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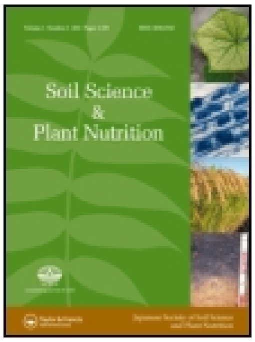

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REVIEW

Nitrous oxide mitigation in UK agriculture

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Abstract

Nitrous oxide (N_2O) makes the single largest contribution to greenhouse gas (GHG) emissions from UK and European Union agriculture. Ambitious government targets for GHG mitigation are leading to the implementation of changes in agricultural management in order to reduce these emissions (mitigation measures). We review the evidence for the contribution of those measures with the greatest mitigation potential which provide an estimated 4.3 t CO_{2e} ha^{-1} y^{-1} GHG reduction in the UK. The mitigation options considered were: using biological fixation to provide nitrogen (N) inputs (clover, *Trifolium*), reducing N fertilizer, improving land drainage, avoiding N excess, fully accounting for manure/slurry N, species introduction (including legumes), improved timing of mineral fertilizer N application, nitrification inhibitors, improved timing of slurry and manure application, and adopting systems less reliant on inputs. These measures depend mostly on increasing the efficiency of N fertilizer use and improving soil conditions; however, they provide the added benefit of increasing the economic efficiency of farming systems, and can often be viewed as "win-win" solutions.

Key words: N_2O, mitigation, greenhouse gas emissions, nitrogen, manure.

INTRODUCTION

Nitrous oxide (N_2O) is a powerful and long-lived greenhouse gas (GHG) with a global warming potential 298 times that of carbon dioxide (CO_2) (IPCC 2007). Agriculture is responsible for 75% of UK N_2O emissions (Skiba *et al.* 2012) and government targets for GHG mitigation therefore rely on the implementation of appropriate management strategies (Moran *et al.* 2011). Emissions of N_2O from agricultural systems are largely associated with the use of nitrogen (N) fertilizers and manures. These generate N_2O as a result of microbial processes going on within the soil to which they are applied (direct emissions), but also as a result of microbial transformations that occur following transport of N away from the site of application by volatilization and leaching (indirect emissions). Direct emissions from fertilizer and manure application and indirect emissions from leaching are the most important agricultural sources of N_2O in the UK (Chadwick *et al.* 2011; Skiba *et al.* 2012).

Nitrous oxide is generated by two microbial processes that occur commonly in soils – nitrification and denitrification – and the organisms responsible are described as nitrifiers and dentrifiers (Baggs 2008). These processes are widespread, and each is favoured by a different set of environmental conditions. However, it is common for both processes to go on simultaneously, making it difficult to attribute emissions to a particular source. There is a well-developed understanding of the extent to which these processes are influenced by controlling variables such as water content, available temperature, soil texture, pH and organic carbon (C) content (Dobbie *et al.* 1999; Flechard *et al.* 2007).

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Our knowledge of the importance of driving variables in influencing N_2O production by soils has led to numerous potential mitigation options (Mosier *et al.* 1998; Burney *et al.* 2010; Luo *et al.* 2010). Across a range of published reports (Mosier *et al.* 1998; Smith *et al.* 2007; Snyder *et al.* 2009), tens or hundreds of individual measures have been proposed. However, this information does not easily help us to identify which measures would be most appropriate for achieving government targets for mitigation, particularly in light of the high degree of spatial variability associated with N_2O emissions.

One method for ranking the appropriateness of different measures is to define the relationship between the biophysical potential (GHG mitigation) and cost, using a Marginal Abatement Cost curve. Using this approach MacLeod *et al.* (2010) identified 10 measures with a combined abatement potential of $4.3\,t\,CO_2\,ha^{-1}\,y^{-1}$ that could be prioritized in relation to C saving and cost. In this paper we review the evidence for mitigation potential by each of the 10 measures and consider their effects on wider environmental and economic criteria.

METHODOLOGY AND RESULTS

A list of relevant literature was developed through a structured combination of a defined systematic literature search and the expert judgment of the authors. First, new reference lists were generated from external, established, scientific databases using defined search terms. These lists were augmented by appropriate "grey" literature sources (e.g., conference proceedings, relevant government- and agency-commissioned studies/reports) and additional relevant references sourced from either existing reference databases held by the project team or supplied by the project's technical advisory group. These lists were then rationalized using the expert judgment of the authors to produce the final reference list for the literature review. This rationalization was necessary to remove the inevitable spurious references that are generated by any search that uses defined search terms. The most common of these were studies from inappropriate geographical areas or of a purely methodological nature.

The baseline against which mitigation measures are assessed is provided partly by the UK's National Report of Greenhouse Gas Emissions (Choudrie *et al.* 2008). However, the methodology used to prepare the national inventory provides little guidance on the likely impact of specific mitigation measures on overall emissions. It was therefore necessary to make assumptions about business as usual projections for farm management that were consistent with existing UK and EU policy in order to define baseline conditions (Moran *et al.* 2008).

Through the literature review process described above, 98 potential N_2O mitigation measures were identified. For many of the potential N_2O mitigation measures, very little research is ongoing and/or very few data exist that could allow an objective analysis of, and comparisons between, measures. Therefore, the authors relied on expert opinion, supplemented by literature where available, to make subjective but informed judgments on:

- the relative temporal potential (current potential, future potential, or speculative potential) of each measure, as well as any uncertainties and knowledge gaps;
- the uncertainty around the estimates of mitigation by that measure;
- whether there is a significant knowledge gap/need for more research before a measure's potential can be assessed.

The next step was to identify, from the large list of 98 potential measures, those thought to have the greatest potential to reduce the amount of N_2O emitted. This task involved an assessment of the retrieved literature on the identified mitigation measures overlaid with expert judgment on the effectiveness of these mitigation measures at a national scale. It should be noted that individual mitigation measures are often reported in the literature on a site-specific basis (i.e., they are based on experiments at a single site or a limited number of sites). In order to upscale the mitigation potential of each measure to a national level, the experts (who are familiar with UK conditions) have made a prediction of its likely national contribution. This approach allowed for a degree of integration of the data from different literature sources to be evaluated, and placed the collected data into a national context. For a topic such as that explored in this project, this approach provides valuable insight into an area where comprehensive data are lacking and absolute values (of national N_2O emissions and their mitigation potential) are unknown. Although it is accepted that experts will not necessarily agree on a particular topic or likely outcome, their aggregated opinions are no less valuable. This approach is well established and has been used by comparable recent studies in this subject area (MacLeod *et al.* 2010; Moran *et al.* 2011). The 10 measures that were agreed to have the greatest mitigation potential are listed in Table 1.

The evidence supporting the selection of each of these measures is briefly described below.

Table 1 The measures identified from the MACC (Marginal Abatement Cost Curve) report (Macleod *et al.* 2010) as having the greatest potential to mitigate nitrous oxide (N_2O) emissions in the UK as selected by an expert group, and expert ranking of uncertainty reported by Moran *et al.* (2008)

Measure	Estimate of measure's abatement rate t CO_2e ha^{-1} y^{-1}	Uncertainty
Using biological fixation to provide N inputs (clover, *Trifolium*)	0.5	Medium
Reduce N fertilizer	0.5	Low
Improving land drainage	1.0	Medium
Avoiding N excess	0.4	Medium
Fully accounting for manure/slurry N	0.4	High
Species introduction (including legumes)	0.5	High
Improved timing of mineral fertilizer N application	0.3	Medium
Nitrification inhibitors	0.3	Low
Improved timing of slurry and manure application	0.3	Medium
Adopting systems less reliant on inputs	0.2	High

N, nitrogen; CO_2e, carbon dioxide equivalent.

REVIEW OF SHORTLISTED MEASURES TO MITIGATE N_2O EMISSIONS

Using biological N fixation to provide N inputs

Biological N fixation provides an input of reactive N to terrestrial systems that can substitute for manufactured fertilizer N (Cassman *et al.* 2002; Erisman *et al.* 2007). Prior to the development of industrial N fixation by the Haber-Bosch process, most of the N provided to support agricultural production originated through biological fixation. During the 20th century an increasing proportion of reactive N inputs to agricultural systems have been provided by manufactured fertilizers (Erisman *et al.* 2008). It has been argued that biological N fixation inputs result in lower losses from agricultural systems as a consequence of the lower rate of addition of N, and a better synchrony of N supply and demand (Jarvis *et al.* 1996; Velthof *et al.* 1998; Gregorich *et al.* 2005). N-fixing crops are widely used in agricultural systems. These crops belong to the family Leguminosae or Fabaceae and are characterized by root nodules which host symbiotic bacteria that are capable of reducing atmospheric N to NH_4^+-based compounds. These include forage crops such as clover (*Trifolium*), lucerne (*Medicago*) and arable crops which in the UK are mainly peas (*Pisum sativum*) and beans (*Vicia faba*). Rates of N fixation vary considerably between crops and different climatic zones.

The magnitude of N_2O emissions associated with N fixation is uncertain. It had earlier been assumed that the nitrogenous enzyme associated with N fixation was responsible for significant emissions of N_2O, and that emissions increased with increasing rates of fixation (O'Hara and Daniel 1989). This was reflected in IPCC inventory calculations which assumed that 1.25% of biologically fixed N was released as N_2O. Experimental studies brought this relationship into question and in 2006 the IPCC revised its estimates of N_2O emission, and the assumption now is that no N_2O emission is directly associated with N fixation (Rochette and Janzen 2005; IPCC 2006). There is, however, a continued assumption that legume residues produced by N-fixing crops contribute to N_2O emissions (Baggs *et al.* 2000; Shelp *et al.* 2000). Again the magnitude of emissions is uncertain, although a number of studies have demonstrated N_2O fluxes resulting from leguminous crop residues that are comparable with emissions reported from fertilizer N addition (Ghosh *et al.* 2002). The UK is currently undertaking a large research program which aims to reduce the uncertainty associated with these emissions and contribute to improved reporting in the national inventory.

The magnitude of emission reductions provided by this measure is uncertain, but would vary according to uptake of the measure on intensively managed grasslands. In many agricultural systems there is potential to reduce emissions by partly replacing N fertilizer inputs with legumes. It was estimated by MacLeod *et al.* (2010) that the annual abatement potential from increased use of legumes would, by 2022, be 0.026 kt N_2O.

Cost/benefit

The lower gross margins associated with grain legume production (when compared with the cereal sector) in the UK are often assumed to be responsible for the relatively low levels of production. A wider evaluation and recognition of the environmental benefits of legume-based farming systems may provide an opportunity for market intervention and policy support in this area. Legume-based forage systems are more widespread within the UK, and are particularly important in organic farming systems. However, likely future increases in costs of fertilizer N being driven up by higher energy costs are increasingly leading to conventional farmers switching to clover-based swards. Further increases in fertilizer costs are likely to lead to a continuation of this trend.

Reducing N fertilization

Reduction in the application rates of fertilizer N is widely recognized as the most effective measure of reducing N_2O emissions. Meta-analyses of relevant experiments have shown that emissions are highly sensitive to the rate of fertilizer application. However, the relationship can be nonlinear (van Groenigen *et al*. 2010; Hoben *et al*. 2011; Pappa *et al*. 2011). Small additions of fertilizer N can result in little change in the N_2O emissions; however, when rates of fertilizer addition exceed the quantity of N required by the crop, emissions can rise sharply. Other studies have shown a more linear response to fertilizer application rates (Schils *et al*. 2008). The recommended rates of fertilizer N used in the UK are based upon the economic optimum rate (which takes into account the N supply provided by fertilizer additions and the soil). This is the quantity of N above which further additions do not result in economic benefit. Although this rate is below the biological optimum (the point at which maximum yield is reached), it does not take account of the environmental impact of modifying fertilizer rates. It has been argued that the calculation of an environmental optimum fertilizer rate should be based upon the quantity of N_2O emitted and nitrate (NO_3^-) leached per unit of grain produced (Hoben *et al*. 2011). Such an optimum rate may well differ from the economic optimum. Making uniform reductions in fertilizer applications (which this measure would require) is distinct from reducing excess applications of fertilizer, since the latter would only affect farmers using more than the recommended fertilizer application. For this reason an overall fertilizer reduction could achieve significant reductions in emissions. It was estimated by MacLeod *et al*. (2010) that the annual UK abatement potential by 2022 would be 0.46 kt N_2O (assuming a 5% reduction in the application of fertilizer N). Larger emission reductions would be achieved by larger reductions in fertilizer N use.

Cost/benefit

Reductions in fertilizer use lead to a direct reduction in fixed costs to a farm manager. The magnitude of benefit, however, will depend upon the extent to which crop yields are reduced. Where N application rates are at or above the optimum rate, the net loss of income is likely to be small in response to small reductions (<10%) in fertilizer applications, but costs would rise sharply with larger reductions in fertilizer application rates.

Land drainage

The release of N_2O to the atmosphere from soils depends mainly on the microbiological processes of nitrification and denitrification. In cultivated soils, at a field or landscape scale, these processes are driven by soil temperature, soil wetness (Smith *et al*. 1998), the addition of N fertilizers and land management (Dobbie *et al*. 1999). Therefore, the emissions are neither spatially nor temporally uniform and will vary with climate and farm type/enterprise. One key aspect for mitigation is the control of soil moisture content through land drainage. Land drainage has been used to improve cultivation in the UK probably since Roman times. Medieval rig and furrow helped to improve aeration in at least part of the field (rig) while modern under-drainage techniques also increase accessibility for farm machinery.

There have been a number of studies undertaken both in Scotland and elsewhere that show the relationship between N_2O emissions and anaerobic conditions induced by soil wetness. Anaerobic conditions promote the release of N_2O by incomplete denitrification (Dobbie and Smith 2006) and can be caused by rises in the water table, restricted downward drainage due to natural and anthropogenic compaction, and transient water-logging due to prolonged or heavy rainfall. Reiners *et al*. (1998), working in Costa Rica, observed that the indirect effect of topography on N_2O fluxes was primarily due to topographic influences on soil moisture contents. Working in eastern Scotland, Ball *et al*. (1997) found increased N_2O emissions associated with micro-topographic hollows. In studying the influence of drainage and texture on N_2O fluxes, Skiba and Ball (2002) reported the greatest fluxes from a sandy loam located in a valley bottom and from imperfectly drained clay loams and sandy clay loams located on level sites.

With respect to N_2O emissions in the UK, there has been much work demonstrating increases in N_2O emissions with increases in soil wetness from field-based experiments (Dobbie *et al*. 1999; Dobbie and Smith 2001; Dobbie and Smith 2003) which gives the opportunity to examine the effect of climate (in particular, rainfall) on emissions of N_2O from Scottish agricultural systems. Smith *et al*. (1998) reported increases in N_2O emissions from the same soils, but over two different years. The greatest emissions were recorded during the wetter of the two years. Dobbie and Smith (2006) found a direct relationship between the height of a perched water table and an increase in N_2O emissions. The perched water table led to an increase in water filled pore space (i.e., increased water contents) within the topsoil above the zone of saturation. The N_2O emissions fell when the water table retreated to below 40 cm, allowing the topsoil to dry. Somewhat in contradiction, Smith *et al.* (1998) found that N_2O emissions declined when the soils were almost saturated. However, it is likely that increased water contents stimulate denitrification in soils that are near to saturation but, as the soil becomes wetter, this process either declines, the soil water physically prevents the gas from escaping (Webb *et al.* 2000) or the denitrification process produces gaseous N_2.

Given that one of the main drivers of N_2O emissions seems to be soil wetness it would seem appropriate to attempt to mitigate losses by improving soil drainage. As Dobbie and Smith (2006) showed, if the water table could be kept to no less than 35 cm below the ground surface, fluxes during the growing season would be reduced by 50%. A reduction in the height of the perched water table to 45 cm below the soil surface could result in an 80% reduction in fluxes. However, even in well-drained soils, transient wetness due to heavy or prolonged rainfall could still lead to increases in N_2O emissions, albeit temporarily (Dobbie *et al.* 1999).

The magnitude of emissions reductions provided by this measure is likely to be significant, but would vary according to the current state of drainage systems. It was estimated by MacLeod *et al.* (2010) that the annual UK abatement potential by 2022 would be 5.84 kt N_2O.

Cost/benefit

The cost/benefit of drainage measures is difficult to assess because of the relatively large uncertainties associated with both the mitigation potential and the potential yield benefits of improving drainage systems. Improvements in soil drainage tend to be associated with high costs, but where drainage systems have become ineffective, the benefits in terms of crop yield are likely to be significant.

Avoiding N excess

Direct N_2O emissions from soil come from the microbially-mediated processes of nitrification and denitrification (Firestone and Davidson 1999). These processes are influenced by factors including soil mineral N content, temperature and moisture content. Consequently, elevated levels of soil mineral N are likely to increase the risks of direct N_2O emissions from soils. Cardenas *et al.* (2010) established non-linear relationships between manufactured fertilizer N application rates and N_2O emissions on grazed grassland, suggesting that N_2O emissions increased when fertilizer N supply exceeded crop demand. Applying fertilizer N from manufactured fertilizers or organic manure applications in excess of crop demand is also likely to increase the risk of over-winter NO_3^- leaching losses (Davies *et al.* 2001) and increase the potential for indirect N_2O losses (Reay *et al.* 2009).

Quantities of N applied from manufactured fertilizers or organic manures should therefore take into account the crop N requirement and the amount of N already available in the soil (the "soil N supply") to ensure no excess. The ability of different soils to supply N to crops is accounted for by UK fertilizer recommendation systems and is a function of soil type, previous cropping and manure use and over-winter rainfall (DEFRA 2010). The recommendation system also advises that N from manufactured fertilizer and manure applications should only be applied to crops when soil N supply is insufficient to meet crop need.

Excess N applications waste money on unnecessary fertilizer applications, cause environmental pollution and can affect crop quality, e.g., lodging in cereals and oilseed rape crops, delayed tuber bulking rates in potatoes and high amino N concentrations in sugar beet (*Beta vulgaris*).

The magnitude of emission reductions provided by this measure is likely to be small, although there is uncertainty about the extent to which excess N applications are currently applied in baseline conditions. It was estimated by MacLeod *et al.* (2010) that the annual abatement potential by 2022 would be 0.93 kt N_2O.

Cost/benefit

Direct N_2O: Not exceeding crop N requirement was estimated to decrease N_2O emissions by ~5%, compared to baseline losses from model farm systems (Moorby *et al.* 2007).

Indirect N_2O: Cuttle *et al.* (2007) estimated that avoiding excess N could reduce NO_3^- leaching from arable land and dairy grassland by about 5%. In addition, ammonia-N losses could be reduced by ~5% as a result of reduced fertilizer (urea) applications.

Fully accounting for manure/slurry N supply

Organic manures applied to agricultural land may be produced on the farm (slurries, farmyard manure and poultry manures) or supplied from other sources such as treated sewage sludges ("biosolids"), composts, digestates and industrial "wastes" such as paper crumble, food industry by-products, etc. Land application is the most sustainable use for organic manures, enabling plant-available nutrients and organic matter to be utilized to help supply crop nutrient demand and maintain soil fertility. Making full use of the N supplied by organic manure applications, and adjusting manufactured fertilizer rates accordingly, is fundamental to minimizing diffuse N pollution from agricultural systems (Chambers *et al.* 2000). However, there are considerable practical issues that make it difficult for farmers to accurately quantify N supplied from applications of organic manures; for example, variable manure nutrient contents, difficulties with ensuring accurate application and the potential for N loss following application (via ammonia and N_2O emissions to the atmosphere and NO_3^- leaching losses to water). This mitigation method details how taking account of manure nutrient contents and minimizing nutrient losses following land-spreading can ensure that the nutrients supplied by manures are fully utilized and losses to the environment are minimized. It differs from the previous measure (avoiding excess N) as it deals solely with how to manage organic manure applications rather than manufactured fertilizer N.

The magnitude of emission reductions provided by this measure is likely to be significant, but would vary according to the uptake of the measure. It was estimated by MacLeod *et al.* (2010) that the annual UK abatement potential by 2022 would be 3.45 kt N_2O.

Cost/benefit

Direct N_2O: Taking full account of the N supplied by organic manures is essential to minimize excess soil N and limit the potential for direct N_2O emissions from soils. Maximizing the N supply from manure applications will also reduce the amount of manufactured fertilizer N applied to crops receiving organic manures. Moorby *et al.* (2007) estimated that taking full account of manure N supply had the potential to reduce N_2O emissions from agricultural systems by ~5% compared with base line levels.

Indirect N_2O: Cuttle *et al.* (2007) concluded that taking full account of manure N could reduce NO_3^- leaching from arable land and dairy grassland by about 5% compared with baseline levels.

Species introduction

Improving NUE is widely recognized as an important mitigation strategy for reducing N_2O emissions (Cassman *et al.* 2002). Ultimately N supply determines yield in all crops and farming systems. Intensively managed croplands regularly recover less than 50% of the fertilizer N applied. The remaining N is then either lost through various pathways (including N_2O emission), or stored in the soil. The extent to which different plant species and varieties recover different amounts of N varies according to intrinsic properties of the plant (root growth rates, N uptake kinetics, etc.) and environmental conditions. Where agronomy is optimized the major gains in NUE are likely to come through genetic improvement (Parry and Hawksford 2010). However, it is important to note that N efficiency has never been a breeding target *per se* in cereal crops. Efficient use of N means both efficient uptake (to minimize fertilizer loss) but also effective utilization within the plant. This will help to reduce losses during the growing season but unfortunately available N left in soil after harvest is another potential source of N_2O loss.

On a worldwide basis NUE in cereals is estimated at 30–50% (Raun *et al.* 2002) although it has been suggested that under UK conditions winter wheat could be as high as 50–60% (Sylvester-Bradley *et al.* 1997). A study of spring barley (*Hordeum vulgare*) varieties bred in the UK over a 75-year period shows that NUE has increased over time; this is associated with an increase in partitioning of N into grain. Foulkes *et al.* (2009) identified a number of traits that could be used by breeders to improve NUE. These include root length density and improve post-anthesis (after flower pollination/fertilization) re-translocation of N from straw. Anbessa *et al.* (2009) suggest that NUE is significantly affected by both genotype and environment. Recent unpublished research (Ian Bingham, pers. comm.) suggests that the amount of available N left in soil post-harvest does not differ significantly between high- and low-input conditions. There is little direct evidence relating to N_2O emissions from different cereal varieties in the field. Significant differences in N_2O emissions between pea (*Pisum sativum*) varieties have been observed in the field (Pappa *et al.* 2011).

Plants which take up N more efficiently have the potential to reduce N_2O losses within livestock-based farming systems (Wilkins and Humphreys 2003). Using varieties that take up N more efficiently can reduce the area needed to produce forage for silage and grazed grass (del Prado *et al.* 2010) thus potentially lowering emissions from the farm system. The magnitude of emission reductions provided by this measure is likely to be small in the near future, but has potential in the longer term (20 years plus) to be important. It was estimated by MacLeod *et al.* (2010) that the annual UK abatement potential by 2022 of species introduction would be 1.23 kt N_2O.

Cost/benefit

Improved use of fertilizer N by crops will have important economic impact on yield and quality. However, if more yield is gained from less N then there may be a need for more phosphorus and potassium to be supplied increasing costs and demand for other fertilizers. However, as the rising price of oil causes increases in the costs of fertilizer-N, it is likely that there will be more pressure to increase NUE and the recycling of organic N resources (Goulding *et al.* 2008).

Improved timing of mineral fertilizer N application

Optimizing the timing for mineral N fertilizers may reduce N_2O emissions by improving the synchrony between N application and crop N uptake (to minimize soil mineral N levels after application). Targeting applications to avoid timings when soils are warm and moist may also have the potential to reduce N_2O emissions.

The best practice guidelines for fertilizer N timing are described in recommendation systems such as the UK Department for Environment, Food and Rural Affairs (DEFRA)'s Fertilizer Manual (RB209) (DEFRA 2010), SAC Technical Notes (Sinclair *et al.* 2009) and the Home Grown Cereals Authority Nitrogen for winter wheat management guidelines (HGCA 2009). In general, for the main arable crop types, it is recommended that the total fertilizer N requirement is applied across two or three separate timings between late February/early March and late April/early May (the period of maximum crop growth). For cut grassland, DEFRA's Fertilizer Manual suggests that 40% of the total N recommendation is applied to the first cut (with 15% In February/March and 25% in April), 35% is applied to the second cut (20% in May and 15% in June) and 25% to subsequent cuts (15% in July and 10% in August).

Previous research has shown that under certain conditions there is a good relationship between soil mineral N surpluses and N_2O emissions at the field level (Schils *et al.* 2008; van Groenigen *et al.* 2008). However, as N_2O emissions in soil are controlled by the microbially-mediated processes of nitrification and denitrification, soil moisture content and temperature in the days following application will also have a bearing on the N_2O emissions that occur after fertilizer N has been applied. Soil moisture in the days prior to fertilizer application can also be important in influencing emissions (Dobbie *et al.* 1999).

Berry *et al.* (2010) suggested that the application of N early in the growing season (i.e., February–March) when soil temperatures are generally low may reduce the potential for N_2O loss. However, this approach would need to be balanced against the greater risk of increased NO_3^- leaching if significant rainfall were to occur following fertilizer application to soils close to field capacity. Berry *et al.* (2010) also suggested that targeting applications when soils are dry would have the potential to minimize N_2O emissions, although the impact of this approach is uncertain because crop N uptake would also be limited under dry soil conditions.

Farmers have limited opportunity to vary the timing of fertilizer applications according to soil moisture and temperature conditions. Ensuring drainage systems are well maintained will encourage soils to dry out as early as possible in spring. Accurate weather forecasting would also be required to enable farmers to choose appropriate conditions to apply fertilizer that may minimize N_2O loss.

The magnitude of emission reductions provided by this measure is likely to be significant, but would vary according to the uptake of the measure. It was estimated by MacLeod *et al.* (2010) that the annual UK abatement potential of improved management of fertilizer N by 2022 would be 3.86 kt N_2O.

Cost/benefit

Direct N_2O: Matching the timing of manufactured fertilizer N applications with crop requirement will make most efficient use of the fertilizer and hence reduce the likelihood of N_2O emissions. In addition, improved timing of fertilizer N application could result in a small (3–5%) increase in crop yield through increased efficiency of fertilizer use (Moran *et al.* 2011).

Berry *et al.* (2010) estimated that if the third N application on all feed wheat (*Triticum aestivum*) crops in England (typically applied in late April/early May) could be applied about 30 days earlier, direct N_2O emissions from fertilizer N could be reduced by 20%.

Indirect N_2O:NO_3^- leaching losses are likely to be reduced if improvements in fertilizer N efficiency are achieved.

Nitrification inhibitors

Nitrification inhibitors (NIs) slow down the first step of the nitrification process [i.e., the conversion of NH_4^+ to nitrite (NO_2^-) and then to NO_3^-] by deactivating the enzyme responsible (Amberger 1989; Di and Cameron 2003). The most common commercially available nitrification inhibitors are Dicyandiamide (DCD) and 3,4-dimethylpyrazole phosphate (DMPP). Initial interest in NIs was mainly concerned with minimizing NO_3^- leaching losses following applications of fertilizer N, livestock slurry or urine returns from grazing livestock by retaining mineral N in the NH_4^+-N form. In addition to reduced NO_3^- leaching losses, reductions in N_2O emissions from both nitrification and denitrification have been observed. Chemicals such as DCD have been evaluated for reducing N losses from autumn-applied slurries for many years, but have generally failed to gain acceptance with the farming community due to their poor cost-effectiveness in terms of giving yield benefits and reduced NO_3^- leaching losses (Chambers *et al.* 2000).

DCD can be applied directly to grazed pastures or following applications of manufactured N fertilizers or organic manures to both arable land and grassland. It is typically applied as a 2% solution at rates of $\sim$10 kg ha^{-1} (500 L ha^{-1}). DCD has also been added to commercial liquid fertilizers and products that can be mixed with livestock slurries before application. DCD contains 65% N so manufactured fertilizer rates should be adjusted to account for this additional N supply. Fertilizer products where DMPP has incorporated into fertilizer prills are also commercially available. DMPP has been shown to be effective at reducing nitrification at application rates of 0.5–1.5 kg ha^{-1}.

The persistence of NIs in soils is likely to be an important factor in their effectiveness at reducing N_2O emissions. UK studies have shown that the majority of N_2O emissions occur within the first 4–6 weeks following application of manufactured fertilizer N and organic manures (Thorman *et al.* 2007b). This suggests that NIs should be applied either with or soon after fertilizer/manure spreading and should persist in the soil for at least two months in order to be most effective at reducing N_2O emissions.

Soil temperatures have been shown to influence the persistence of NIs in soil. Merino *et al.* (2005) suggested that DMPP was most effective at inhibiting N_2O emission at temperatures of between 6 and 11°C because at higher temperatures (>16°C), DMPP was likely to degrade quickly in soil. Workers in New Zealand suggested that that the half-life of DCD at soil temperatures of <10°C was $\sim$85 days compared with $\sim$50 days at 15°C (Kelliher *et al.* 2008).

The magnitude of emission reductions provided by this measure is likely to be significant, but would vary according to the uptake of the measure. It was estimated by MacLeod *et al.* (2010) that the annual UK abatement potential by 2022 would be 1.23 kt N_2O.

Cost/benefit

Nitrification inhibitors have the potential to reduce N_2O emissions from UK farming systems. However, there is still uncertainty regarding the magnitude of the emission reductions under UK agroclimatic zones. Nitrification inhibitors also have the potential to reduce NO_3^- leaching losses from grassland and arable systems; however, further research is required to fully quantify their effectiveness.

Additional product costs have been estimated to range between £10–£50 ha^{-1}, although the costs may be partly offset by reductions in fertilizer costs resulting from improvements in fertilizer N use efficiency (NUE).

Improved timing of slurry and poultry manure application

Livestock slurries (cattle and pig) and poultry manure have high contents of readily-available N [i.e., ammonium-nitrogen (NH_4^+-N); plus, for poultry manures, uric acid N], compared with (straw-based) farmyard manure which is low in readily-available N – i.e., most of the N is organically bound. Matching the application timing of slurries and poultry manure with the period of maximum crop N uptake will reduce the source of inorganic N in the soil that is at risk either of nitrification or denitrification loss, thus reducing direct N_2O emissions. Improved application timing will also reduce indirect N_2O losses, by reducing NO_3^- leaching losses.

As NH_4^+-N is rapidly converted in the soil to NO_3^--N, slurry and poultry manure applications during the autumn or early winter period should be avoided, since there is likely to be sufficient over-winter rainfall to leach a large proportion of this NO_3^- out of the soil before the crop can use it (Chambers *et al.* 2000). Applications later in winter present less of a risk, because low temperatures slow the rate of conversion of NH_4^+ to NO_3^-, reducing the opportunity for direct N_2O losses. Thorman *et al.* (2007a) measured a 50% reduction in N_2O emissions from free-draining grassland soils by changing cattle slurry application timing from autumn to spring. However, soil moisture and temperature conditions at the time of application are important factors in controlling N_2O emissions (Cardenas *et al.* 2010).

In arable rotations, rapid incorporation of slurry and poultry manure before the establishment of oilseed rape (*Brassica napus*), which has a recognized crop N requirement in the autumn, is likely to be as effective as spring top dressing at minimizing N losses.

The magnitude of emissions reductions provided by this measure is likely to be significant, but would vary according to the uptake of the measure. It was estimated by MacLeod *et al.* (2010) that the annual UK abatement potential by 2022 would be 3.45 kt N_2O.

Cost/benefit

Moorby *et al.* (2007) suggested that spreading manures at appropriate timings would reduce N_2O emissions by between 2 and 10%. Spring application timings for slurries and poultry manures will also reduce the risks of NO_3^- leaching losses and increase the efficiency of crop utilization of N supplied by slurry and poultry manure (Chambers *et al.* 2000; DEFRA 2010). Cuttle *et al.*

(2007) estimated that on arable land, NO_3^- leaching losses would typically be reduced in the range of 5–15% by moving slurry applications from autumn to spring, and on dairy grassland by 10–15%. Improved timings for slurry and poultry manure applications will increase the utilization of manure N and reduce the need for manufactured N fertilizer applications to meet crop demand.

Chambers *et al.* (2006) estimated that the capital costs associated with increasing slurry storage capacity to allow spring application timings on pig and dairy farms ranged between £3500 and £5880 per year (annualized over 20 years) to an average farm size of 50 ha. Additional capital costs associated with purchasing slurry band spreader equipment were £3250 per year (amortized over 10 years).

Adopting systems less reliant on inputs

Moving from agricultural systems that rely heavily on external inputs of nutrients and pesticides to reduced input systems may offer a range of benefits in terms of N_2O mitigation. These benefits will come in part from the implicit direct implementation of some of the more specific mitigation measures such as fertilizer reduction. However, what makes the adoption of lower-input systems a measure in its own right is the combination of a number of these specific measures within a single farming system. Such systems, when operating as a whole, may have very different characteristics than a simple sum-of-parts analysis might suggest (del Prado *et al.* 2010), and this makes interpretation of the impact of this measure particularly problematic.

A further challenge when considering this measure is the definition of what is considered a system. Many workers have used this word to refer to what happens inside the physical boundaries of an individual farm, while others encompass the whole production system from the off-farm manufacture of consumables to the sale of final products (life-cycle analysis or LCA), or consider impacts on rural communities. A good example of this is the finding that while the production of a loaf of bread from organic wheat had a significantly lower global warming potential (GWP) than one made from conventional wheat, this advantage was negated by the often greater transport distance to market of the organic product (Meisterling *et al.* 2009).

For the purpose of this review we have assumed that this measure is a movement from a conventional system to a LEAF (Linking Environment and Farming; Leaf 2011) system at the scale of a single farm. The LEAF approach requires farmers to commit to environmental planning and auditing in return for certification by the scheme. This retains the option of applying inputs (e.g., synthetic fertilizers) that would not be possible in a move to a certified organic system, while retaining the flexibility to allow an exploration of the effects of the latter. It does, however, have the disadvantage that "low-input" encompasses a wide range of approaches, and that there is some evidence that low-input, as distinct from certified organic, systems may be no better than conventional systems in maintaining soil organic C stocks (Kong *et al.* 2007).

The magnitude of emissions reductions provided by this measure is likely to be small, but would vary according to the extent of reductions in input. It was estimated by MacLeod *et al.* (2010) that the annual abatement potential by 2022 would be 0.034 kt N_2O.

A general feature of reduced input systems is that they accept possible yield reductions in return for environmental benefits. Mondelaers *et al.* (2009) reported a land use efficiency of 83% for organic compared with conventional farming based on a meta-analysis of 10 studies in developed countries, while de Ponti *et al.* (2012) using 362 studies showed that organic yields of individual crops were on average 80% of conventional yields across a wide range of crops and environments, but variation was substantial (standard deviation 21%). To maintain yields in the face of this possible 10–20% reduction in output, more land area would need to be brought into cultivation. While this introduces other issues that will be explored below, it also means that comparisons between systems face a fundamental choice as to whether system properties are related to a quantity of agricultural production (e.g., per liter of milk) or to the production area (per hectare).

A recent meta-analysis of the differences in environmental impacts between organic and conventional farming in Europe found that emissions of both N_2O and total GHG from organic farming were about 60% of those from conventional farming, on an area basis. However, this difference was not significant when expressed per unit of production (Mondelaers *et al.* 2009). Analysis of the literature selected for the current review (Table 2) supports their conclusions.

Of the eleven relevant studies identified, four found that N_2O emissions on a land area basis were lower from reduced input systems (most commonly organic) than from conventional farming, while two found no difference (Table 2). When compared on the basis of quantity of produce, two out of five studies found that emissions were lower in reduced-input systems.

One of the most common reasons cited for the differences between comparative studies is the often-high variability found in the emissions from reduced-input systems (e.g., Weiske *et al.* 2006). There are several likely causes of this. One is that many of the above comparisons deal with "organic vs. conventional"

Table 2 References and their findings regarding nitrous oxide (N_2O) emissions between reduced input (RI) and conventional systems

Land area comparison	
RI < conventional	Casey and Holden 2005; Stalenga and Kawalec 2008; Mondelaers *et al.* 2009; De Gryze *et al.* 2010
No difference	Syvasalo *et al.* 2006; Chirinda *et al.* 2010
Product Quantity Comparison	
RI < conventional	Casey and Holden 2005; Thomassen *et al.* 2008
No difference	Weiske *et al.* 2006; Mondelaers *et al.* 2009; Snyder *et al.* 2009

farming without distinguishing between, for example, dairy, mixed or arable systems. In addition there is higher inherent variability in the physical structure of reduced-input farms in terms of farm and field sizes, the variety of farming activities carried out and the skill level of individual farmers. These also vary with region as a result of both climatic variations and differences in local farming practices, and the importance of accounting for these has been recognized (e.g., Bareth *et al.* 2001). A further factor is that N_2O emissions are often related to the farm N surplus (Olesen *et al.* 2006; Schils *et al.* 2007) or total N input (Petersen *et al.* 2006). In a conventional system, where the aim is to meet crop N demand by adequate fertilizer addition, the soil N levels are likely to be less variable than in systems where fertility building relies on the more variable release of N from plant residues.

Taken together, these difficulties mean that objective comparisons of fundamentally different farming systems are at best problematic. However, given the number of studies now available, especially those that conduct rigorous meta-analyses, it would seem that on the sole basis of reducing N_2O emissions, adoption of a reduced input system offers potential advantages. However, a whole range of wider issues must also be considered.

Cost/benefit

Given the scale and scope of this measure, it is currently not possible to estimate costs/benefits. More research is needed on the methods to assess fully all costs involved, and to set these against a wide range of benefits.

Conclusions

This review has outlined the evidence for implementing a range of mitigation measures that will help to deliver reductions in N_2O emissions from agricultural soils in the UK. Although this review focuses on the UK, the approaches discussed will have a wider relevance to other temperate agricultural systems. Many of the options discussed are based on implementing best management practices in agricultural systems. While in some circumstances this will happen anyway, increasing the uptake of these measures offers to both reduce GHG emissions and improve the efficiency and profitability of farming enterprises.

In many areas there is a need to improve our understanding of N_2O emissions in response to climatic and management drivers in order to help us build a better inventory of emissions and from which to develop mitigation activities. There is uncertainty regarding the nature of interactions between mitigation measures and the extent to which they contribute to pollution swapping (MacLeod *et al.* 2010). Mitigation potential can show significant regional variability as a consequence of differences in soils and climate. The UK is currently undertaking a large program of work to achieve better understanding of this variability in emissions, with the aim of developing regionally specific emission factors and mitigation potential using IPCC Tier 2 Emission Factors (Chadwick *et al.* 2011).

In the longer term more significant changes involving the implementation of spatially explicit farm management practices (precision farming) that account for the spatial heterogeneity of soil properties, the use of nitrification inhibitors, and the introduction of new plant varieties could further reduce emissions, although effective application of these approaches still requires further research.

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