitable that this means of mitigation will be explored at least by individual crop buyers seeking to reduce GHG intensities of their products, and it is possible that locational mitigation practices could have significant benefits globally through highlighting real geographical contrasts in N<sub>2</sub>O emissions. However, legislators and regulators must clearly recognise some complexities here because, at a global scale, GHG mitigation by crop location may become confounded if production of GHG-sensitive products tends to occur on low GHG-emitting land, whilst production of GHG-insensitive products becomes displaced onto high GHG-emitting land. It may even be that mitigation by location would prove *counter-productive* at a global scale, through displacement of crops causing high N<sub>2</sub>O emissions onto high GHG-emitting land. In attempting to resolve policies that optimise mitigation by location, it will be necessary to recognise that low GHG-emitting land (with low rainfall and soil with low clay content) tends to be less productive than high GHG-emitting land, so it will be important that crop yield effects are addressed directly in any schemes favouring locational mitigations. Unfortunately, the complexities are such that satisfactory resolution of locational mitigation issues is beyond the resources of this project.

The MIN-NO consortium has actively considered possible mitigation practices involving land drainage, cultivation practices including controlled traffic farming, and other aspects of good land management and has liaised with relevant researchers. Whilst there is much evidence that cultivations affect mineralisation and some evidence that compaction, such as in tractor wheelings, causes anaerobism hence risk of increased N<sub>2</sub>O emissions (Chamen *et al.*, 2015; Vermeulen and Mosquera, 2009), the challenge of defining verifiable cultivation practices that would accurately indicate worthwhile differences in N<sub>2</sub>O emissions is beyond the scope of this project. Suffice it to say that effects of cultivation timing, intensity and depth on N<sub>2</sub>O emissions are likely to show complex interactions with soil texture through the profile, SOM, recent cropping, fertiliser use and weather (e.g. Rochette *et al.*, 2008). Thus, to avoid over-sophistication of GHG accounting systems it may prove best to discriminate simply according to whether farming systems involve soil cultivation, or not (i.e. perennial cropping). Certainly this distinction accounts for a large difference in NO<sub>3</sub> leached, hence in indirect N<sub>2</sub>O emissions (Cardenas *et al.*, 2013) and, when all recent UK research is summarised (in current Defra Projects AC0114 and AC0116), it may also account for a significant difference in background emissions (Kim *et al.*, 2013a).

Previous reviews have considered whether some constraints on the timing of N applications, especially of fertiliser N, might offer a feasible means of N<sub>2</sub>O mitigation, and it is frustrating that modelling of the relevant MIN-NO data to show the detailed inter-relationships between N<sub>2</sub>O, NO<sub>3</sub>, WFPS and soil temperature (Section 6) proved beyond the resources of this project; an acceptable model of the daily dynamics of these variables should offer scope to optimise more confidently

between N timing strategies. However, our present synthesis of MIN-NO data indicates that soil N<sub>2</sub>O emissions are minimised when soils are cold and / or dry (Section 6.6), whilst crop N uptake tends to maximise in warm, moist soil conditions (King *et al.*, 2001). Thus, it seems unlikely that studies to mitigate N<sub>2</sub>O simply by 'clever' N timing will find a satisfactory solution; clever timing, even if weather forecasts can provide adequate accuracy, is likely to entail costs, risks and constraints that compromise any potential benefits. Ultimately best N timing options should probably be considered as part of mitigation strategies which also involve choice of fertiliser chemistry. This could usefully be a subject for a feasibility study.

Six recent UK experiments on fertiliser forms and nitrification inhibitors (Misselbrook *et al.*, 2014) indicated that N<sub>2</sub>O emissions from urea tend to be less than from AN, and that dicyandiamide (DCD) reduces direct N<sub>2</sub>O emissions after application of AN or urea by 39% or 69% respectively. Three of these experiments were on grass; mean effects from just the three arable sites were – 36% due to using urea rather than AN, and –34% or –73% due to using DCD with AN or urea respectively. Whilst this evidence-base is not large, and the DCD application technique (over-spraying straight after fertiliser application) was probably not the most amenable for on-farm adoption, worldwide evidence (e.g. Akiyama *et al.*, 2010) supports the conclusion that nitrification inhibitors generally cause substantial (>30%) mitigation of direct soil emissions of N<sub>2</sub>O after fertiliser applications. Given the commercial availability of urease inhibitors (to mitigate NH<sub>3</sub> emissions) as well as nitrification inhibitors, and the prospect that joint inhibition of urease and nitrification could have additive effects (Khalil *et al.*, 2009; Harty *et al.*, 2015), there are strong indications that inhibitors could provide substantial scope to mitigate N emissions to air overall, including direct soil emissions of N<sub>2</sub>O. Direct soil emissions of N<sub>2</sub>O are responsible for a substantial part of the GHG intensity of the wheat and OSR produce e.g. for wheat grain and rapeseed, 32-34% using IPCC (2006) EFs, or 15-17% using all the smart EFs (Table 45). Hence the use of specific N forms with inhibitors, perhaps along with fertiliser placement and specific timing constraints, may become attractive to some supply chains, even if they have no immediate economic advantage to the farmer (they are likely to increase fertiliser costs without significantly improving agronomic efficiency; Misselbrook *et al.*, 2014).

In light of MIN-NO findings concerning emissions from crop residues, the most obvious mitigation option would appear to be the removal of (removable) residues that are green (contain >2%N). Thus, for the crops considered here, removal of sugar beet leaves (and perhaps immature pea haulm) represent possible mitigations, but cereal or OSR straw removals do not. Removed leaves are most likely to be used for animal feed or in anaerobic digestion. However, given the apparent dependence of residue emissions on their N content, and costs of replacing major nutrients contained in green residues (the P and K in sugar beet leaves can be worth ~£300 ha<sup>-1</sup>) further possible mitigating practices might involve genetic, cultural or mechanical methods of reducing

residue N contents; for example, leaf protein might be extracted (for subsequent use in feeds) during the harvest process, or the leaves might be mixed with chopped straw before incorporation. (Dilution calculations show that about 5 t ha<sup>-1</sup> of beet leaves mixed with 2 t ha<sup>-1</sup> of wheat straw would reduce the N content of returned residues by about 0.4%, say from 2.2% to 1.8%.)

### ***Chosen mitigation options for further evaluation***

Having considered above the wide range of options that have been proposed hitherto for mitigation of GHG intensities of arable crop products, the next section evaluates the four options that appear to offer most scope for significant impact, and these are considered in apparent priority order for overall UK GHG mitigation, as follows:

- Development and adoption of **Fertiliser Systems** that maintain crop responses to fertiliser, whilst reducing N<sub>2</sub>O emissions. Many technologies contribute to such systems e.g. chemical inhibitors. Potential impacts will be explored by analysing systems having a range of total N<sub>2</sub>O emissions (from manufacturing, direct from soil and indirect) from 12 kg CO<sub>2</sub>e kg<sup>-1</sup> N applied, as was assumed before the MIN-NO project started, to near nil.
- **Reduced Rates** of N application, supported either by (i) chemical or genetic improvement in crops' responses to fertiliser N, such that less N is used i.e. improved 'N Use Efficiency' (NUE), or (ii) economic incentivisation of reduced N use, or regulatory constraint on N use. It will be important to determine the extent to which improved NUE, say by adoption of genotypes with minimum feasible N requirements, might mitigate GHG intensities. It will also be important to determine the maximum extent to which nil fertiliser use would minimise GHG intensities, so as to assess the associated costs of loss of production of constraining N use. This will inform supply chains and governments about the scale of incentives that would be necessary to support this most direct form of GHG mitigation.
- **Locational sourcing** of crop produce. An assessment will be made of the scope for supply chains to mitigate GHG intensities of crop products, assuming that they adopt Tier 2 EFs. This will take into account possible confounded differences in crop productivity.
- **Green crop residue disposal**, such as sugar beet tops, but also vining pea haulm (given the uncertainty about its N content). An assessment will include consideration of removal or in-field management or processing.

Both the benefits in mitigation of GHG intensities, and the economic costs will be quantified. Where options involve lost production, effects on indirect land use change (ILUC) must also be considered. Some mitigation options may be more applicable to initiation by a supply chain (e.g. low N genotypes) and others may be more applicable to initiation by regulation (e.g. constraints on N rate) so effects on both emissions and intensities must be assessed. As we cannot yet compute comprehensive national emissions based on IPCC (2006) EFs (see Section 10.3.1) we will not

explicitly examine national implications of each option, but emissions on a land area (ha<sup>-1</sup>) basis will be used to indicate national implications.

#### 10.4.2. Approach to quantifying mitigation impacts

Impacts of mitigations on emissions and intensities are estimated below by simply applying the standard and smart EFs already described in Section 10.3, or proposed mitigations of them, to the activity data for the benchmark crops described in Section 5, or proposed mitigations of them.

However, there are two aspects that require preliminary explanation: (i) how effects of ILUC were estimated and (ii) how economic effects of mitigations were evaluated through optimised rates of fertiliser N.

##### **Estimating ILUC and its effects**

Additional GHG emissions caused by ILUC (GHG<sub>ILUC</sub>; kg CO<sub>2</sub>e ha<sup>-1</sup>) were estimated in a similar way to that described by Kindred *et al.* (2008) and Berry *et al.* (2010), as:

$$GHG_{ILUC} = \left( \frac{(Y_{max} - Y_N) P_{ILUC}}{Y_{ILUC}} \right) e_{ILUC} \quad \text{Equation 11}$$

where  $Y_{max}$  is the maximum achievable grain yield (obtained from LpE curves with BER=0),  $Y_N$  is the yield achieved at a given N rate,  $P_{ILUC}$  is the proportion of lost production that will be met by ILUC (set at 0.5; this value is affected by complex global economics, particularly the supply of new land for tillage and the elasticity of demand for crop produce; Golub *et al.*, 2007),  $Y_{ILUC}$  is the yield of the area where ILUC occurs (world average yields were taken from FAOSTAT; wheat, 2.79 t ha<sup>-1</sup>; OSR, 1.70 t ha<sup>-1</sup>; sugar beet, 44.4 t ha<sup>-1</sup>) and  $e_{ILUC}$  is the EF for ILUC (set at 5,000 kg CO<sub>2</sub>e ha<sup>-1</sup> year<sup>-1</sup>).

##### **Estimating economic effects of mitigations**

Many fertiliser or genetic mitigations are intended to affect the way crop yields respond to fertiliser N, so mitigations were evaluated after fertiliser N rates had been optimised for either (i) economic margin ha<sup>-1</sup> (N<sub>opt</sub>), or (ii) minimum GHG intensity of crop produce. Note that N rate affects GHG intensity both by increasing output hence diluting the intensity (an effect that diminishes as rates increase) and by increasing GHG emissions, hence increasing the intensity (a linear effect).

Consequently, depending on the extent of each effect, there was always a certain N rate that minimised GHG intensity, termed N<sub>minGHG</sub>. The difference between margin over N cost at N<sub>opt</sub> and at N<sub>minGHG</sub> was used to evaluate the economic cost of a mitigation.

To assess the effects of fertiliser N rate, emissions were estimated at 1 kg ha<sup>-1</sup> increments from 0 to 400 kg ha<sup>-1</sup> N, according to the functions used conventionally (to derive fertiliser

recommendations) to describe the crops' responses in yield to applied N (Figure 55). Whilst the LpE function was used for cereal and OSR yields, as described in Section 6.4.7, sugar beet yields were described using a bilinear function, because Jaggard *et al.* (2009) found this fitted sugar beet responses better than the LpE function, and because this bilinear function has been adopted to underpin the fertiliser recommendations for sugar beet (Defra, 2010). The 'typical' values for parameters setting the shapes of the crops' responses (i.e.  $c$  and  $r$  for LpE, and slopes of the two lines for bilinear, as given in Table 51) were derived from ADAS-held data for wheat and OSR, and from the data of Jaggard *et al.* (2009) for sugar beet. However, intercepts and asymptotes were adjusted so that the functions gave the same values for  $N_{opt}$  and  $Y_{opt}$  as the benchmarks for fertiliser N and crop yield used in Section 5 (Table 1). GHG intensities were then calculated, both with and without  $GHG_{ILUC}$ , using the activity data for benchmark crops and using both standard and smart EFs. Crop yield responses, and emissions not associated with N fertiliser, were held the same for all estimates of GHG intensity.

**Table 51. Parameters of Linear plus Exponential (LpE; for wheat grain and OSR seed yields) or bilinear (for clean sugar beet yield, adjusted for sugar content) functions used to describe yield responses to fertiliser N.**

LpE parameter	Wheat	OSR	Sugar beet (bilinear)	
<b>A</b>	9.66	3.92	Maximum beet yield, t ha <sup>-1</sup>	57.1
<b>B</b>	-6.05	-1.85	Slope 1 <sup>st</sup> line, kg beet kg <sup>-1</sup> fertiliser N	154
<b>C</b>	-0.0040	-0.0005	Slope of 2 <sup>nd</sup> line, kg beet kg <sup>-1</sup> fertiliser N	0
<b>R</b>	0.99000	0.99076	Bilinear break point, kg ha <sup>-1</sup> fertiliser N	93

To assess the economic cost of operating at  $N_{minGHG}$  rather than  $N_{opt}$ , changes in economic margins (grain value less N fertiliser cost) were estimated assuming prices for AN fertiliser of £260 t<sup>-1</sup>, for wheat of £150 t<sup>-1</sup>, for OSR of £300 t<sup>-1</sup> and for sugar beet of £17 t<sup>-1</sup> (set to give a BER of 0.044 t kg<sup>-1</sup> N; Jaggard *et al.*, 2009).

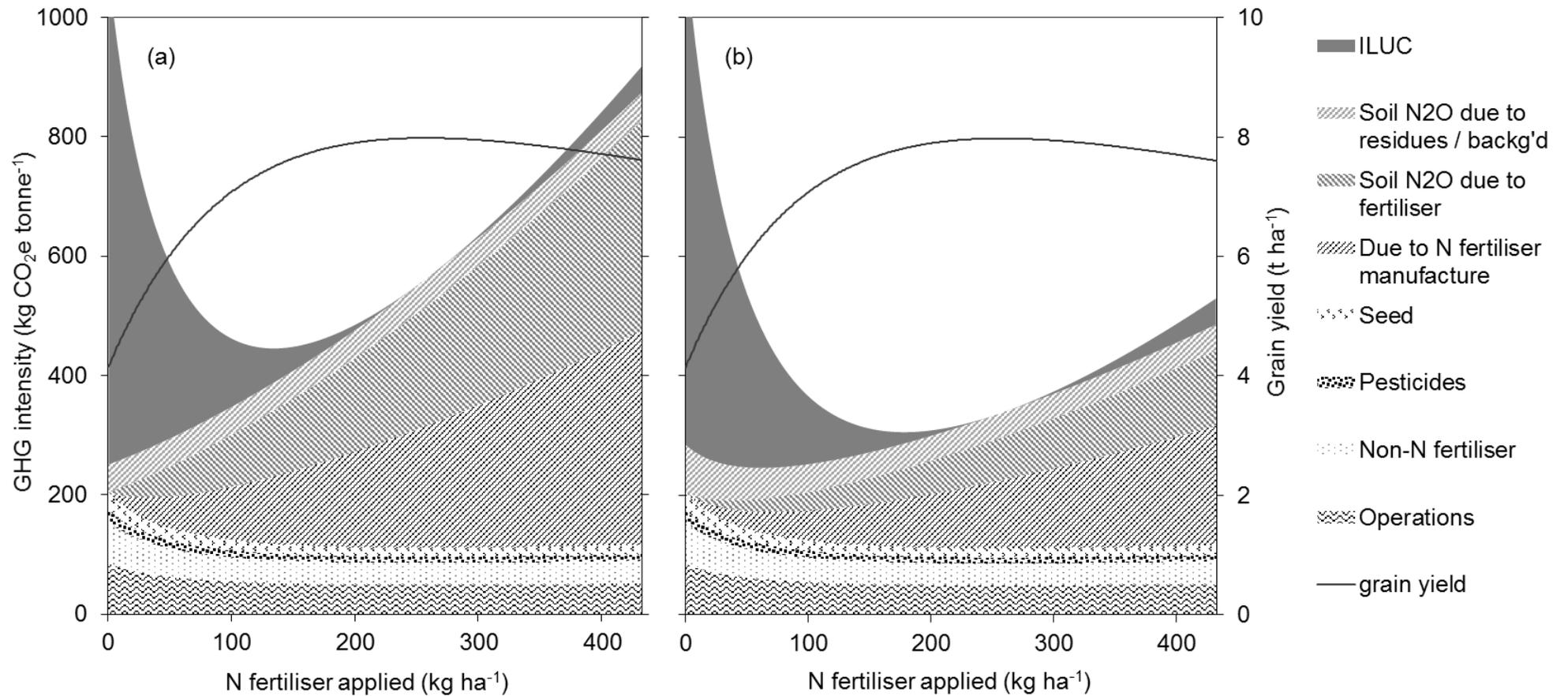
Overall GHG emissions (and intensities) for benchmark crops at  $N_{opt}$  with smart EFs rather than with standard EFs were 23% less for wheat, 27% less for OSR and 12% less for sugar beet. N rate affected wheat and OSR similarly so just the wheat example serves here to show how the various contributions to total GHG intensity were affected by N rate (Figure 55). With standard EFs, fertiliser N-related emissions were split about equally between manufacturing emissions and direct soil N<sub>2</sub>O emissions, whereas with smart EFs the manufacturing emissions tended to exceed the direct soil N<sub>2</sub>O emissions. However, the patterns of response in total GHG intensity were quite different between the scenarios because of a contrast in contribution of fertiliser N-related emissions to total GHG intensity (73% versus 44% at  $N_{opt}$  respectively).

Considering total intensity *without ILUC* effects, with standard EFs (Figure 55a), all fertiliser N rates caused increases, whereas with smart EFs (Figure 55b), small fertiliser N rates served to decrease the emission intensity because the large effect of N on yield was sufficient to dilute the smaller additional emissions. But then as fertiliser rates exceeded  $N_{\min\text{GHG}}$  at about  $70 \text{ kg ha}^{-1} \text{ N}$  (Figure 55b), the yield response to added N diminished, whilst the fertiliser-related emissions continued to increase proportionately, hence the emission intensity began to increase, and became near-linear as yield maximised.

Effects of *including ILUC* emissions were large, particularly with nil or small N rates, adding emissions of  $3,270$  and  $1,150 \text{ kg CO}_2\text{e ha}^{-1}$  at the  $N_{\min\text{GHG}}$  rates with standard and smart EFs respectively. Hence, even though ILUC effects diminished as yields increased, they made a large difference to GHG intensities and to  $N_{\min\text{GHG}}$ ; with standard EFs (Figure 55a)  $N_{\min\text{GHG}}$  increased from nil to  $133 \text{ kg ha}^{-1}$  due to ILUC, and with smart EFs (Figure 55b)  $N_{\min\text{GHG}}$  increased from  $70$  to  $178 \text{ kg ha}^{-1}$  due to ILUC.

Economic, crop production and environmental consequences of growing wheat, OSR and sugar beet at  $N_{\min\text{GHG}}$  rather than  $N_{\text{opt}}$  are shown in Table 52. Evidently growing wheat with nil fertiliser N would reduce GHG emissions substantially (by  $2.2 \text{ t CO}_2\text{e ha}^{-1}$  or 73%) but would also reduce yields substantially (by  $3.6 \text{ t ha}^{-1}$  or 47%) hence would reduce profitability substantially (by  $\text{£}416 \text{ ha}^{-1}$ ), so the maximum mitigation potential for the benchmark crop with standard EFs would be very costly to implement ( $\text{£}100 \text{ t}^{-1}$  grain) and the marginal cost of mitigating  $\text{CO}_2$  in this way would be high ( $\text{£}189 \text{ t}^{-1} \text{ CO}_2\text{e}$  saved). Similar effects apply to OSR (Table 52; Figure 56). However, the bilinear response to applied N of sugar beet led to more anomalous outcomes;  $N_{\min\text{GHG}}$  was nil with standard EFs but was equal to  $N_{\text{opt}}$  with smart EFs and with standard EFs plus ILUC effects (Figure 56). Thus adoption of smart EFs for sugar beet completely nullified any scope for mitigation by reduced fertiliser use.

The difference between  $N_{\text{opt}}$  and  $N_{\min\text{GHG}}$  was much affected by the EFs and GHG accounting approach used (i.e. whether ILUC was recognised and estimated, or ignored). Hence these also affected mitigation potentials (Table 52); use of smart rather than standard EFs increased  $N_{\min\text{GHG}}$  from nil to  $70 \text{ kg ha}^{-1}$  hence markedly decreased maximum possible mitigation to  $0.6 \text{ t CO}_2\text{e ha}^{-1}$  and decreased production costs of maximum mitigation to  $1.3 \text{ t ha}^{-1}$  and economic costs to  $\text{£}112 \text{ ha}^{-1}$  without much decreasing the marginal cost of  $\text{t}^{-1} \text{ CO}_2\text{e}$ .



**Figure 55. Effect of fertiliser N rate on GHG intensity of 'benchmark' wheat, showing components of the GHG emissions calculated (a) using Standard EFs and (b) using smart EFs. The grain yields are the same for both scenarios (full line; right axis).**

**Table 52. Results from assessing response in GHG emissions to N fertiliser for the benchmark crops of wheat, OSR and sugar beet defined in Section 5 using standard or smart EFs, without or with accounting for ILUC. NA = not applicable.**

	Excluding ILUC		Including ILUC	
	Standard EFs	Smart EFs	Standard EFs	Smart EFs
<b>Wheat</b>				
N <sub>opt</sub> (kg ha <sup>-1</sup> )	176	176	176	176
GHG emission at N <sub>opt</sub> (kg CO <sub>2</sub> e ha <sup>-1</sup> )	3,007	2,295	3,007	2,295
GHG intensity at N <sub>opt</sub> (kg CO <sub>2</sub> e t <sup>-1</sup> )	396	305	396	305
N <sub>minGHG</sub> (kg ha <sup>-1</sup> )	0	70	133	178
GHG saving at N <sub>minGHG</sub> (kg CO <sub>2</sub> e ha <sup>-1</sup> )	2,202	608	538	NA
GHG intensity at N <sub>minGHG</sub> (kg CO <sub>2</sub> e t <sup>-1</sup> )	242	269	416	305
Lost yield at N <sub>minGHG</sub> (t ha <sup>-1</sup> )	3.65	1.28	0.31	NA
Cost of N <sub>minGHG</sub> (£ ha <sup>-1</sup> )	£416	£112	£15	£0
Cost of N <sub>minGHG</sub> (£ t <sup>-1</sup> grain)	£100	£17	£2	£0
Cost of N <sub>minGHG</sub> (£ t <sup>-1</sup> CO <sub>2</sub> e saved)	£189	£185	£27	NA
<b>Oilseed Rape</b>				
N <sub>opt</sub> (kg ha <sup>-1</sup> )	188	188	188	188
GHG emission at N <sub>opt</sub> (kg CO <sub>2</sub> e ha <sup>-1</sup> )	3,040	2,204	3,040	2,204
GHG intensity at N <sub>opt</sub> (kg CO <sub>2</sub> e t <sup>-1</sup> )	878	640	878	640
N <sub>minGHG</sub> (kg ha <sup>-1</sup> )	0	42	113	180
GHG saving at N <sub>minGHG</sub> (kg CO <sub>2</sub> e ha <sup>-1</sup> )	2,352	838	938	46
GHG intensity at N <sub>minGHG</sub> (kg CO <sub>2</sub> e t <sup>-1</sup> )	410	527	863	639
Lost yield at N <sub>minGHG</sub> (t ha <sup>-1</sup> )	1.43	0.86	0.29	0.02
Cost of N <sub>minGHG</sub> (£ ha <sup>-1</sup> )	£289	£148	£30	£0
Cost of N <sub>minGHG</sub> (£ t <sup>-1</sup> grain)	£140	£56	£9	£0
Cost of N <sub>minGHG</sub> (£ t <sup>-1</sup> CO <sub>2</sub> e saved)	£123	£176	£32	£6
<b>Sugar Beet</b>				
N <sub>opt</sub> (kg ha <sup>-1</sup> )	93	93	93	93
GHG emission at N <sub>opt</sub> (kg CO <sub>2</sub> e ha <sup>-1</sup> )	2,553	2,250	2,553	2,250
GHG intensity at N <sub>opt</sub> (kg CO <sub>2</sub> e t <sup>-1</sup> )	45	39	45	39
N <sub>minGHG</sub> (kg ha <sup>-1</sup> )	0	93	93	93
GHG saving at N <sub>minGHG</sub> (kg CO <sub>2</sub> e ha <sup>-1</sup> )	1,164	0	0	0
GHG intensity at N <sub>minGHG</sub> (kg CO <sub>2</sub> e t <sup>-1</sup> )	42	39	54	39
Lost yield at N <sub>minGHG</sub> (t ha <sup>-1</sup> )	14.36	0	0	0
Cost of N <sub>minGHG</sub> (£ ha <sup>-1</sup> )	£175	£0	£0	£0
Cost of N <sub>minGHG</sub> (£ t <sup>-1</sup> grain)	£4	£0	£0	£0
Cost of N <sub>minGHG</sub> (£ t <sup>-1</sup> CO <sub>2</sub> e saved)	£150	NA	NA	NA

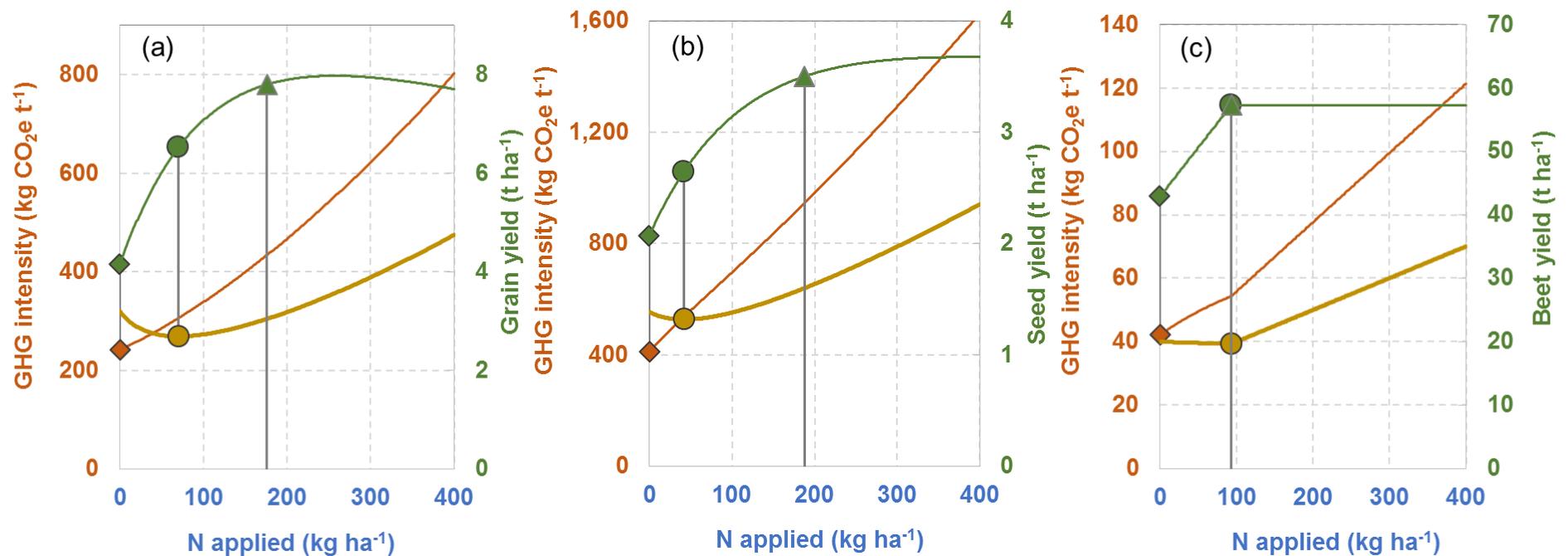


Figure 56. Effect of fertiliser N rate on crop yield (green line; right axis) with N optima ( $N_{opt}$ ; triangles), and on GHG intensity (orange) with  $N_{minGHG}$  (orange / yellow symbols) of 'benchmark' (a) wheat, (b) OSR and (c) sugar beet crops with the GHG emissions calculated ignoring ILUC and using standard EFs (orange line, diamonds) and smart EFs (bold yellow line; circles). Note that scales of all x-axes are the same but those of all y-axes differ.

Anything that increased GHG emissions  $\text{ha}^{-1}$  at zero-N (e.g. including invoking background emissions instead of crop-residue-related emissions) also increased the importance of yield in reducing GHG intensities per tonne, so that  $N_{\text{minGHG}}$  was increased. Any reduction in the EFs related to N fertiliser use reduced the penalties of using N, so again increased  $N_{\text{minGHG}}$ . However, the impacts of EFs and methodology on  $N_{\text{minGHG}}$  were modest and, in all situations including the use of non-linear  $\text{N}_2\text{O}$  EFs,  $N_{\text{minGHG}}$  remained less than  $100 \text{ kg ha}^{-1}$ , hence the costs (both economic and to ILUC) of meeting it remain too high for reduced N rate to be considered a viable option.

Regarding ILUC, whilst huge uncertainties apply to the assumptions used to estimate the amount of additional land that may be converted elsewhere in the world to meet a reduction in production in the UK, and huge uncertainties also apply to the carbon stored in any converted land and in how much of this is released, it is clear that the GHG consequences of land use change can dwarf emissions associated with fertiliser N and other crop inputs. Indeed, for all crops  $N_{\text{minGHG}}$  was very similar to  $N_{\text{opt}}$  when smart EFs and ILUC-related emissions were employed, and the impact on crop GHG intensity of reducing N rates by  $10 \text{ kg ha}^{-1}$  were negligible. It is possible that modest reductions in N fertiliser use could be incentivised, but impacts on final product GHG emissions would generally be small unless ILUC effects were ignored and standard EFs were used. Thus other forms of mitigation appear desirable.

### ***Reduced impacts of crop mitigations on intensities of crop products***

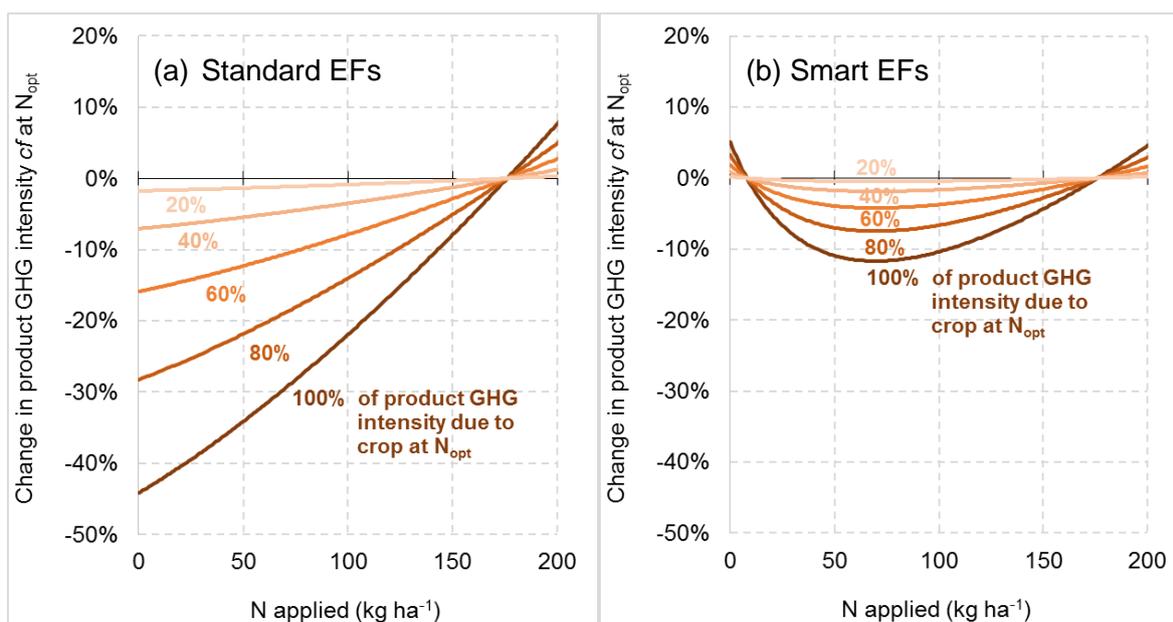
As already explained in Section 10.3.3, the proportions of the GHG intensities of crop products attributable to the crop itself varied across a wide range, from a modest minority (15-30%) to a large majority (70-85%). Thus the proportions of the GHG intensities of crop products attributable to fertiliser N (and other  $\text{N}_2\text{O}$  emissions) also varied widely, from 7% to 69% (Table 50). The change from standard to smart EFs only reduced these attributions for fertilised crops slightly, by about 8% on average.

Thus, to avoid having to reiterate how each N-related mitigation (discussed in the following sections) was diluted by the different non-N GHG emissions contributing to the GHG intensities of the crop products chosen as examples here, Figure 57 shows how the effect of fertiliser N rate (as an example mitigation option) affected the GHG intensity of wheat products, according to the proportion of the product's intensity that is attributable to the crop, from 20% to 100%.

Without any N applied, the GHG intensity of wheat itself using standard EFs decreased by 44% compared to wheat with  $N_{\text{opt}}$  applied. This mitigation became 35%, 26%, 18% or 9% as the proportion of a product's GHG intensity due to the crop diminished from 80% to 60%, 40% or 20%. What is more, using smart EFs, a maximum mitigation of only 12% was achievable for

unprocessed wheat through reducing fertiliser N use (to 70 kg ha<sup>-1</sup>), and this mitigation became only 9%, 7%, 5% or 2% as the proportion of a product's GHG intensity due to the crop diminished from 80% to 60%, 40% or 20%.

Thus, if GHG accounting were to adopt the smart EFs suggested here, impacts achievable through crop mitigations involving N-related emissions were generally small, and even with standard EFs and products such as cooking oil or biofuels, potential impacts were only modest.



**Figure 57. Effects of the proportion of a product's GHG intensity (ignoring ILUC effects) that is due to crop (at the economic optimum,  $N_{opt}$ ) on the extent to which reducing fertiliser N rate can mitigate that product's GHG intensity, for the benchmark wheat crop (a) with standard EFs or (b) smart EFs.**

### 10.4.3. Mitigation by N Fertiliser Technologies or Systems

#### ***Summarising possible fertiliser-related mitigations***

As is discussed in Section 10.4.1, there are many technologies that underpin the multiple components of N fertiliser systems, including (i) a range of N fertiliser manufacturing processes, (ii) many N fertiliser products, often chemically simple such as urea and AN but some involving biological as well as chemical sophistication, (iii) additives that may augment fertiliser performance and direct emissions, such as urease or nitrification inhibitors, (iv) physical fertiliser forms ranging from solutions and slurries to prills, granules and super-granules (e.g. Savant & Stangel, 1990), (v) application techniques from soil injection through soil surface sprinkling to foliar sprays, and (vi) application timings from all pre-sowing to delayed and repeated top-dressings. It is not possible to consider all of these components individually here. However, the range of possibilities and the dynamic nature of the modern fertiliser industry as influenced by commercial competitiveness and

modern global concerns for improved sustainability provide confidence that the momentum in fertiliser improvements will be maintained into the future.

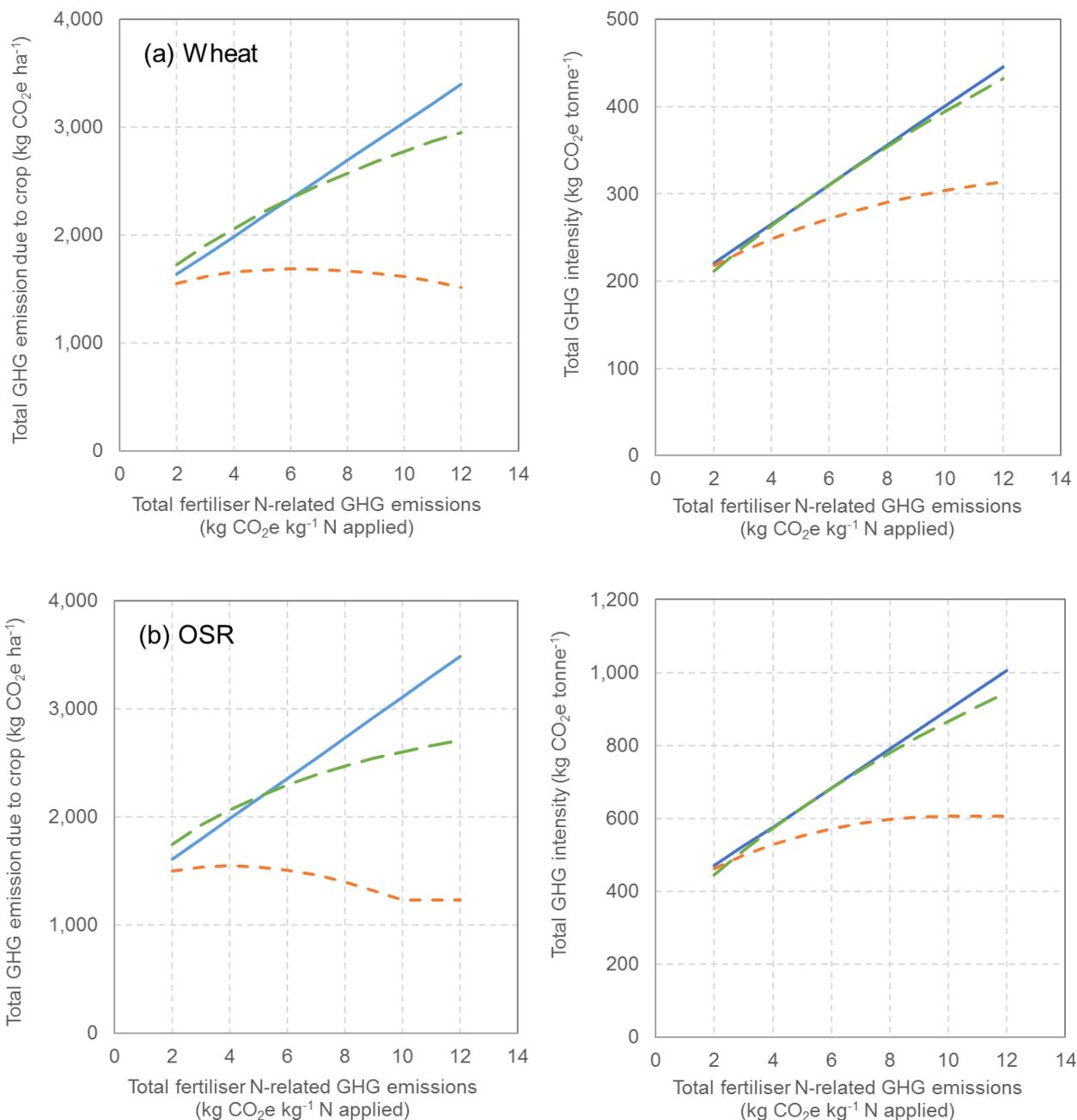
It is evident that manufacturing technologies have achieved substantial N<sub>2</sub>O mitigation even during the course of this project (Table 2) and recent work with nitrification inhibitors (Misselbrook *et al.*, 2014) demonstrates prospects for substantial further progress in reducing the total GHG emission per unit of N delivered to the crop (kg CO<sub>2</sub>e kg<sup>-1</sup> N applied). Thus the approach taken here is to explore the impacts on crop GHG intensities of further reducing the 'total GHG intensity of added N' (or 'TIAN'; kg CO<sub>2</sub>e kg<sup>-1</sup> N), a term used here to summarise all progress in mitigating N<sub>2</sub>O emissions attributable to delivery systems for extra crop-available N. Thus no specific mitigations are addressed here, but just a composite quotient of technological progress with nutrient-supply or fertiliser systems is examined.

### ***Effects of reducing the Total GHG Intensity of Added N ('TIAN')***

At the outset of the MIN-NO Project AN top-dressed on arable crops had a TIAN of 12.5 kg CO<sub>2</sub>e kg<sup>-1</sup> N using standard EFs (6.3 kg CO<sub>2</sub>e kg<sup>-1</sup> N from manufacture plus 6.2 kg CO<sub>2</sub>e kg<sup>-1</sup> N from direct and indirect N<sub>2</sub>O emissions, Table 45); this now compares with a TIAN of 5.7 kg CO<sub>2</sub>e kg<sup>-1</sup> N using the new smart EFs (Table 45). The difference is partly due to real mitigations in European fertiliser manufacturing processes (-2.8 kg CO<sub>2</sub>e kg<sup>-1</sup> N) and is partly due to the re-assessment of direct and indirect N<sub>2</sub>O emissions through the MIN-NO project (-4.0 kg CO<sub>2</sub>e kg<sup>-1</sup> N). This section therefore considers the impact of these and further such changes in TIAN. It is arguable what the ultimate feasible objective for TIAN could be but, as an extreme example, biological N fixation currently appears to constitute a technology and a practicable system for providing a crop's entire N requirements with near-nil GHG emissions. Thus the ultimate objective for TIAN should possibly be near-nil kg CO<sub>2</sub>e kg<sup>-1</sup> N. The concept of growing oilseeds with biological nutrition (e.g. soybeans) is already well established but the prospect of meeting the large and increasing demands starchy grains for food, feed or fuel without manufactured fertilisers is far from imminent. The current TIAN for AN fertiliser of 5.7 kg CO<sub>2</sub>e kg<sup>-1</sup> N is comprised of 3.5 kg CO<sub>2</sub>e kg<sup>-1</sup> N from fertiliser manufacturing and 2.2 kg CO<sub>2</sub>e kg<sup>-1</sup> N from in-field emissions (Table 45). Considering the recent progress in manufacturing, it is probably unrealistic to expect further rapid progress here; however, the promising results with inhibitors indicates a prospect of reducing in-field emissions by more than 50% and feasibly to less than 0.5 kg CO<sub>2</sub>e kg<sup>-1</sup> N. Thus the ultimate TIAN may be around 4 kg CO<sub>2</sub>e kg<sup>-1</sup> N for crops requiring manufactured fertilisers, and nil for other crops.

Note that, however reductions in TIAN might be achieved, *any* reduction (even by re-evaluation of EFs) also reduces the impacts (in GHG terms) of other mitigation options, such as genetically reducing crop requirements for fertiliser N. Thus as TIAN decreases, the potential impacts of all mitigation options should be re-evaluated; these interactions will be considered in Section 10.5.2.

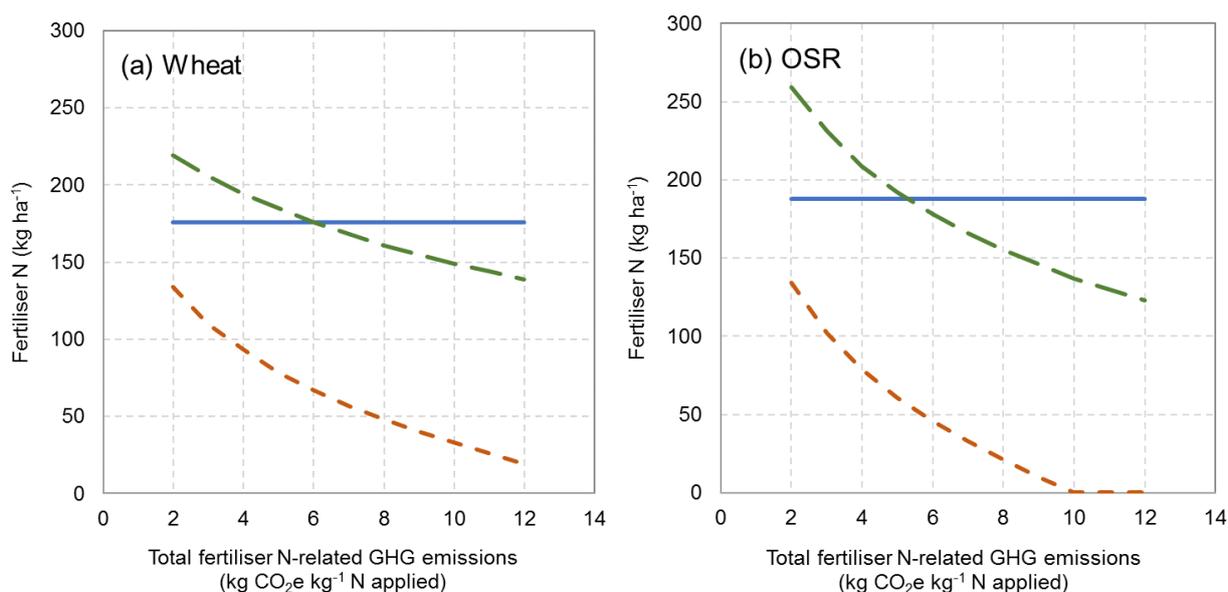
Assuming fertiliser use remains at the  $N_{opt}$  rate, reductions in TIAN cause linear and relatively large reductions in GHG emissions and GHG intensities (Figure 58). The reductions in TIAN during the MIN-NO project just due to mitigated fertiliser manufacturing (i.e. from 8.5 to 5.7 kg CO<sub>2</sub>e kg<sup>-1</sup>) have reduced total GHG emissions (ha<sup>-1</sup>) and intensities (t<sup>-1</sup>) by 18% for wheat and 19% for OSR. Further reductions in TIAN to say 4 kg CO<sub>2</sub>e kg<sup>-1</sup> N would reduce total crop emissions and intensities by a further 35% and 40%, giving a total potential mitigation (since before the project began) of 45-50%.



**Figure 58. Effects of mitigating total fertiliser N intensity (TIAN, kg CO<sub>2</sub>e kg<sup>-1</sup> N) on total GHG emissions (kg CO<sub>2</sub>e ha<sup>-1</sup>; left graphs) and crop produce intensities (kg CO<sub>2</sub>e t<sup>-1</sup>; right graphs) of (a) wheat and (b) OSR, with economic optimum N rates (full line, blue) or N<sub>minGHG</sub> with (green dashed line) or without (orange dotted line) accounting for ILUC.**

Effects of TIAN on  $N_{\min\text{GHG}}$  with and without accounting for ILUC are shown in Figure 59 both for wheat and OSR.  $N_{\min\text{GHG}}$  increases as TIAN decreases.  $N_{\min\text{GHG}}$  when ILUC effects are included significantly exceeds  $N_{\min\text{GHG}}$  when ignoring ILUC. The mean effect of accounting for ILUC on estimates of  $N_{\min\text{GHG}}$  was +108 kg ha<sup>-1</sup> N for wheat and +135 kg ha<sup>-1</sup> N for OSR. At the level of TIAN currently estimated with smart EFs (5.7 kg CO<sub>2</sub>e kg<sup>-1</sup>)  $N_{\min\text{GHG}}$  with ILUC was remarkably similar to the benchmark  $N_{\text{opt}}$ , whereas without ILUC  $N_{\min\text{GHG}}$  only approached  $N_{\text{opt}}$  when TIAN was less than 2 kg CO<sub>2</sub>e kg<sup>-1</sup> N.

Results for sugar beet are not included in Figure 59 because effects were simple. When TIAN exceeded 6 kg CO<sub>2</sub>e kg<sup>-1</sup> N (and when ignoring ILUC)  $N_{\min\text{GHG}}$  was nil kg ha<sup>-1</sup> but as TIAN decreased to less than 6 kg CO<sub>2</sub>e kg<sup>-1</sup> N,  $N_{\min\text{GHG}}$  switched to being the same as  $N_{\text{opt}}$ . However, when taking ILUC into account,  $N_{\text{opt}}$  and  $N_{\min\text{GHG}}$  were identical for the whole range of TIAN from 2 to 12 kg CO<sub>2</sub>e kg<sup>-1</sup> N. The implication here is that, for sugar beet, irrespective of ILUC, TIAN based on smart EFs (i.e. <6 kg CO<sub>2</sub>e kg<sup>-1</sup> N) is already less than the level at which efforts to incentivise or impose mitigation (by under-fertilising) is pointless.



**Figure 59. Effect of total GHG intensity of added N (TIAN) on  $N_{\text{opt}}$  (full blue line) and  $N_{\min\text{GHG}}$  of (a) wheat and (b) OSR either without accounting for ILUC (brown dotted line) or with accounting for ILUC (green dashed line). TIAN is estimated as 12.5 kg CO<sub>2</sub>e kg<sup>-1</sup> with standard EFs, and 5.7 kg CO<sub>2</sub>e kg<sup>-1</sup> with smart EFs.**

As is indicated in Figure 56 reducing TIAN caused the difference between  $N_{\text{opt}}$  and  $N_{\min\text{GHG}}$  to diminish, especially if ILUC effects were ignored, thus the potential to mitigate intensities of wheat and OSR or their products diminished as TIAN decreased (Figure 59).

#### 10.4.4. Mitigation by increasing N efficiency and reducing fertiliser N requirements

It was envisaged at the outset of the MIN-NO project that, for products such as biofuels where GHG intensities may directly affect product value, some form of incentivisation for reduced rates of applied N might prove viable. However, the previous sections have now shown that GHG intensities of arable products from UK farms tend to be significantly less than was first supposed, and thus the scope for incentivising reduced N rates now appears more limited. Nevertheless, fertiliser or N<sub>2</sub>O-related emissions remain responsible for the majority of the GHG intensities of all the benchmark crops examined here (Table 50) and, ignoring ILUC effects, these are responsible for from 7% to 56% of the intensities of the nine crop products considered. Reduced rates of applied N therefore remain an important option for arable GHG mitigation, even if it is unviable to incentivise them directly. Additionally, most applied N entails financial cost and environmental costs other than through GHG emissions (e.g. atmospheric pollution by NH<sub>3</sub> and water pollution by NO<sub>3</sub>). Hence, there are forces other than GHG mitigation which may serve to reduce N rates.

Improvements in efficiency of nutrient delivery to crops should in theory be incentivised simply by economic forces; fertilisers have an economic cost, more efficient fertilisers should be attractive because they allow less fertiliser N to be used without penalties in crop production, hence economics should favour more efficient fertilisers allowing reduced rates of applied N. However, only gross differences in NUE are detectable at a farm level (or even at the research level; Sylvester-Bradley *et al.*, 2014), so there is no great sophistication in the relationships between fertiliser N efficiency and rates of fertiliser N application. For example, there is no evidence that growers alter N application rate according to the choice of N fertiliser, and UK fertiliser recommendations, which are versed in terms of AN, tend to be optimistic about the efficiency of urea, so that application rates of urea-N are not greater than AN-N.

Other than chemical differences affecting NUE, there is evidence of wide variation in fertiliser N recovery, hence in NUE; very often this remains unexplained, but predictable effects are caused by genetic as well as soil differences (Sylvester-Bradley & Kindred, 2009). Of course crop species differ markedly in NUE, sugar beet standing out as the most efficient of the UK's arable crops. Sugar beet's efficiency is achieved (i) by acquiring more soil-derived N than other crops, through its growth pattern allowing more soil N mineralisation, and possibly also as a result of deeper rooting, (ii) by maintaining its green canopy during all of the brightest months, and (iii) by transferring less N from its leaves to its harvested organs than other crops, thereby leaf N is used for photosynthesis for as long as possible; typical N harvest index for sugar beet is about 50% compared to 70-80% for cereals. Whilst these large interspecific contrasts have received little scientific attention, much current research focusses on intra-specific varietal improvements in NUE, target traits including improved soil N recovery, fertiliser N recovery, and crop N utilisation especially through reduced grain (or other harvested organ) N concentrations (Foulkes *et al.*, 2009;

Sylvester- Bradley *et al.*, 2015). Hence genetical reductions in fertiliser N requirements are clearly feasible.

In analysing the scope for N<sub>2</sub>O mitigation through both chemical and genetic means, cereal yield responses to applied N have been modelled, in line with an ultimate limit to NUE set by capturing 120% of soil N available in spring (i.e. assuming uptake of N mineralised during growth, in addition to the normal N available at sowing on long-term arable soils, of 50-100 kg ha<sup>-1</sup>) and recovery of 90% of applied N (compared to the current average of 60%). It is not envisaged that the N required to create and maintain a full photosynthetic leaf canopy (160-200 kg ha<sup>-1</sup> for most crop species, including C3 and C4 cereals, oilseeds, pulses and sugar beet e.g. Sinclair & Horie, 1989) will prove tractable in the near future.

### ***Defining NUE mitigations by fertiliser chemistry or plant breeding***

Two scenarios were examined whereby NUE was deemed to have been improved through (i) better conversion of soil N into harvestable grain, such as through practices that encourage soil N mineralisation during crop growth, or through better deep rooting, termed 'soil NUE' or (ii) better conversion of fertiliser N into harvestable grain, such as through improved fertiliser chemistry or better topsoil rooting or reduced grain N concentration, termed 'fertiliser NUE'. In practice it is likely that a combination of better soil and better fertiliser NUE will be used to improve overall crop NUE, but these are chosen to exemplify contrasting effects on N<sub>2</sub>O mitigation.

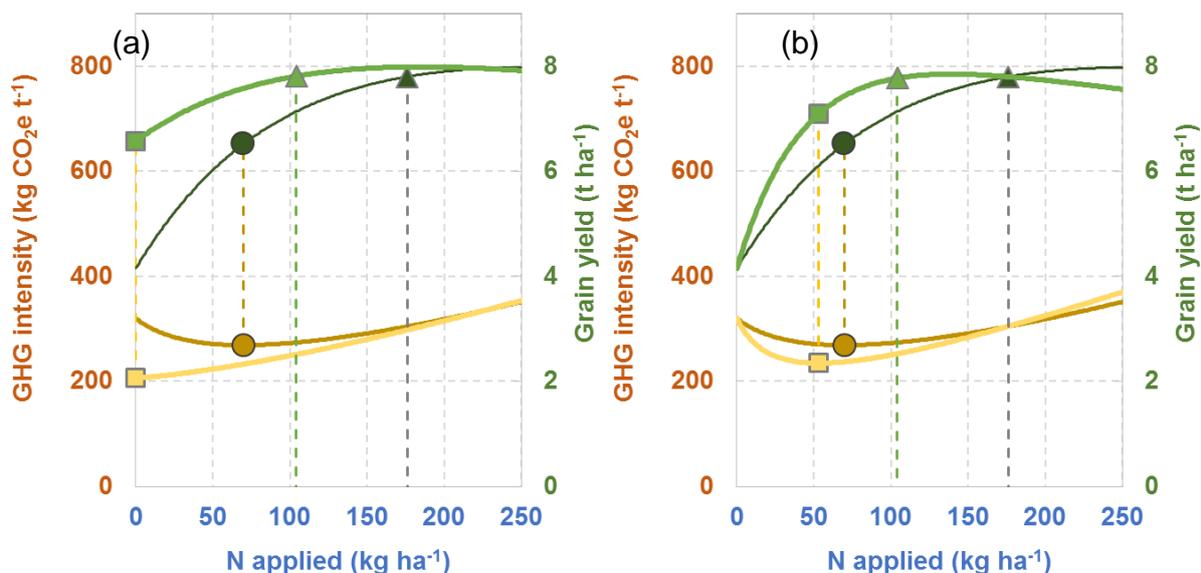
Parameter values for the standard LpE response curve for the benchmark wheat crop were adjusted (Table 53) such that Y<sub>opt</sub> of the mitigated crops were all held the same (7.8 t ha<sup>-1</sup>) but N<sub>opt</sub> values were reduced by 72 kg ha<sup>-1</sup> (40%) from 176 to 104 kg ha<sup>-1</sup>. Thus, assuming an arbitrary value for soil N supply of 60 kg ha<sup>-1</sup> in all cases, overall NUE was increased from 27 to 40 kg harvested DM kg<sup>-1</sup> available N. This is proposed as being close to the minimum feasible total supply of available N (from both soil and fertiliser) for a fully functioning photosynthetic canopy in UK conditions, even assuming near complete recovery of all available N, as discussed above. Of course crop yields well in excess of 7.8 t ha<sup>-1</sup> are commonly achieved and it is to be hoped that crop productivity enhancements will be large and frequent in future. However, for the purposes of making direct comparisons between high-NUE and benchmark crops here, yield effects have been nullified.

**Table 53** Parameter values and derived values for the LpE function used to describe the typical ‘benchmark’ wheat crop and comparable theoretical responses of crops bred for improved N Use Efficiency.

Parameter descriptions	Units	Benchmark wheat crop	Better Soil NUE	Better Fert. NUE
LpE parameter A		9.4000	9.1219	8.5720
LpE parameter B		-5.2500	-2.5500	-4.4220
LpE parameter C		-0.0040	-0.0040	-0.0040
LpE parameter R		0.9900	0.9900	0.9768
Optimum N applied ( $N_{opt}$ )	$kg\ ha^{-1}$	176	104	104
Grain yield with nil N applied	$t\ ha^{-1}$	4.15	6.57	4.15
Optimum grain yield ( $Y_{opt}$ )	$t\ ha^{-1}$	7.80	7.81	7.77
N Use Efficiency	$kg\ DM\ kg^{-1}\ N$	28.1	40.4	40.2
Total GHG emission at $N_{opt}$	$kg\ CO_2e\ ha^{-1}$	2,295	1,882	1,883
Total GHG intensity at $N_{opt}$	$kg\ CO_2e\ t^{-1}$	305	251	253
$N_{GHGmin}$	$kg\ ha^{-1}$	70	0	53
GHG saving at $N_{GHGmin}$ cf. $N_{opt}$	$kg\ CO_2e\ ha^{-1}$	608	598	294

### ***Effects of increasing NUE by chemistry or plant breeding***

This section considers separately the theoretical possibilities of just improving grain production from soil N by plant breeding (Figure 60a), or just improving grain production from fertiliser N by either plant breeding or new fertiliser technologies (Figure 60b). The two parts of Figure 60 show that better soil NUE (a) results in a shallow yield response to applied N, similar to that of the benchmark sugar beet crop (Figure 56c), whereas better fertiliser NUE (b) results in a much sharper yield response to applied N.



**Figure 60. Effect of fertiliser N rate on yield (green lines; right axis, with N optima, N<sub>opt</sub>; triangles) and GHG intensity (yellow lines, left axis, with N<sub>minGHG</sub>, yellow symbols) of ‘benchmark’ wheat (dark colours), and high NUE wheat (with N<sub>opt</sub> reduced by 40%, light colours) either by better soil N capture and conversion (a), or by better fertiliser N capture and conversion (b). GHG emissions were calculated using smart EFs, without accounting for ILUC effects.**

Crops with the latter characteristics would be much more sensitive economically to imprecise use of fertiliser N than benchmark crops or crops with high soil NUE; thus there might be more reluctance from growers of ‘N fertiliser efficient’ crops to fully exploit any improvements in fertiliser efficiency by reducing fertiliser N rates, than if crops had better soil NUE. In both cases, the 40% reduction in fertiliser N required caused less than a 20% reduction in total GHG emission or intensity. Only in the case of better soil NUE (Figure 60a) did N<sub>minGHG</sub> change substantially, thus scope for further mitigation by N rate was maintained with better fertiliser NUE but was halved with better soil NUE (Table 53).

#### 10.4.5. Mitigation by sourcing crop produce according to location of production

Table 43 (and to a smaller extent, Table 44) shows that soil (i.e. clay) and climate (i.e. rainfall) have large effects on N<sub>2</sub>O emissions. Thus sourcing crop produce say from sandy soils in East Anglia and avoiding produce from clay soils in Cornwall or Scotland may seem like an effective way of achieving an immediate reduction in the GHG intensity of a crop product. However, any GHG accounting scheme which differentiates EFs according to location, hence which allows estimation according to locational criteria, should also address the likely displacement effects of such sourcing. Taking the view that, with only 1-3% of arable land being left idle globally (Roques *et al.*, 2011), total production from all arable land is largely insensitive to the distribution of sub-categories of produce (i.e. crop produce displaced from low EF land will be fully replaced by produce from higher EF land) it becomes less obvious that such sourcing would achieve any real mitigation of GHGs from cropping at say a national or global scale. Possibly the global relevance of

locational sourcing may assume greater real impact as scales become larger, and as the remit of the mitigator assumes a larger scale. Take for example Europe’s policy of importing the majority of its protein from South America; it is widely accepted that European conditions are eminently suitable for production of protein crops (Linnemann & Dijkstra, 2002) and these may well have a significantly different GHG intensity from imported protein; thus the attractiveness of home production in terms of global GHG emissions will depend on the extent to which displacement of existing crops changes total GHG emissions, including those in South America (Weightman *et al.*, 2011). Hence, whilst the following analysis provides estimates of the mitigation potential of locational sourcing at a small scale within the UK, it should be recognised that the estimated effects should be discounted to some extent, to provide any global validity.

The impacts of using all the smart EFs, but adjusting these according to locational scenarios (i.e. 5%, 25% or 45% clay with low rainfall, and 500, 700 or 1000 mm average rainfall with 25% clay, as in Table 43 are now related to GHG intensities for Scenario E (25% clay, medium rainfall) for which EFs almost exactly correspond to national average EFs (Section 10.3.2). The smart EFs affected by soil type and rainfall concern (i) direct N<sub>2</sub>O emissions due to fertiliser N, (ii) background emissions, and (iii) indirect emissions associated with leached NO<sub>3</sub>. Results are shown in Table 54 to Table 56.

The changes in emissions from least (Scenario A, light soils with low rainfall) to most (Scenario H, and others with high rainfall) for crop produce were quite large, ranging from -20% in the driest conditions to +30% or so in the wettest conditions, thus a range exceeding 50%. For ‘sensitive’ crop products such as cooking oil or biofuels the range was ~45% from least to most, whilst for insensitive products such as chicken or whisky the full range was only ~8%.

**Table 54. Effects of smart EFs for different locations on GHG intensities of crop produce. The comparisons with Scenario E is for benchmark crops with all smart EFs applied (Step 5 in Table 45). Scenarios and EFs are defined in Tables 42 and 43.**

Location Scenario	Feed wheat	Bread wheat	OSR	Sugar beet	Vining peas
<i>Change in GHG intensity</i>					
A Light soil low rainfall	-20%	-20%	-22%	-17%	-23%
B Medium soil low rainfall	-16%	-16%	-19%	-14%	-21%
C Clay soil low rainfall	-12%	-12%	-15%	-10%	-19%
H Medium soil high rainfall	31%	30%	33%	26%	35%

**Table 55. Effects of smart EFs for different locations on GHG intensities of food products. The comparison is with benchmark conditions and all smart EFs applied (Step 5 in Table 47). Scenarios and EFs are defined in Tables 42 and 43.**

Location Scenario	Chicken meat 1 kg	Loaf of bread 800g	Cooking oil 1 litre	Bag of sugar 1 kg	Frozen peas 1 kg	Bottle whisky 70cl
<i>Change in GHG intensity</i>						
A Light soil low rainfall	-3%	-7%	-18%	-7%	-8%	-3%
B Medium soil low rainfall	-3%	-6%	-15%	-6%	-7%	-2%
C Clay soil low rainfall	-2%	-4%	-12%	-4%	-6%	-2%
H Medium soil high rainfall	5%	10%	27%	10%	12%	4%

**Table 56. Effects of smart EFs for different locations on GHG intensities of biofuel products. The comparison is with benchmark conditions and all smart EFs applied (Step 5 in Table 45). Scenarios and EFs are defined in Tables 42 and 43.**

Location Scenario	Wheat bioethanol	OSR biodiesel	Sugar beet bioethanol
<i>Change in GHG intensity</i>			
A Light soil low rainfall	-15%	-16%	-7%
B Medium soil low rainfall	-12%	-13%	-5%
C Clay soil low rainfall	-9%	-10%	-4%
H Medium soil high rainfall	23%	24%	10%

Note that, over-and-above the scenarios considered here, choice of crop location may in reality (whether or not all factors are recognised in GHG accounting schemes) affect GHG emissions through factors other than soil texture and rainfall, particularly SOM content (see Table 15) but also possibly crop yield; thus the location of cropping could be a major component of any strategy to mitigate GHG emissions. However, the consequences for GHG emissions of crop displacement should not be overlooked. For example, it might be possible to reduce the GHG intensities of biofuels by sourcing feedstocks from regions with low rainfall and soils with low organic matter and low clay contents (Table 55) but, because there is little surplus arable land in the UK, this would displace production of other products, mainly feeds or foods, to wetter locations with more emissive soils. Thus the advantages of location-based EFs might be significantly positive for particular businesses, or industry sectors, but could be near neutral for the global climate. Hence it will be a matter for administrators of GHG accounting schemes, whether locational factors should be recognised or not.

#### **10.4.6. Mitigation by residue removal**

Whilst sugar beet has a high NUE, 20% of its GHG emissions (estimated using smart EFs) is attributable to direct emissions from its green leaves returned to the soil at the time of harvest.

Ironically, retention of green leaves until harvest is partly responsible for the crop's high NUE since this allows the crop to maintain photosynthesis until harvest whereas, if the leaves were to die earlier, as occurs with most crops, the direct emissions might be avoided but the crop's productivity would be reduced. One key challenge for sugar beet GHG mitigation is therefore to find a way of avoiding direct emissions from leaves retained until harvest, the most obvious being leaf removal.

Thus far in Section 5 crop residues from vining peas have been treated as having  $N\% < 2$  and therefore no direct emission has been included in the GHG estimates for this crop. However, as has been discussed already (Section 7.4.2 and Table 39), vining pea haulm may often have  $N\% > 2$ , thus the analysis of mitigation effects in this section also includes an assessment of the impact of removing the pea haulm, should this be deemed to cause direct emissions.

The impact of removing green residues of benchmark sugar beet and vining pea crops was estimated assuming that 80% of AG residues could be mechanically removed and used for (say) animal feed or in anaerobic digestion; for the sake of this exercise it was assumed that any subsequent emissions could be attributed entirely to the subsequent product(s) rather than to the parent crop, and only major consequential impacts of residue removal were addressed<sup>6</sup>.

Benchmark residue quantities assumed here were as given in Table 10. GHG intensities were calculated with standard EFs and smart EFs except with the following changes:

- Additional GHG costs of residue removal were considered, including possible machinery and diesel use requirements, estimated at 12 litres per ha for both sugar beet and vining peas,
- Consequences for the following crop in removed P and K, assuming residues contained 0.5%  $P_2O_5$  and 2%  $K_2O$  in biomass, and that these would need to be replaced by fertilisers (i.e. assuming the soil P or K Index is below 3),
- Effects of the removed N on soil N supply and hence fertiliser N requirements of the following crop were calculated, assuming that the fertiliser N requirement of the following crop would increase by an amount equivalent to the ~10% of N in crop residues that would have become available (derived from Table 36 and Table 38).

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<sup>6</sup> As discussed in Section 5, all GHG emissions associated with crops should properly be 'allocated' or divided between the harvested produce and the crop residues, if these have some value; this was not done here. Also, counterfactual impacts of residue removal on GHG emissions should include not only diesel costs and costs of replacing with fertiliser the N, P and K removed in the residues, as considered here, but they should also include consequential effects on soil C sequestration, displacement of animal feed, or the more complex displacement of conventional energy by biogas, and displacement of N, P & K in fertiliser by digestate. Impacts of reduced residue return on SOM were not estimated because these were expected to be small (Nicholson *et al.*, 2014). The other considerations were deemed too complex for this exercise and were omitted, noting that, if they had been included, it is likely that they would have *enlarged* the GHG savings from crop residue removal as estimated here.

**Table 57. Effects of removing 80% of AG crop residues on GHG intensities of benchmark crops of sugar beet and vining peas, using either Standard EFs or MIN-NO smart EFs.**

	Sugar beet Standard EFs	Vining peas	Sugar beet Smart EFs	Vining peas
	<i>GHG emission intensity (kg CO<sub>2</sub>e t<sup>-1</sup>)</i>			
Benchmark crop	45	294	39	402
with 80% removed AG residues	37	203	33	327
with extra diesel GHG costs	38	211	34	335
plus GHG due to replacing soil P & K	39	225	35	349
plus GHG due to 'replacing' soil N	42	262	36	366
	<i>Change in GHG intensity from residue removal</i>			
from Benchmark	-21%	-45%	-19%	-23%
with extra diesel GHG costs	-19%	-39%	-17%	-20%
plus GHG due to replacing soil P & K	-16%	-31%	-13%	-15%
plus GHG due to 'replacing' soil N	-7%	-12%	-9%	-10%

At first glance, the estimated impacts of crop residue removal on intensities of sugar beet and vining peas appeared to be sizeable (Table 57). After accounting for estimated GHG costs, including diesel use and soil P and K replacement, residue removals (80%) reduced GHG emissions (estimated with smart EFs) by ~300 kg CO<sub>2</sub>e ha<sup>-1</sup> for sugar beet and ~250 kg CO<sub>2</sub>e ha<sup>-1</sup> for vining peas. However, both sugar beet tops and vining pea haulm contain substantial amounts of N with a low C:N ratio, hence the GHG account must recognise that some of this would become readily available to the following crop. From the results in Section 7.3, the overall average effect of residue removal on crop residue N returned (excluding the non-significant result at Terrington) was 87 kg N ha<sup>-1</sup> (Table 38) and the effect of this on the soil N supply for the following crop was 10 kg ha<sup>-1</sup> (Table 36). Since fertiliser recommendations (e.g. Sylvester-Bradley, 2009) presume that fertiliser N has 60% of the effectiveness of soil-available N we can presume a fertiliser N requirement of 19% of the removed residue N to replace its effect. The GHG costs of this fertiliser N significantly deplete any benefit of reducing direct N<sub>2</sub>O intensities by crop residue removal such that, with smart EFs it becomes only about 10% (Table 57). There may also be other benefits from returning residues that would also be lost, such as the return of other macro- (especially S and Mg) and micro-nutrients, and the possible enhancement of SOM. Residue removal could also give other possible negative impacts on soils, agronomy and farm logistics through the harvesting and collection process, including increased soil compaction. It therefore seems difficult to conclude that complete removal of green pea or sugar beet residues represents a worthwhile means of GHG mitigation.

On the other hand, there may be other means of mitigating direct emissions from green crop residues. Whilst green plant materials are always likely to contain substantial quantities of

nutrients, so devaluing the option of complete removal, means might be found to increase their C:N content, hence minimise direct N<sub>2</sub>O emissions, for example by:

- Changing crop processes either genetically or chemically (e.g. using desiccants) so that the green leaves senesce shortly before harvest, their N being resorbed into the harvested organ. Note that this might be positive for some crops (e.g. pulses) but detrimental for others; for instance, sugar beet quality might be reduced and beet prices might be discounted due to increased amino-N contents.
- Mixing the leaf material with chopped cereal straw; about 2 tonnes of straw mixed with 5 tonnes of green material could reduce its N content from 2.2 to 1.8%. Whether this would be logistically feasible and effective in mitigating N<sub>2</sub>O emissions needs to be tested.
- Extracting the leaf protein (say by pressing juice from the leaves within the crop harvester; e.g. Arlabose *et al.*, 2011) hence returning only a residue with increased C:N. The extracted protein could become a new product of crops harvested green (leaf protein has high quality for non-ruminant nutrition, so could be used to replace some imported soya-based protein).

Needless to say the logistical, mechanical and commercial issues governing each of these practices may render them impracticable. However, there are clearly potential means of mitigating emissions from green crop residues without residue removal, and these may merit further R&D, preceded and informed by more exact consequential LCA than it was possible to undertake here.

As was noted in Section 7, current farming strategies and EU 'greening' policies aimed at enhancing SOM increasingly involve catch crop cultivation and soil-incorporation; it would seem necessary to check that these practices are not enhancing GHG emissions, and if they are, to develop counter-measures that mitigate the N<sub>2</sub>O emissions. Also relevant here may be the significant emissions arising from legume root nodules killed prematurely. Such emissions may be more wide-spread than just from the modest areas of vining peas; for example, catch crops can include clovers and vetches which are leguminous, so involve destruction (mechanically or by frost) and incorporation of root systems likely to still have actively fixing nodules. These crops may cause significant additional N<sub>2</sub>O emissions. Genetic, chemical (e.g. PGRs) or mechanical treatments may be needed to ensure that root nodules senesce before crop destruction, so allowing resorption of their N by the shoot system, and so avoiding the risk of stimulating N<sub>2</sub>O emissions.

## **10.5. Discussion**

This chapter has brought together all findings from the MIN-NO LCA and sensitivity analyses (Section 5), experiments with fertilisers and crop residues (Sections 6 & 7), modelling and upscaling (Sections 8 & 9), and has worked through their implications for the best approach to GHG accounting, re-evaluated overall emissions, deduced the key areas for mitigation, and then

worked out their feasible impacts, using the re-evaluated EFs. The final (next) section will set out the overall conclusions of the project; thus Section 10.5 summarises and discusses the key findings on which these conclusions must be based by considering the validity and importance of (i) the smart EFs and (ii) the proposed mitigations.

#### **10.5.1. Proposed 'smart' EFs**

Given the uncertainties and variabilities in N<sub>2</sub>O emissions, no one method for calculating N<sub>2</sub>O emissions can be held to give the 'right' answer, nor can any currently used method be said to be 'right' or 'wrong'. Most current methodologies for calculating N<sub>2</sub>O emissions from crops use the IPCC (2006) approach with the EF for direct N<sub>2</sub>O emissions due to fertiliser N set at 1%. The 'Cool Farm Tool' is one exception to this, instead using Stehfest and Bouwman's (2006) approach (whereby emissions differ according to N fertiliser type). The IPCC approach has the merits of being defensible and reasonably consistent with national inventory reporting (although the current inventory is still based on the IPCC 1996/2000 approach). However, experimental results from this project indicate that, at least for most arable areas in the UK, N<sub>2</sub>O emissions are substantially less than the 1% IPCC default. It is also clear that real emissions vary with soil type and rainfall, as well as with N fertiliser rate. A more sophisticated approach to calculating N<sub>2</sub>O emissions would help allow these differences to be accounted for, perhaps ultimately affecting purchasing decisions and mitigation practices. However, the handling of any such approach needs careful consideration. For example, it only makes sense to base such calculations on long-term average weather data rather than to attempt 'real' annual calculations which would be very variable and would undermine confidence in any smaller but more consistent effects of GHG-targeted management strategies.

The model developed in WP3 (Section 8), whilst clearly needing validation on more soils and years, gives the best estimates so far possible for N<sub>2</sub>O emissions from UK arable soils. The model has very limited input data requirements (just fertiliser N rate, annual rainfall and soil clay content), so it could be built into GHG accounting methodologies quite easily. The model differs from using a Standard EF in two important respects; firstly, it includes an intercept which varies with rainfall and clay content; secondly, it is slightly non-linear in its response to N.

The non-linearity in the MIN-NO model is found to be limited with little impact on the calculation of N<sub>2</sub>O emissions. The use of linear EFs derived from the MIN-NO model, as calculated on a geographic basis and on a representative soil type basis (Figure 51 and Table 43), therefore provides a simpler method for incorporating smart EFs into GHG footprint calculations.

The new model, along with other evidence assembled above, shows emissions from UK arable cropping and intensities of arable crop products (except frozen peas) to be less than is estimated by most current GHG foot-printing procedures. A substantial reduction in GHG emissions from

manufacturing of AN fertiliser following investment in N<sub>2</sub>O abatement by the fertiliser industry has reduced total crop GHG intensities by around 15% for wheat and OSR and 10% for sugar beet (Table 47). From field experiments and modelling in this project the best estimate of direct N<sub>2</sub>O emissions associated with fertiliser N use on UK arable land is 0.46% of N applied, considerably less than the widely used default EF of 1% (IPCC, 2006), causing a further reduction in estimated GHG intensities of around 14% for wheat and OSR. A further small reduction (~2%) arises through downward revision of indirect emissions resulting from ammonia volatilisation and re-deposition. Removal of dead crop residue emissions from GHG estimates would give a further reduction of around 10%, but this is almost fully counteracted if background emissions are included.

As described above and set out in Table 45, some smart EFs are more certain than others, however it seems highly likely that (i) direct soil emissions of N<sub>2</sub>O from most UK arable land amount to significantly less than 1% of the fertiliser N it receives, and (ii) (working as much or more from the international literature than from the new UK empirical evidence presented here) low N% residues cause little or no N<sub>2</sub>O emissions within 12 months of soil incorporation. Also it seems indisputable that (iii) there is a significant background emission which (taking arable land as being by definition anthropogenic, and which is thereby eminently manageable by man) is a legitimate target for mitigation. However, further extensive deliberation will be required about the possible representation of background emissions within GHG accounts for arable cropping; on one hand, it would seem misleading to relate emissions to crop residues when no such emissions are detectable, whereas on the other hand it might seem misleading to represent full background emissions within a GHG account if these are excluded from accounts for other land uses such as forestry or moorland. Perhaps, after further research and review, it might be possible to identify a portion or amount of arable background emissions that could be deemed 'natural', hence could be excluded. However, this quantity is likely to vary with environmental conditions, so would not be easy to estimate.

As regards green crop residues, it would seem inadequate to adopt simple binary discrimination of green from dead residues because, if green residues are assumed to emit ~1% of their N as N<sub>2</sub>O, the effect on GHG estimates of deeming its residues to be green would often be large. For the benchmark vining pea crop here, this would add almost 100 kg CO<sub>2</sub>e t<sup>-1</sup> (23%) to the GHG intensity of its produce. Thus there will be a need for further research to identify typical N concentrations of the multifarious crop residues that might be considered green when incorporated, and this research will need to check relationships with direct emissions, at least over the initial period of crop residue degradation. It may be possible to develop a standard laboratory technique to effect this, say by mixing soil and residues in pots, to identify the most appropriate basis for residue discrimination.

Similar arguments to those for green crop residues apply to the emissions that appear to be associated with decomposition of prematurely killed legume root nodules. Associating a fixed emission of  $0.8 \text{ kg N}_2\text{O-N ha}^{-1}$  with such crops adds another 23% to the total GHG intensity of vining peas, a large change. Given the increasing use of legumes as cover crops which are usually destroyed prematurely, a way of expressing this effect on a sliding scale needs to be developed.

Whilst not encountered in the experiments and modelling within the MIN-NO project, it became clear in the sensitivity analysis conducted in Section 5 that the range of SOM in UK arable land classed as having 'mineral soils' (Defra 2010) i.e. 0-10%, has a significant influence on availability of soil-derived mineral N for crop uptake, hence this affects fertiliser use and GHG accounts significantly. The SOM range from ~4% to 10% is probably particularly important not only for its effect on fertiliser use but probably for effects on background soil  $\text{N}_2\text{O}$  emissions. At present there is no recognition of these major SOM effects in GHG accounting procedures. Thus a thorough review is required, probably along with some experimentation, to enable more exacting and sophisticated recognition of SOM effects on  $\text{N}_2\text{O}$  emissions.

#### **10.5.2. Impacts of feasible mitigations**

Having considered all obvious options, it appears that the potential for further mitigation of arable  $\text{N}_2\text{O}$  emissions is limited. On paper, GHG intensities of crop products could be minimised (by up to ~50%) through cuts in fertiliser N rates (to less than  $100 \text{ kg ha}^{-1}$ ). However, in practice such reductions in N use would reduce UK crop production which would then need to be met from elsewhere. Our calculations suggest that the indirect effects of such reduced production would almost entirely negate any GHG benefits (Table 52) via carbon losses through land use change elsewhere in the world (Figure 54). In any case, the financial costs of using reduced fertiliser rates would render this mitigation measure poor value ( $>£100 \text{ t}^{-1}$  GHG saved; Table 52) compared to mitigation options in other sectors (e.g. Lutsey & Sperling, 2009).

At N rates close to  $N_{\text{opt}}$  effects of varying N rate on yields are slight, and effects on GHG intensity are greater so it is just possible that modest reductions in N fertiliser use might be incentivised to allow commercial exploitation of any value in modest reductions in GHG footprints of some products, for example in cases where this would mean that a threshold for acceptance as a biofuel might be met. However, whilst price criteria dominate food marketing, it would seem unlikely that such opportunities will become common in arable crop production.

Initial examination of results in Table 58 would indicate that low intensity fertiliser systems, low N optimum growing systems and locational sourcing could all have similar and significant GHG impacts, each having the potential to reduce GHG intensities by  $>20\%$  for wheat and OSR and 12% to 17% for sugar beet. Whilst not quantified properly, it seems that crop residue removal

might also have a sizable (~15%) impact for the crops that have green residues at harvest. However, it should not be assumed that all impacts can be cumulative; as has already been shown in this section when mitigations relate to fertiliser N, each mitigation tends to reduce the potential impact of other mitigations, because mitigated emission factors are multiplicative not additive [(1-20%) x (1-20%) = (1-32%), not (1-40%)]. Only the impact of green crop residue removal is additive, however, green crop residue removal is only applicable to a minority of crops.

Thus, given that locational mitigation has dubious global value, the two main means of mitigation would seem to be reducing TIAN and reducing  $N_{opt}$ . If both of these mitigation methods were encouraged, it seems probable that they would jointly achieve little more than the impact of the more successful approach. Hence a choice of targeted mitigation methods would seem to be wise, before significant investment is made in their development. From the estimates of feasible impacts compared in Table 58, it seems certain that there is scope for some further mitigation, especially through fertiliser technology (incl. type, formulation and augmentation, placement and timing).

Given its negative interaction with low intensity fertiliser systems, it would seem that the investment in developing germplasm with reduced N demands must be justified by wider environmental impacts, rather than just through mitigating GHG emissions. For example better germplasm should also be justified by allowing higher productivity on soils with low fertility, cycling less N through livestock, and reducing  $NH_3$  and  $NO_3$  emissions.

Whilst considering genetic improvement of N Use Efficiency, some useful contrasts between species were made in the experiments conducted here, so the characteristics of the most efficient species, sugar beet probably merit further study. Possible traits that contribute to sugar beet's efficiency are spring sowing causing increasing soil N mineralisation in spring, deep rooting enabling deep soil N capture, and prolonged canopy life into the autumn enabling maximum photosynthetic returns from the investment in canopy protein. From this point of view, it is probably not by chance that sugar beet has green crop residues at harvest. Crops with long lived canopies will tend to be more N efficient and will tend to return their photosynthetic N to the land after harvest rather than export it to the grain; recent work on triticale has shown it to have a low N harvest index (Sylvester-Bradley *et al.*, 2015), whereas the parameters from which NUE is calculated indicate that NUE should be positively correlated with N harvest index.

**Table 58 Estimated potential impacts on total GHG intensity of each benchmark crop due to each individual mitigation measure analysed in Section 7, with the compounded effect of all measures.**

Mitigation with (metric for the mitigation)	Change assumed	Feed wheat	Bread wheat	OSR	Sugar beet	Vining peas, haulm <2%N	Vining peas, haulm >2%N
<i>Maximum potential change in total GHG intensity using all smart EFs &amp; ignoring ILUC effects</i>							
Low emission fertiliser N systems (TIAN, kg CO <sub>2</sub> e kg <sup>-1</sup> N applied)	-32%	-23%	-25%	-25%	-13%	0%	0%
Reduced N <sub>opt</sub> (kg ha <sup>-1</sup> N) due to better NUE	-50%	-21%	-23%	-23%	-12%	0%	0%
Locational sourcing (light soil, dry climate (Scenario A, Table 43)	<i>various</i>	-21%	-20%	-23%	-17%	-23%	-19%
Green crop residue removal (proportion of AG N removed)	-80%	0%	0%	0%	-16%	0%	-15%
<b>Maximum mitigation potential</b>		<b>-28%</b>	<b>-29%</b>	<b>-31%</b>	<b>-38%</b>	<b>-23%</b>	<b>-34%</b>

At the level of farm produce, the combined potential impact of all four mitigation methods mooted here are around -30% (Table 58). Given that further improvement in TIAN is likely to be in terms of field emissions rather than fertiliser manufacturing emissions, this maximum estimated mitigation of around 30% is also applicable to the arable component of the UK GHG Inventory for AFOLU.

If expressed at the level of the crop products analysed here (with 20-80% of their footprints attributable to crop produce), indicative values can be estimated for maximum potential mitigation of product intensities (Table 58); these range between -5% (e.g. for whisky) and -30% (e.g. for sugar) and appear to be relatively small impacts for the levels of investment (in fertiliser technology, plant breeding or husbandry innovations) required to achieve them.

Leaving aside the political forces that may affect mitigation choices, the mitigations suggested here contrast significantly in the technical, economic, administrative and, in some cases, legislative issues that might affect their uptake.

**Table 59 Estimated maximum potential impacts on GHG intensities of typical arable crop products (having contrasting proportions of their initial GHG footprints attributable to crop produce) due to the compounded effects of all GHG mitigation measures analysed in Section 7. Estimates for which no example products were studied in this project are in grey.**

Product type, according to contribution of crop GHG intensity to initial GHG intensity of crop product	Approx. crop contribution	Source of crop produce					
		Feed wheat	Bread wheat	OSR	Sugar beet	Vining peas, haulm <2%N	Vining peas, haulm >2%N
<i>Maximum potential change in total GHG intensity using all smart EFs &amp; ignoring ILUC effects</i>							
<b>Low</b> e.g. chicken, whisky, frozen peas	20%	-6%	-6%	-6%	-8%	-5%	-7%
<b>Moderate</b> e.g. sugar, beet bioethanol, bread	40%	-11%	-12%	-12%	-15%	-9%	-14%
<b>High</b> e.g. wheat bioethanol, biodiesel, cooking oil	80%	-23%	-23%	-25%	-30%	-18%	-27%

**Technical:** Some technologies already exist for reduced TIAN of fertiliser systems (e.g. through fertiliser forms and inhibitors), green crop residue removal (through anaerobic digestion) and locational sourcing, so these could become widely adopted within just a few years, whereas improvement of NUE to reduce  $N_{opt}$  will require much longer, requiring the development of (i) new crop trading and processing arrangements to derive the same products from different crop species (e.g. biofuels from a low  $N_{opt}$  species such as Triticale) and (ii) recognition by growers of low  $N_{opt}$  varieties of currently used species (e.g. by reporting of  $N_{opt}$  within Recommended Lists of varieties).

**Economic:** Given the slight commercial effects of GHG mitigation on the overall value of most arable products, incentives for their widespread adoption will tend to be small, so progress may only be slow. However, several examples already exist of high-profile processors and retailers seeking market advantage through labelling product GHG footprints, and the highly competitive nature of food supply could well encourage GHG mitigation initiatives to become more widely adopted.

**Administrative & Regulatory:** Mitigations must be recognised by the relevant GHG accounting system, and by any regulatory legislation, before they acquire commercial value. For instance incentives for biofuels (for which the primary purpose is GHG mitigation) are dependent on their specific GHG accounting systems being formulated and administered in such a way that these particular mitigations are recognised at the scale at which the mitigations might take place (from individual farm enterprise, to the supply base of a biofuels plant, to national). For example, an individual farm may be able to change its fertiliser system, or choose a better variety, but the effects of this will not be recognised if accounting occurs at national scale.

## 11. Conclusions

The main conclusions from the MIN-NO project are incorporated in the Executive Summary, at the start of this report (page 12). It was not possible there or here to develop all the issues and implications raised by such a large project and such a long report. In any case, this report is not the final word, since it precedes those from even larger concurrent relevant UK projects. It is our strong hope that firm generic conclusions will shortly emerge through subsequent publications arising from all of this work in refereed scientific journals. Thus the conclusions within the Executive Summary merely aim to extract in as succinct a form as possible the main findings from this work and the main messages for stakeholders in this project, so that they can be taken forward for further deliberation. It just remains here to list some of the challenges and uncertainties arising from the project and to clarify some of the uncertainties, by listing requirements identified for further research.

### ***Challenges and uncertainties***

- The experiments reported here were intensive and taxing. It is unlikely that such a large programme will be repeated. The experiments have created a greater body of internally-consistent data on arable N<sub>2</sub>O emissions than has been created previously. In particular the simultaneous measurement of emissions, substrate (SMN) and environment (water-filled pore space [WFPS] and temperature) across such diverse and well-documented conditions is rare and these data offer great potential, with further analysis, to improve understanding. The late change in modelling approach during the project obviated adequate interpretation of these data here, so there remains an urgent challenge to reveal the understandings that these data must hold.
- The contrasting emissions between the much wetter and cooler Scottish experimental site and the drier and warmer English sites were measured by different research teams and although the same protocols were followed by both teams, there is a remaining concern that some subtle differences in methodology may have exaggerated or even caused the differences. In particular, large background emissions found in Scotland were unexpected and unexplained; these need to be compared critically with other emission measurements from unfertilised land in both Scotland and other similar UK environments such as in Northern Ireland.
- The emergent UK biofuels industry found economic and process difficulties in realising its initial production targets, so the expected expansion of interest in low carbon feedstock production did not materialise during the project.
- Global top-down studies of atmospheric N<sub>2</sub>O levels indicate that existing global bottom-up estimates of N<sub>2</sub>O emissions, such as were made with standard EFs here, are probably about right (Davidson & Kanter, 2014). If it is now accepted that N<sub>2</sub>O emissions from UK

arable land are less than previously thought, and if similar findings occur in other countries, two consequences arise (i) emissions somewhere else must be greater than previously thought, or (ii) top-down estimates are wrong.

### **Requirements for further research**

The following requirements for further research are listed below. They are in approximate order of importance, in relation to the aims of this project.

- a. The MIN-NO model should be validated on independent data (from the UK or analogous conditions, if available), and then it should be used to develop appropriate emission estimates at level 2 the Nomenclature of Units for Territorial Statistics (NUTS2) in support of GHG accounting for biofuels.
- b. Direct measures are needed of N<sub>2</sub>O emissions according to its SOM content so that a much more sophisticated approach can be adopted to recognise relationships between emissions and SOM levels. Producers will need to know their SOM contents so they can better estimate their emissions.
- c. Direct measures are needed of N<sub>2</sub>O emissions on light land with heavy rainfall and irrigation. These conditions were omitted from those addressed here. Land with a range of SOM contents could be included in this work.
- d. Direct measures are needed of N<sub>2</sub>O emissions from land growing potatoes and vegetable crops since both of these (i) receive significant quantities of applied N, (ii) may involve incorporation of green crop residues, and (iii) are often irrigated.
- e. Direct measures are needed of N<sub>2</sub>O emissions from land with major contrasts in systems of cultivation, particularly ploughing, minimal cultivations, and direct drilling. Research should recognise that effects of these practices on GHG emissions are very likely to interact with soil type and climates (as noted above).
- f. A meta-analysis is required of UK background N<sub>2</sub>O emissions from all UK land, including arable, to determine influential factors, and means of mitigation. Parallel national deliberation is also required on whether a division can and should be made between natural and anthropogenic components of background emissions. This study may need to be augmented by experiments on effects of SOM, land management e.g. drainage, cultivation frequency and intensity, etc. The work must build on the findings of Kim *et al.* (2013a) on global background emissions.
- g. According to the IPCC 2006 methodology a survey of experts is required to obtain national data on the amounts of above-ground crop residue removed (Frac<sub>Remove</sub>). The IPCC advocate that if data on Frac<sub>Remove</sub> are not available, nil removal should be assumed. Thus N<sub>2</sub>O emissions will be over-estimated unless UK data on crop residues are collated.
- h. IPCC Default factors for estimation of N contents of crop residues (IPCC, 2006, Table 11.2) need to be improved and augmented to include all major UK crop types e.g. OSR and sugar

beet, as well as vegetable crops. Some defaults are based on sparse data, or inappropriate data from another climate or another species e.g. root crops are modelled on groundnuts! A literature review of UK and other relevant country data is proposed, followed by a programme of UK specific measurements of above and below ground N contents, harvest indices, etc. In particular, to improve emissions estimates associated with green crop residues, a survey of N concentrations in a range of modern crop products and feed-stuffs is required. This research should recognise the value of MAFF Booklet 2087 (MAFF, 1980) for many purposes and its need for updating.

- i. Research should be considered on possible means of reducing direct N<sub>2</sub>O emissions from green crop residues, for example by (i) extracting the leaf protein at harvest or (ii) by mixing leaves with straw.
- j. Whilst this project has suggested a radical new way of estimating indirect emissions due to leached NO<sub>3</sub>, no new evidence was generated. Even if the new approach is accepted without further research, there remain questions of how leached NO<sub>3</sub> depends on SOM, crop residue N contents, soil cultivations and other factors. Research to investigate this would best be coordinated with work on effects of SOM on SMN, soil nitrogen supply (SNS) and, hence, crop N requirements.
- k. The estimate made here of legume nodule emissions is highly uncertain due to scant evidence. A review is required of factors affecting nodule biomass production, nodule N concentration, longevity and decomposition. This review should target species differences and should be followed by experimentation on nodule N mineralisation, and then soil transformations (including the evolution of N<sub>2</sub>O) and plant acquisition of this mineralised N.
- l. The MIN-NO Project has generated a rich dataset describing the dynamics of top-soil available N following fertiliser N application. Research is required to model this dataset so as to understand how crop recovery of available N might be improved, as well as to understand N<sub>2</sub>O emissions and mitigation opportunities, for instance by timing N applications in relation to soil moisture status, or altering N forms or inhibiting soil N transformations.
- m. A desk study is required on locational GHG mitigation to determine whether effects of more localised GHG accounting are likely to be positive, neutral or negative at national or global levels. It is feasible that Tier 2 GHG accounting could encourage counter-productive practices, say by high emitting crops being displaced onto high emitting land, or favouring production on low GHG land which is also less productive. This work could examine data over different scales from the sub-national influence of a biofuels plant to the transcontinental effect of Europe's protein supply coming from South America.
- n. Research is required to review the key repercussions of the constraints on N use in Denmark. For the UK this represents an invaluable and unique experiment at a national scale, so there are major opportunities to improve understanding, as well as highlighting likely outcomes of contrasting policies. There are major analogies between the farming and

the financial conditions in the UK and Denmark. One evident outcome of Danish N constraints is the breeding of more efficient wheat varieties; there are also effects on the livestock feed industry and protein importation.

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### **13. Acknowledgements**

We are grateful to M and L Blake-Kalff (Hill Court Farm Research) for soil chemical analyses. This research was sponsored by the UK Department for Environment, Food and Rural Affairs and Scottish Government through Sustainable Arable LINK Project LK09128, and we acknowledge the contributions of ADAS, Agricultural Industries Confederation, AHDB Cereals & Oilseeds, Bayer CropScience, British Sugar, CF Fertilisers UK, the Co-operative Group, Country Land and Business Association, Frontier Agriculture, Hill Court Farm Research, NFU, North Energy Associates, PGRO, Renewable Energy Association, Rothamsted Research (North Wyke), Scotch Whisky Research Institute, SoilEssentials, SRUC, Vivergo fuels, Warburtons and Yara UK.

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## 15. Appendix

**Table 60. Overall annual fertiliser rates (kg N ha<sup>-1</sup>) and the distribution of rates by crop type (BSFP 2010 survey data)**

<b>England &amp; Wales</b>	Overall rate (kg N ha <sup>-1</sup> )	% of crop area by field application rate (kg N ha <sup>-1</sup> )													
		<b>0</b>	<b>12.5</b>	<b>37.5</b>	<b>62.5</b>	<b>87.5</b>	<b>112.5</b>	<b>137.5</b>	<b>162.5</b>	<b>187.5</b>	<b>212.5</b>	<b>237.5</b>	<b>262.5</b>	<b>287.5</b>	<b>312.5</b>
Crop		<b>0</b>	<b>12.5</b>	<b>37.5</b>	<b>62.5</b>	<b>87.5</b>	<b>112.5</b>	<b>137.5</b>	<b>162.5</b>	<b>187.5</b>	<b>212.5</b>	<b>237.5</b>	<b>262.5</b>	<b>287.5</b>	<b>312.5</b>
W wheat	194	1	0	2	1	3	3	7	15	19	25	12	7	3	2
W barley	141	2	0	2	2	5	19	25	29	9	2	3	2	0	0
S barley	101	8	1	5	9	16	26	28	6	1	0	1	0	0	0
Rape (oilseed)	200	0	0	0	2	1	1	6	13	22	28	13	8	3	1
Sugar beet	93	5	3	8	10	21	32	20	1	0	0	0	0	0	0

<b>Scotland</b>	Overall rate (kg N ha <sup>-1</sup> )	% of crop area by field application rate (kg N ha <sup>-1</sup> )													
		<b>0</b>	<b>12.5</b>	<b>37.5</b>	<b>62.5</b>	<b>87.5</b>	<b>112.5</b>	<b>137.5</b>	<b>162.5</b>	<b>187.5</b>	<b>212.5</b>	<b>237.5</b>	<b>262.5</b>	<b>287.5</b>	<b>312.5</b>
Crop		<b>0</b>	<b>12.5</b>	<b>37.5</b>	<b>62.5</b>	<b>87.5</b>	<b>112.5</b>	<b>137.5</b>	<b>162.5</b>	<b>187.5</b>	<b>212.5</b>	<b>237.5</b>	<b>262.5</b>	<b>287.5</b>	<b>312.5</b>
W wheat	183	0	0	0	8	3	0	9	14	19	27	21	0	0	0
W barley	155	0	0	1	6	2	12	15	24	27	11	1	0	1	0
S barley	95	2	1	5	14	27	37	13	1	0	0	0	0	0	0
Rape (oilseed)	177	0	0	1	4	11	0	7	18	19	16	19	5	0	0
Sugar beet	93	5	3	8	10	21	32	20	1	0	0	0	0	0	0

**Table 61. Counties and their IDs for the county scale runs and the corresponding crop areas used in WP4 (see Section 9).**

County ID	County	Crop areas (ha)				
		Winter Wheat	Winter Barley	Spring Barley	OSR	Sugar Beet
1	Avon	7447	1312	1813	2220	1
2	Bedfordshire	33808	1651	2141	12093	192
3	Berkshire	14709	2284	2675	5168	18
4	Bucks	31115	2683	1988	9774	11
5	Cambridgeshire	117231	7946	5141	32616	15672
6	Cheshire	14244	5120	3133	2600	55
7	Cleveland	6903	2355	297	2091	1
8	Cornwall Isles Scilly	10094	6870	12330	1162	49
9	Cumbria	6052	6219	12765	676	24
10	Derbyshire	14916	5721	2320	5631	50
11	Devon	22891	8236	12954	3287	47
12	Dorset	24564	4831	8772	9202	7
13	Durham	20309	8174	2106	8195	5
14	E Sussex	10720	816	2288	3569	7
15	Essex	110084	7691	4976	31152	2790
16	Gloucs	32521	4880	7717	13608	31
17	Greater London	2152	172	95	354	0
18	Greater Manchester	1822	736	934	364	5
19	Hampshire	40621	4765	18615	18536	10
20	Hereford & Worcs	52958	6637	4808	14840	769
21	Hertfordshire	37396	3427	2565	10757	846
22	Humberside	107119	23895	11243	39019	3743
23	Isle of Wight	3629	153	531	740	2
24	Kent	60913	4248	2347	21698	22
25	Lancashire	8834	1568	3492	853	49
26	Leics	52979	5982	1601	23837	711
27	Lincolnshire	189917	18783	15528	70711	20510
28	Merseyside	3558	1207	1235	731	9
29	Norfolk	102414	32990	29406	26954	42973
30	N Yorks	103773	39719	16831	34452	3119
31	Northamptonshire	59707	3748	2339	30662	195
32	Northumberland	36007	15470	8780	13975	11
33	Notts	45373	6982	4811	20736	4778
34	Oxfordshire	54896	7394	6803	19706	26
35	Shropshire	45425	11408	7713	12016	1226
36	Somerset	24277	3990	3254	4340	17
37	S Yorks	19523	4999	1802	8148	994
38	Staffs	25481	5686	3254	8297	479
39	Suffolk	101148	18961	9476	27389	18259
40	Surrey	5198	880	691	1630	4
41	Tyne & Wear	3730	1366	515	1609	1
42	Warwickshire	44334	4874	1643	13759	136
43	W Midlands	2938	606	355	1300	109
44	W Sussex	17017	1545	3955	3842	11
45	W Yorks	10961	3611	2220	4756	516
46	Wiltshire	50192	7886	15292	20620	3
47	Clwyd	4110	1557	1630	487	0
48	Dyfed	4973	2308	6775	1373	0
49	Gwent	6186	754	709	1550	0
50	Gwynedd	514	264	1694	10	0
51	Mid Glamorgan	637	368	231	135	0
52	Powys	5396	1374	1418	966	0
53	South Glamorgan	2381	542	352	919	0
54	West Glamorgan	151	113	348	0	0
55	Borders	21233	8124	17706	7387	0
56	Central	2541	637	5147	407	0

County ID	County	Crop areas (ha)				
		Winter Wheat	Winter Barley	Spring Barley	OSR	Sugar Beet
57	Dumfries & Galloway	3108	3145	9573	286	0
58	Fife	16405	4263	15864	2405	0
59	Grampian	17650	19953	95217	12712	0
60	Highland	4116	1007	20369	1274	0
61	Lothian	17950	3809	13899	3901	0
62	Orkney_Isles	36	29	4320	0	0
63	Shetland_Isles	0	0	7	9	0
64	Strathclyde	2459	1597	13443	347	0
65	Tayside	25935	5446	46739	7245	0
66	Western_Isles	3	0	80	28	0